Proceedings of the

Third Conference on Hydrogeology, Ecology, Monitoring, and Management of Ground Water in Karst Terranes

December 4 – 6, 1991

Maxwell House/Clarion
Nashville, Tennessee

Presented by
the U.S. EPA and the Association of Ground Water Scientists and Engineers,
a division of NGWA

Ground Water Management
Book 10 of the Series

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Abstract

The 1991 Karst Conference was comprised of 2 days of technical presentations, and a day-long field trip.
A total of 46 papers were presented at the meeting.

Sessions were devoted to the following topics:

Ground Water Modeling and Hydrogeology
Site Characterization
Geophysical and Other Techniques for Studying Karst Aquifers
Hydrogeology and Processes Occurring in Karst Aquifers
Case Histories
Ecology of Caves and Karst Terranes
Ground Water Monitoring
Emergency Response and Ground-Water Management
Ground Water Management

The meeting was co-sponsored by the Environmental Protection Agency, and the Association of Ground Water Scientists and Engineers (a division of NGWA).

This bound volume is a compilation of papers that were presented at the meeting.

Materials appearing in this publication are indexed to Ground Water On-Line, the data base of the National Ground Water Information Center at (614) 761-1711.

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Papers published by *Ground Water Management* were presented at conferences sponsored by the Association of Ground Water Scientists and Engineers, a division of the National Ground Water Association.

Staff

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Nick Crawford
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A limited number of copies of the 86-page guidebook can be purchased from: Friends of the Karst, Box 40922, Nashville, Tennessee 37204.

We wish to thank Geary Schindel for coordinating the field trip, and for compilation of the guidebook. We also wish to thank Eckenfelder Inc. of Nashville for its support in preparation of the guidebook, and Grassmere Wildlife Park for allowing the field trip participants to have access to the Park.

The following individuals served as session moderators:

Tom Aley
Dick Benson
Gareth Davies
Ralph Ewers
Jim Quinlan

Most of all, we want to thank the Environmental Protection Agency for partially but very significantly underwriting the cost of the Conference. The Agency's financial support made it possible for more people to attend the meeting and to vigorously participate in it.

Dedication

The National Groundwater Association's Third Conference on Hydrogeology, Ecology, Monitoring and Management of Ground Water in Karst Terranes was held in Nashville, Tennessee, in December 1991. Our field excursion on December 5th took us to the shallow limestone plateaus around Murfreesboro, 30 miles to the southeast. A terrible, futile battle of the American Civil War was fought there in December 1862 and January 1863; no doubt, some very elderly people in Nashville today can pass on stories about it, told to them by oldsters who, as children, heard the guns. It is just two generations away in recollection. Returning to our hotels that evening, we learned that civil war flared again in Croatia, with an artillery bombardment of Dubrovnik, one of the most beautiful limestone cities in the world. This deeply saddened the many of us who have made our pilgrimages to Yugoslavia and come to know and love it.

Yugoslavia (= "land of the southern Slavs") is the birthplace of karst studies in the western tradition. "Karst" itself is a Germanicization of slavic and older words meaning "stony ground" -- describing the rugged, denuded hillsides of Istria, where much of the soil had been swallowed underground into karren pits and into grikes. In Roman times, the regional name was Carsus. As early as 1150 AD the city council of Trieste was promulgating statutes to limit karstic soil erosion -- officers of the modern EPA take their place in a long tradition! Between 1850 and 1910, the first intensive studies of karst geomorphology and hydrology commenced in Slovenia and spread south through Croatia, Serbia, Bosnia, Herzegovina, and Monetenegro. These established the framework of academic karst science, and many principles of karstland management as well, because Yugoslav engineers have excelled in their careful, comprehending development and regulation of the underground water resources.

Today the world karst specialists must travel there in order to measure their particular karsts against the "classical" Dinaric model. When they do so, they become enamored with the peoples and cultures of these beautiful lands. In a regional that is comparable to the Appalachians in length and breadth, there is an astonishing variety of languages and faiths. Driving in the south with my family one day, we encountered 3 different alphabets along one hundred kilometers of highway. Diversity is surely the greatest feature of our world cultural heritage. We know that it can create local stresses, but grieve that these have led to warfare in present-day Yugoslavia as they did in the United States long ago. This volume is respectfully dedicated to all the peoples and cultures of the southern Slav lands. We urge them to resolve their difficulties without further armed conflict.

Derek Ford
January 1992

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Keynote Address:

Protecting Ground-Water in Karst Terranes: An EPA Priority for Action

LaJuana Wilcher,
Assistant Administrator,
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KEYNOTE ADDRESS

PROTECTING GROUND WATER IN KARST TERRANES: AN EPA PRIORITY FOR ACTION

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Washington, D.C.

ABSTRACT

Karst ground waters are unusually sensitive to contamination because they have rapid flow rates, low attenuation of contaminants, and large recharge areas that could contribute contaminants to a receptor. Caves and karst areas contain unusual and unique organisms whose habitats could be degraded before these organisms and their environments can be fully characterized. Each EPA program is increasingly recognizing the importance of pollution prevention as the prescription for protecting karst ground water. This conference is an expression of EPA's attention to the uniqueness of karst ground water resources and to the ecological consequences of failing to properly manage those resources. EPA has multiple ongoing programs and procedures to protect karst ground water, including: 1) Prioritizing karst as highly sensitive to contamination under EPA and State Comprehensive Ground-Water Protection Programs, 2) Detailed characterization, monitoring and analysis requirements for siting RCRA Hazardous Waste Treatment Storage and Disposal Facilities, 3) Recommended protocols for field surveys, ground water monitoring, such as for spring location, 4) Methods for and results of delineating Wellhead and Springhead Protection areas in Missouri and Kentucky, and 5) Health assessment of dyes used for tracing.

TEXT OF ADDRESS

It's a pleasure to join you for The Third Conference On Hydrology, Ecology, Monitoring and Management Of Ground Water In Karst Terranes.

In fact, it is a dual pleasure, personal and professional. On the personal side, my roots go deep into the karst terrane of south-central Kentucky. From the time I was two years old, family outings to Mammoth Cave to see its wonders were a special treat. Add to that school field trips, Girl Scout camping trips, my days as a National Park Service naturalist and cave guide, and my ongoing involvement with the Cave Research Foundation, and you can understand my enthusiasm for the work you're doing.

It's no wonder that I am pleased to be here, for I have been lucky enough to grow up among limestone outcroppings, to watch the River Styx flow from the hillside at Mammoth Cave down to the Green River, and sometimes back again, to rappel 60 feet into the earth, and then tramp, crawl and wade for two hours through the cave's dark passageways, to happen upon an undisturbed pool, and discover in it the magic of an eyeless, translucent, troglobytic crayfish.

Yet, too long we have abused these important parts of our ecosystem, at real damage to the long-term sustainability of the people and the "critters" that share this planet. So I am pleased to participate in a conference that seeks to learn and share information about karst terranes.

Professionally, it's a pleasure to represent the U.S. Environmental Protection Agency as one of the sponsors of this conference, along with the National Ground Water Association. As you may know, the National Ground Water Association, formerly the National Water Well Association, is a well-established organization with a new name. It's a powerful thing, a group's name. It's a beacon to all the world of your goals and aims. This name change reflects the growing understanding in the U.S. of the importance of our sub-surface water resources. It suggests a logical and holistic approach to the development and protection of our ground water. We look to the National Ground Water Association to continue to provide leadership as we move beyond a piecemeal, narrow view of the world to one in which we recognize connections and ecosystems, and focus on relationships—relationships between human activity and the quality of our environment, between surface water and ground water, and among all of the inhabitants of an ecosystem. We must work on behalf on those inhabitants whether they walk on two legs or four; whether they fly, or swim, or slither above or under the ground.

Ground Water and Karst Terranes—A Unique Link

I'm probably preaching to the choir to tell you that even though less than one per cent of the Earth's water is ground water, it plays a disproportionately important role in our lives. In this country, about one-fourth of all fresh water we use comes from the ground. It supplies half of our drinking water, enough for 125,000,000 thirsty people. In eight states, more than 90 per cent of the population depends on ground water for potable and agricultural use. Many Eastern states use nearly a third of their ground water for industry. In my home state of Kentucky, industry uses ground water for nearly 60 per cent of its needs.

With that as background, it's worth asking why we as a nation don't know more, understand more or care more about ground water, and about the karst in which so much of it lies. After all, the characteristics of karst are easily observed, and have been recognized for centuries.

Early Europeans recognized the unique characteristics of karst. Their name for it meant "bare, rocky place." The Germanic word we use comes from the name for the Carso region of Italy and Slovenia, which is blessed with a good deal of soluble carbonate rock.

Karst aquifers underlie about 20 per cent of the continental United States. Karst is found in almost two fifths of the Eastern and Southern U.S., and in much of the American Southwest. And in the West, pseudo-karst in some lava flows mimics many karst characteristics.

The mechanisms by which karst terranes are sculpted has been pretty well understood for well over 100 years in this country. In 1870, Ralph Seymour Thompson described his visit to Mammoth Cave in a delightful account, The Sucker's Visit To The Mammoth Cave ("sucker", by the way, was his friendly term for a resident of rural Illinois.) Thompson wrote: "...a thousand, or a million, or a hundred million, [years ago] as you please—a little stream of water penetrated the soil somewhere in Kentucky, [and] wound its way along through the crevices of the rock... Ages roll away, and still this little stream follows its course...through limestone...dissolving away the rock through which it passes... Slow work it is true; but He to whom a thousand years are but as yesterday when it is passed, has plenty of time. ...[the spring] becomes a great river, with branches in every direction, and draining a great scope of country...."

While some understanding of karst terranes has been around a long time, our behavior in protecting it has lagged far behind our knowledge. We've made dramatic improvements over the last 20 years in improving surface water quality. Why haven't those advances been matched by protection and improvements of our karst aquifers? The reason, I suppose, has to do with human nature. The Middle-Ages philosopher Thomas a' Kempis said it best: "and when he is out of sight, quickly also is he out of mind."

It is like the terrible famines and natural disasters on the other side of the world. Unless we happen across a news story or a TV special, we "tune them out." Karst has been "tuned out" for years. But Nature has been producing several "Specials" that are getting our attention.

The Consequences of Abuse

When Nature puts on a drama, her stage is impressive.

We have seen the land actively rebel against our uses and our abuses of karst terrane. For example, Bowling Green, Kentucky won dubious fame in a 1921 article in Popular Mechanics magazine as "the city with a million year old sewer system." The "system," of course, was an open drain into sinkholes. Out of sight, out of mind. And it wasn't just sewage. Since 1969, chemical fumes from underground were known to drift into homes there. Ten years ago, five houses had to be evacuated because of gasoline fumes rising from caves into their basements. The next year the Lost River Cave beneath Bowling Green was filled with fumes. Cave explorers found benzene and methylene chloride in the underground river and traced them to a chemical mixing facility with leaking underground storage tanks. Bowling Green has had two Superfund emergencies in the past six years relating to explosive or corrosive cave fumes.

And it's not just Bowling Green. A sewage treatment plant in West Plains, Missouri leaked 18 million gallons of effluent into the karst aquifer in May, 1978. The result—at least 759 cases of viral gastroenteritis.

Just three years ago, in October, 1988, four cavers were preparing to descend a passage in Hicks Cave, near Horse Cave, Kentucky. Suddenly a ball of blue flame appeared before them and exploded. The cave ceiling boiled with flame. The cavers ran for their lives, and one was seriously burned. The cause—leakage from a propane gas refueling station half a mile away!

In Puerto Rico in the 1970s, three underground chemical storage tanks ruptured, dumping more than 15,000 gallons of very hazardous chemicals into the karst aquifer below. And that was just the "tip of the iceberg." Pumping to recover the chemicals sucked up over twice as much as had been lost from the rupture. The combined releases to the shallow aquifer forced authorities to deny its use for drinking water.

A Puerto Rican landfill, in use for 20 years, has been found to actually be in a large sinkhole. Everything dumped into it has been immediately transported to ground water. Since the landfill is upgradient of most water wells in the area, it is likely that residents have been drinking their own waste for years.

The caves, windows, sinkholes, fissures and fractures of karst are among nature's most intriguing geological creations. They form a complex puzzle, challenging those of us working to map it, or clean it up. Discrete flow paths and the changes in flow caused by storms makes it very difficult to pinpoint ground water in karst, let alone a slug of contamination in it.

Because of these problems, it is especially important that the Agency embrace a new way of doing business in karst terranes; a new direction. As United States Supreme Court Justice Oliver Wendell Holmes said, "I find the great thing in this world is not so much where we stand, as in what direction we are moving:...we must sail sometimes with the wind and sometimes against it—but we must sail, and not drift, nor lie at anchor."

When I think of directions, I remember the story of an old man, travelling by train. The conductor came by, collecting tickets. The old man rummaged through his pockets but couldn't find his ticket, so the conductor suggested that he mail it in later. The man looked at him sternly and said, "Sir, the question is not where is my ticket. The question is, where am I going?" The question for us is, where is EPA going with regard to protecting karst?

EPA's Role in Karst Protection

For a number of years, EPA's direction sometimes conflicted with those of the states concerning ground water and karst protection. Today I'm happy to say that EPA's Office of Water, through its Ground Water Division, is providing leadership in charting our course toward new policies and more focussed attention upon karst protection. The help and the leadership of people like Phil Berger and Ron Hoffer in EPA's Office of Ground Water and Drinking Water, and Ron Mikulak in our Regional Office in Atlanta, is getting us moving in the right direction.

Our efforts signal a new, comprehensive approach to environmental protection. Through various program offices, EPA administers five different statutes containing ground water protection language: the Clean Water Act, the Safe Drinking Water Act, the Resource Conservation and Recovery Act, the Comprehensive Environmental Response, Compensation and Liability Act (Superfund), and the Federal Insecticide, Fungicide and Rodenticide Act. Every one of the Agency's program offices, from pesticides to hazardous wastes, is working together on a coordinated approach to ground water protection. The Office of Water is pleased to be leading this innovative effort.

EPA Administrator Bill Reilly recognized the importance of such efforts recently when he released EPA's new <u>Ground Water Strategy for the 1990s</u>. The Report re-states our strong belief that states are in the best position to protect their own ground water, and it sets a new policy direction for the Agency to work closely with the states to develop and implement Comprehensive Ground Water Protection Programs. We want to ensure that karst has the highest priority in those state programs. EPA will help states fill any gaps they might find, in order to ensure comprehensive ground water protection. Our staff is working with every state and national organization representing state managers involved in ground water issues. We intend to produce national guidance next year on Comprehensive State Ground Water Programs.

Our Wellhead Protection Program includes technical mapping and model protective ordinances. We are actively working on ways to help states designate wellhead protection areas in karst. In fact, we have two case studies, one in Missouri and one in Kentucky, that will serve as models.

Our Safe Drinking Water Act Surface Water Treatment Rule and forthcoming Ground Water Disinfection Rule will have significant implications in karst as the public water supply systems comply with setbacks and separation distances between rural wells and septic tanks. We need you to work with EPA to help us understand what the rule should provide for the special conditions in karst. The Office of Water is promoting evaluation of "Class Five" or, shallow, Underground Injection Wells, to minimize effects on karst aquifers. Septic tanks, agricultural and storm water drains, including "improved" sinkholes, should all be evaluated. EPA has been pleased to fund the work of Al Ogden of the Tennessee Technological University and his colleagues on the effects of these wells on ground and surface water quality in karst areas.

But we have a lot of catching up to do. EPA's Office of Solid Waste and Emergency Response has listed an incredible 160 hazardous waste treatment, storage and disposal facilities on karst terranes in the United States! As you well know, hazardous waste spills into karst are a remedial nightmare, and complete remediation is often not possible at all, especially considering the effects of the subterranean eco-system. Contaminant flows of 25 miles in 12 days have been documented. And the costs of attempted clean-ups can be astronomical. EPA estimates the cost of remediating a hazardous waste spill in karst at between $7,000,000 and $1.2 billion per incident, not including the environmental costs of land lost or habitat damaged. EPA is now developing a proposed rule mandating strict siting criteria for hazardous waste treatment, storage and disposal facilities in karst. That rule should be out sometime next Spring. In the meantime, as of 1987, nine states had acted on their own to ban such facilities in karst, two states had mandatory setback distances from karst areas and one state, Kentucky, required a double-liner and leak-detection system or a demonstration that karst features were sealed and filled. 24 states have siting criteria that may restrict such facilities.

In addition, EPA's new municipal landfill rule, issued just two months ago, creates the first meaningful national solid waste landfill siting standards. As of October, 1993, developers must demonstrate engineering standards to maintain the structural integrity of a facility sited in an area of disruption. The methods may include re-routing surface water and a demonstration that karst features are sealed and filled.

But our regulations are only as good as our science. EPA's Office of Research and Development has assessed the risks and benefits of using ground water dyes as tracers so that we can predict the fate and transport of hazardous wastes leaked into karst ground water at Superfund sites. It now appears that many dyes can be used safely if they are used properly. Our final risk/benefit study may allay the concerns some states have about dye-tracers. The work of EPA scientists Malcolm Field, Ron Wilhelm and Charles Auer will be presented at this conference, to give you a more detailed picture of what we're doing in this regard.

And there's more. EPA's Office of Pesticides and Toxic Substances is promoting State Management Plans for pesticides, tailored to differing local pesticide use and ground water vulnerability. Increased monitoring will be required in areas such as karst which are particularly susceptible to contamination.

And Congress is getting the message that ground water in karst areas needs special protection. Late last year, Congress passed the Food, Agriculture, Conservation and Trade Act of 1990, the Farm Bill, which included a new program to provide assistance to farmers to implement on-farm water quality protection plans. The areas of attention include those "in shallow karst topography areas where sinkholes convey runoff water directly into groundwater." Next year, when Congress again debates re-authorization of the Resource Conservation and Recovery Act, it will have on the table a bill, introduced by Senator Max Baucus of Montana, which includes proximity to karst terrane as a criterion when siting landfills.

While we are making great strides, we have much more to do.

A Public Role in Karst Protection

Government has an important role in ground water protection, but government alone is not going to do the job. We need to enlist <u>everyone</u> in this battle. EPA has chosen to do that by promoting preventive and corrective actions <u>before</u> the environment is damaged. Pollution prevention is especially vital in karst terranes in order to prevent contamination in the first place, rather than trying to clean it up later.

In the case of karst, pollution prevention means wellhead protection programs and comprehensive ground water protection plans. It means identifying potential point and nonpoint pollution sources, and it means setting priorities for addressing those pollution sources. It means geologic and hydrologic mapping and water quality characterization.

But to implement pollution prevention effectively, we must reach out to people and businesses about causes and effects—about environmental stewardship. At the bottom line, those who generate pollution—and that means every one of us—must take action to keep it in check. People need to know about the connections between ground and surface water, especially in karst terranes. We have to get the word out that people who dump into sinkholes might just as well dump into the nearest river.

As part of that process we must learn more and educate the public more about the non-human subterranean environment. We know that caves and karst areas contain unusual and unique organisms—things like the Alabama Cavefish, the Kentucky Cave Shrimp and the Comanche Springs Pupfish. Their habitats continue to be needlessly contaminated, wiping out populations and whole species.

Our Ecological Responsibility

We need to act on the advice of EPA's Science Advisory Board last year, when it said that we should "...attach as much importance to reducing ecological risk as ...[we do] to reducing human health risk, because productive natural eco-systems are essential to human health and to sustainable, long-term economic growth, and because they are intrinsically valuable in their own right..."

While subterranean critters are valuable in their own right, we also know that they can help us by showing us the health of ground water. Our holistic approach to improving water quality has led us to begin developing biological indicators of the health of surface water, and we must develop similar indicators to measure the health of ground water. And, as we learn, we need to do our characterizations and evaluations with the least possible intrusion, so that other life-forms are as little affected by human activities as possible.

To promote understanding of the eco-system and development of biological ground water indicators, we are sponsoring the First International Conference on Ground Water Ecology, April 27th through the 29th, 1992, in Tampa, Florida. We intend it to be the <u>other</u> must-do ground water event on your calendar.

The Challenge

Our job, and our challenge, is to learn, to teach, and to protect. As the African environmentalist Baba Dioum so eloquently said, "For in the end, we will conserve only what we love, we will love only what we understand. We will understand only what we are taught."

Our task is not an easy one, but we must not fail! I assure you that we in EPA have a clear sense of the importance of our goals, and that we will be working right along with you to expand the volume of our knowledge. I wish you an exciting learning experience here at the conference, and success in the important work you have ahead of you.

BIOGRAPHICAL SUMMARY

LaJuana S. Wilcher, the Environmental Protection Agency's Assistant Administrator for the Office of Water, is responsible for planning, policy and national program management of EPA's water programs. Under her leadership, the Office of Water has intensified its ecosystem protection efforts, complementing its human health protection activities. Wilcher emphasizes wetlands and aquatic habitat protection, a stronger scientific basis for regulatory decision making, pollution prevention, and geographic targeting. Prior to her appointment as Assistant Administrator, Wilcher was a partner in a 90-lawyer Washington, D.C. law firm. She specialized in environmental litigation, legal counseling, and regulatory interpretation of water, air, hazardous waste, and resource recovery issues. Her background includes extensive legal and federal government experience and work as a naturalist and cave guide at Mammoth Cave, Kentucky. Wilcher holds a B.S. in biology (magna cum laude) and a Juris Doctorate.

QUESTION:

1. If we are protecting aquatic life in karst aquifers, should we apply ambient water quality criteria (AWQC) as ground water protection levels instead of maximum contaminant levels (MCLs)? {AWQC are to protect aquatic life. MCLs are to protect human life.}

ANSWER:

EPA's aquatic life criteria methodology is based on demonstrated adverse effects of particular toxicants on a wide spectrum of aquatic plants and animals, including sensitive fish species. Therefore, in order to protect aquatic life no matter where located, e.g., above ground or below ground, EPA's recommended aquatic life criteria are the criteria of choice.

However, if both aquatic life beneficial uses and public water supply uses are to be protected, the more stringent of these criteria should be applied. As a general rule, the aquatic life criteria are more stringent than MCLs for non-carcinogens. For carcinogens, MCLs are more stringent.

Please address questions regarding which criteria apply as a matter of State or Federal law to the appropriate State Agency or to EPA's Office of Water.

QUESTION:

2. In the light of unique problems associated with karst areas and the poor performance record of most regulatory agencies in recognizing them and dealing with many of them as judiciously as might be desired, would it not be desirable for EPA to set up a specific program to review environmental problems in karst terranes? Such a program could include procedures for or assistance with identification, evaluation, and remediation of karst-related problems. If not, why not?

ANSWER:

While there is no question that the unique nature of karst and the problems associated with it merit special attention, it does not necessarily follow that new programs must be established in order to attract that attention. EPA's Office of Water, through its Ground Water Division, is charting a course toward new policies and more focussed attention upon ground water protection, including that in karst terranes. Every one of the Agency's program offices, from pesticides to hazardous wastes, is working together on a coordinated approach to ground water protection. The Office of Water is pleased to be leading this innovative effort. It is true that we must do more to understand and protect karst terranes, and EPA intends to be in the forefront of such efforts.

Invited Lecture:

Ground Water Modeling in Karst Terranes: Scale Effects, Data Acquisition and Field Validation

*Georg Teutsch,
Institut fur Wasserbua, Universitat Stuttgart, Germany
and Martin Sauter,
Universitat Tubingen

GROUNDWATER MODELING IN KARST TERRANES: SCALE EFFECTS, DATA ACQUISITION AND FIELD VALIDATION

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ABSTRACT

Karstified limestone aquifers are often a very prolific source of groundwater and have therefore been studied in many countries to a great extent. In recent years the focus of many studies has switched from 'water resources development aspects' towards a 'water resources protection' point of view. Consequently, the demand for quantitative contamination assessment has increased. In contrast, groundwater flow in karst terranes is still considered hardly quantifiable. Predictions, based on classical hydrogeological investigation methods seem to fail quite frequently; similarly most attempts to employ standard mathematical models did not prove satisfactory so far.

This paper presents a recently developed concept for the consistent classification of groundwater flow problems in karst terranes which proved helpful in understanding some of the 'unexpected' hydraulic phenomena observed. This concept is based on a scale hierarchy approach originally used to describe highly heterogeneous porous formations. Within this concept the spatial averaging properties of different investigation methods like pumping tests, tracer tests and spring flow measurements are related to the size of the prevailing heterogeneities and the size of the investigated domain. Similarly, the applicability of various proposed modeling approaches is analysed, taking into account the type and matureness of the respective karst system.

To demonstrate its practical application a case study from the Swabian Alb limestone karst aquifer in Southern Germany is briefly presented. In this classical karst research area a considerable amount of hydrological, hydraulic and tracer test data has been gathered over the past 10 years and analysed using single- and double-continuum flow and transport models of different complexity.

INTRODUCTION

In general, the effective primary porosity of carbonate rocks is considered to be negligable. Consequently, in karst terranes the effective porosity is a result of a combination of secondary porosity fissures and also a tertiary porosity in areas where the fissure system has been enlarged to conduits by chemical dissolution. As a result, groundwater flow in karst aquifers may show in some areas a diffuse-type and in other areas a more conduit-type flow characteristic. This is confirmed from field observations where hydraulic conductivity investigations conducted at adjacent locations may produce very different results. Similarly, measurements from the same location (e.g. one borehole) but taken at different scales (volumes of integration) may yield very different hydraulic conductivities.

In order to develop a more consistent understanding of karst terranes and their variability, various attempts were made employing classification schemes which are based on (mainly qualitative) descriptions of the various flow phenomena observed in the field (e.g. White 1969, 1977; Thrailkill, 1986). In a more recent study, Smart and Hobbs (1986) developed a scheme where also the recharge distribution and the storage properties of the system are included. Although very useful in providing a conceptual framework for the description of various observations made in karst areas, there still remains a need for a more quantitative analysis of the field data commonly available. This quantitative flow analysis may be based either on physical models (e.g. Snow, 1965; Irmay, 1964; Romm, 1966; Long et al. 1985; Smith et al. 1987, Teutsch, 1988; Kraemer and Haitjema, 1989) or a transfer function approach (e.g. Duffy and Harrison, 1987; Aiguang et al., 1988; Simpson, 1988; Dreiss, 1989a, 1989b).

In general the physical model is preferrable to the transfer functions (black box) approach which commonly describes only a single-parameter input-output relationship. Due to the non-physical basis, these transfer functions have only very limited predictive properties. On the other hand, physical models may imply considerable parameter identification problems when applied to practical field conditions in karst terranes, where hydraulic measurements are usually scarce and highly variable.

It should be noted, that in karst areas the major parameter identification problem results from the fact that data is not available at the scale of the flow dominating heterogeneities, i.e. at the scale of the main conduits. Up til now, there is no standard investigation method available which permits the reliable detection of the major conduits within a given catchment area. The chance to encounter a conduit in a borehole is extremly small if no surface indication like a sinkhole or a dry valley is available. Bearing in mind that in a natural karst system the hydraulic conductivity contrast between a hardly karstified zone and a fully developed conduit system could be as high as 5 to 7 orders of magnitude, the attempt to try and quantify groundwater flow in a karst catchment appears almost impossible at first.

On the other hand, due to the high risk of groundwater contamination in karst areas there is a growing demand for reliable protection measures. This has lead to a series of studies which were initiated in 1981 at the University of Tübingen (Behringer, 1988; Teutsch, 1988; Merkel, 1991; Sauter, 1990; Sauter, 1991 - this issue) and also at the University of Stuttgart (Teutsch, 1989, 1990b; Lang et al. 1991a, 1991b). The aim of these studies is the development of a consistent methodology for the investigation and quantification of karst flow systems including physical modeling. The area of investigation is the Upper Jurassic limestone aquifer of the Swabian Alb. The aquifer is by far the most prolific source of groundwater in South Germany and is extensively used for the municipal water supply in that region.

Figure 1 shows the Swabian Alb outcrop with the various regions selected for detailed investigations. Region 2, 3, 4 and part of region 5 belong to the socalled 'Deep Karst' whereas region 1 and part of region 5 belong to what is called the 'Shallow Karst'. The term 'Deep Karst' refers to the area of the aquifer which drains to the river Danube. There the piezometric levels do not reach the aquifer bottom, the piezometric surface being controlled by the topographical level of the surface streams. In the 'Shallow Karst' the original drainage pattern to the river Danube has been deviated during the Quaternary as a result of the erosional processes caused by the river Rhine tributary system. There the aquifer piezometric levels are controlled by the position of springs which developed at the aquifer base just above the underlying aquiclude. Because of different aquifer histories the observed flow characteristics are quite different in the 'Deep Karst' and in the 'Shallow Karst'. It is generally believed that in the 'Deep Karst' a diffuse-type system dominates the flow, whereas in the 'Shallow Karst' the original diffuse-type flow system has matured to a conduit-type flow system during the recent geological past.

Applying the scale hierarchy concept presented below, the range of field findings obtained from different investigation methods is discussed.

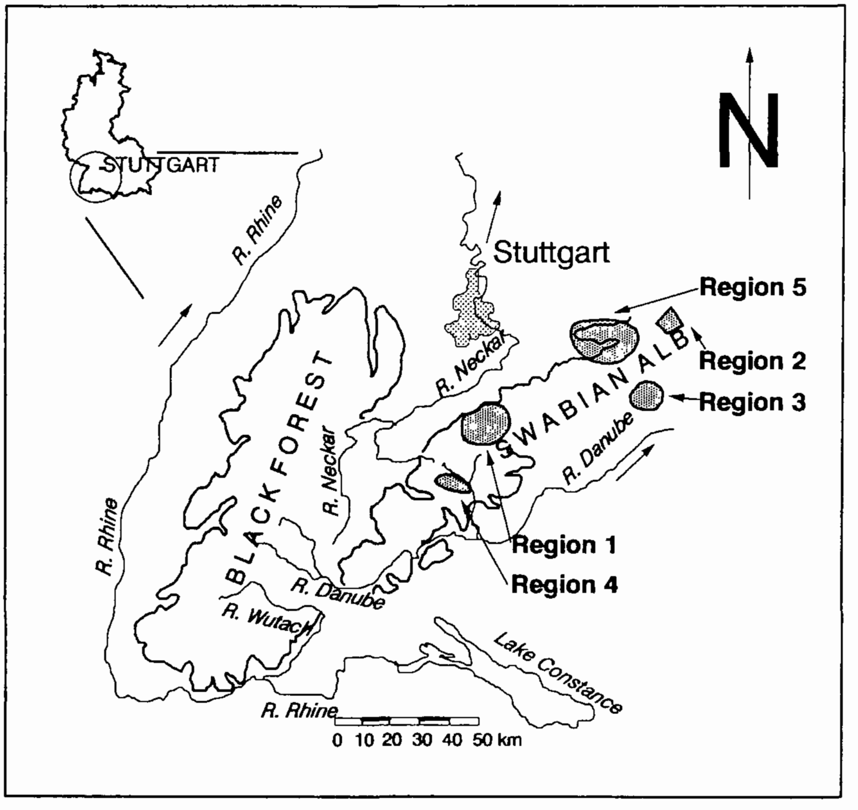

Figure 1: Swabian Alb - Areas of Investigation

THE SCALE HIERARCHY CONCEPT

The scale hierarchy concept as developed by Haldorsen (1986) proved to be extremly useful in the analysis of flow and transport problems in heterogeneous formations (Teutsch, 1990b; Teutsch et al., 1990). This paper describes its extention to the analysis of fractured and karstified terranes where a scale dependent heterogeneity is commonly observed.

Employing the terminology introduced by Dagan (1986), three different scales are to be considered:

a) L, the length-scale of the flow or transport domain

b) I, the length-scale of the flow dominating heterogeneities

c) D, the length-scale (averaging scale) of the detection method used

Figure 2 shows a schematic example of a heterogeneous system with the I, L and D length scales. The example might be viewed as a simplified cross-section through a fractured or karstified aquifer system, where L could be the size of the catchment, I the average length of the fissures and D for example the size of the drawdown cone during a pumping test. The heterogeneity of the system is extremly high at the scale of an individual fissure but certainly far less visible at the larger scale D of the pumping test. Therefore, any movement of the detection window D, i.e. conducting the pumping test at different locations, would lead to 'smoothly' varying parameters but not to an abrupt change in the values. Consequently, due to the size of the detection window one would not expect to locate individual fissures or conduits. On the other hand, any more selective, i.e. a smaller scale, detection method like for example a borehole flowmeter-log or a point-source artificial tracer test would produce a much higher parameter variability.

For our example the scale hierarchy may be described as I < D_{pmp} < L. Most conveniently, it can be shown that many flow problems may be consistently classified on the basis of the scale hierarchies only, i.e. I, L and D ratios without the need for absolute figures. From the above it is quite clear, that a laboratory analysis of micro-fissures at core scale may yield the same variability of results as a catchment scale investigation dealing with major conduits, possibly at the kilometer scale.

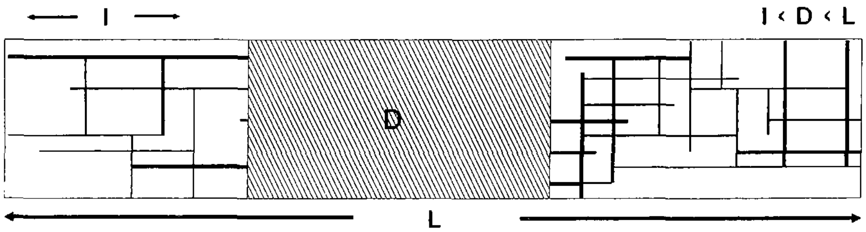

Figure 2: Schematic example of length scale hierarchy (I: heterogeneity-, D: detection-, L: domain-scale)

The scale hierarchy concept as described above can be used to analyse various hydraulic phenomena observed in heterogeneous systems like karst terranes. In the following paragraph a few examples of commonly observed karst aquifer features are discussed applying the framework of the scale hierarchy concept.

TYPICAL FIELD OBSERVATIONS

The field examples presented below are grouped into hydraulic phenomena ranging from regional to local scale and transport phenomena based on areal and point source input.

Hydraulic Phenomena

a) Large scale hydraulic tests

In region 2 a large scale pumping test was conducted in 1982. Region 2 is located in the 'Deep Karst' north of Heidenheim. Due to a limited shallow karstification, the vast majority of the groundwater flow is believed to occur in a network of fairly narrow fissures, continuous at a scale larger than the equivalent elementary volume, probably in the range of 100 meters. The log-linear distance-drawdown (steady-state) relationship as observed at the end of the pumpingtest shows a straight line response of the karst aquifer wells. This reflects homogeneous subsurface conditions at the given detection scale. This finding was further supported in a subsequent regional modeling study where ground water levels from 26 observation wells were adequately simulated employing a single-continuum porous equivalent model. The scale hierarchy for this problem would be I << D <= L.

b) Spring flow hydrographs

Figure 3 shows the hydrograph from the Echazquelle in region 1 which is located in the 'Shallow Karst' where flow is believed to be dominated by large conduits. Typically for a karst spring in that area, it shows a high Q_{max}/Q_{min} ratio and at the same time a sustained recession of about 6 months during the summer. This observation is inconsistent with a simple single-continuum concept. A conduit-type of flow would explain the high Q_{max}/Q_{min} ratio, but could not explain the long recession and vice versa. It is important to realize that spring hydrographs reflect the response of the entire flow system to areal recharge events, i.e. they describe the spatially averaged unsteady-state characteristics of the aquifer at catchment scale. The scale hierarchy for this flow problem may be described as I <= D = L, i.e. the system heterogeneity is visible at the detection scale. However, because the detection scale is equivalent to the catchment scale, the distribution of the hydraulic conductivity within the catchment is not detectable.

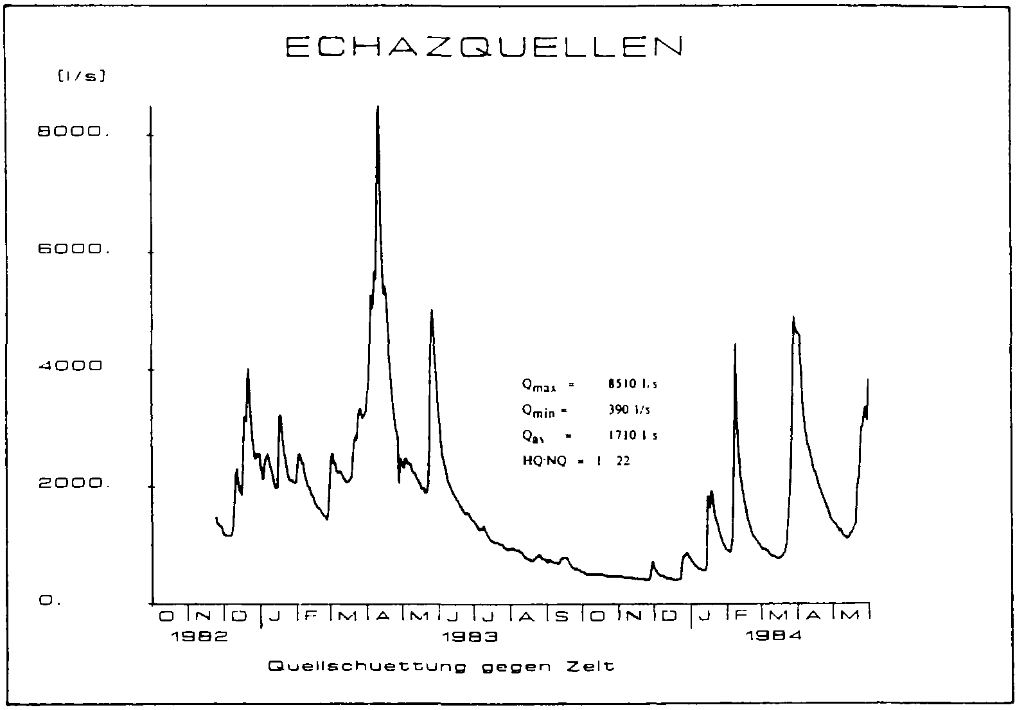

Figure 3: Echazquelle hydrograph for the period Nov. 1982 to May 1984 (Teutsch, 1988)

c) Groundwater level measurements

The analysis of piezometric levels with respect to their spatial distribution and temporal variation provides important information on the aquifer characteristics. This is especially true for any heterogeneous system like a karst aquifer. Figure 4 shows the average piezometric levels as measured in region 5 which is located partly in the 'Shallow Karst' and partly in the 'Deep Karst'. The map appears fairly structured with the major (european) water divide extending in NE-SW direction. Surprisingly, the location of the major spring outflow regions in the 'Shallow Karst' can be clearly identified on the piezometric map. In these areas the groundwater flow is concentrated (conduit-system), leading to a convex shape of the equipotential lines. This example shows that groundwater level readings, even though being taken at single points (boreholes), may represent an area much larger than the actual observation point. In other words, the piezometric surface varies only gently between individual observation locations. Therefore, the hydraulic effect of single conduits is far reaching and may be observable without having to tap them directly by boreholes. The scale hierarchy for this problem is therefore I < D < L, where I represents a combination of small fissures and large conduits.

Another way to look at groundwater levels is to analyse their temporal variation. Figure 5 shows the groundwater levels versus time as observed in one of the boreholes within region 5. The first part of the curve is based on weekly readings only. After the 4th of August 1988, an automatic recorder system was installed in the borehole. The resulting change in the characteristics of the water level record is dramatic. The high time resolution of the automatic recording reveals the high frequency components on top of the smooth seasonal water level variations. This 'dual-frequency' feature can be attributed to the 'dual-porosity' characteritsic of the karst system in that area. This example demonstrates clearly the importance of the time domain resolution as compared to the spatial resolution discussed above.

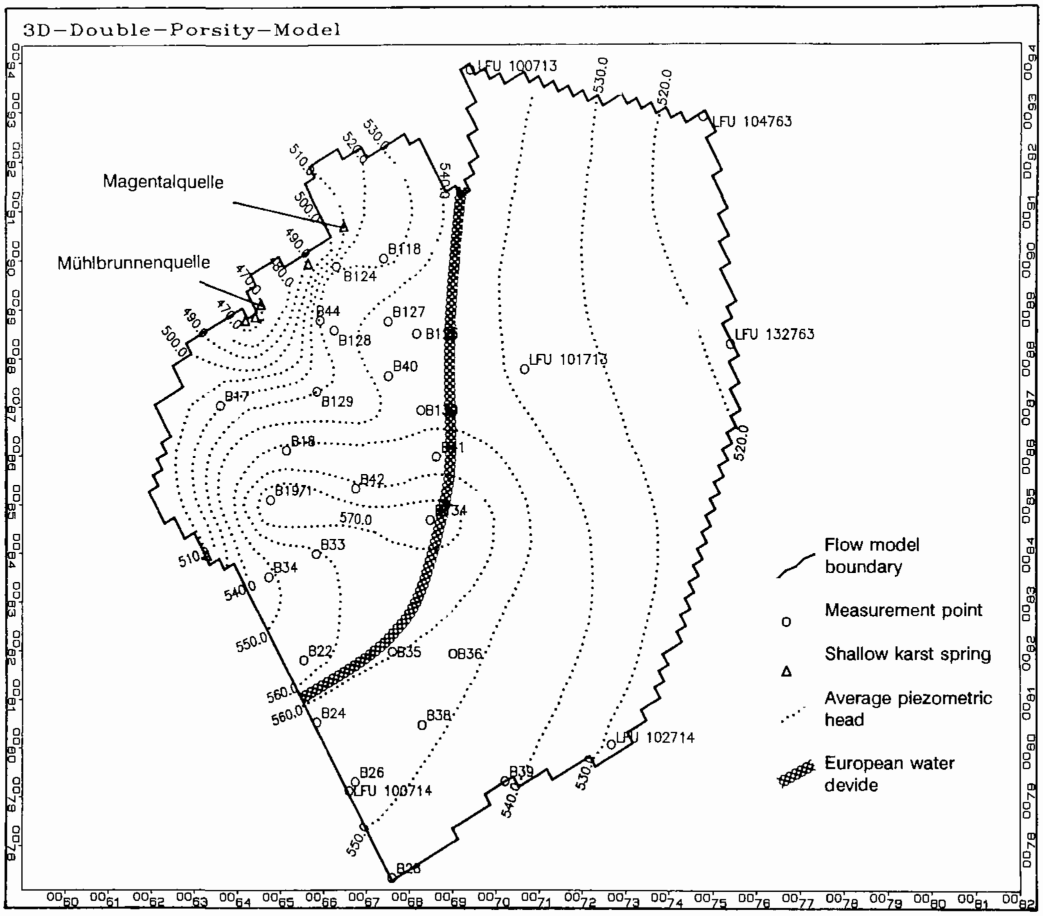

Figure 4: Groundwater level map - 'Stubersheimer Alb' (Lang et al., 1991a)

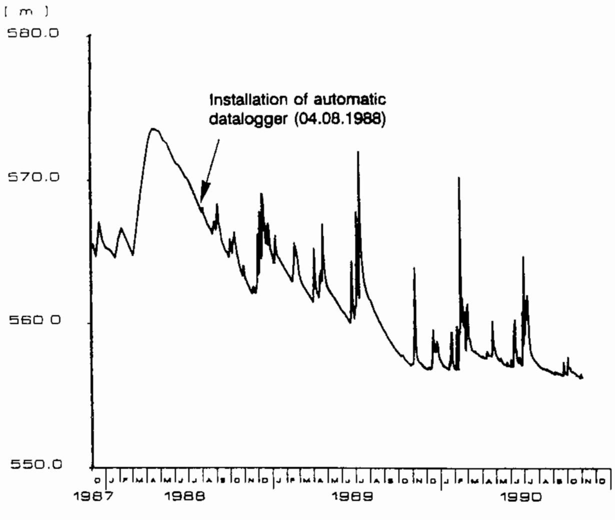

Figure 5: Groundwater levels at B33 with and without automatic recording (Lang et al., 1991a)

d) Small scale hydraulic tests

Figure 6 displays two types of plots of the recovery data of slug tests in the Gallusquelle catchment (region 4). A log-log plot of test B8_3 (Sauter, 1991), which clearly reveals 'double-porosity' response during early times, and an analytical plot used for parameter evaluation according to the procedure of Bouwer et al (1976). The analytical plot depicts three sets of recovery data from the same borehole. The three tests only differ in the initial displacement and the resulting hydraulic conductivity decreases with increasing initial displacement. This feature demonstrates very clearly the particular heterogeneous characteristics of a double- porosity system, which have also been described by Streltsova (1988). If the well is directly connected to the fissure-conduit system, short tests will only reveal the response of the highly permeable conduits, whereas the longer the test (i.e. the larger the displacement), the more will the test data reflect the reaction of the entire system including the fissures. The radius of investigation, which can be determined according to Ramey et al. (1975) and Sageev (1986) to range between 10 and 20 m, increases with increasing initial displacement. The scale hierarchy for this flow problem can be described as $I <= D_t << L$ at times $t = 1, 2, ..., n$ with $D_1 < D_2 .... < D_n$.

Increasing the displacement depth even further leads to an assymptotic levelling off of the hydraulic conductivity value. At this scale, the system responds almost homogeneously.

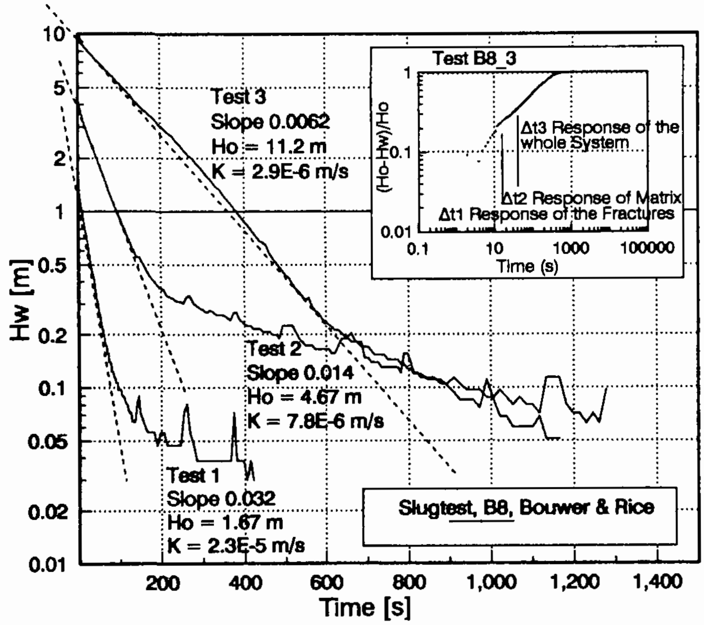

Figure 6: Slugtest analysis at B8 (Sauter, 1991)

Transport Phenomena

e) Areal source tracer tests

It has been recognised for some time that spring water quality variations provide valuable information on the flow system within a karst aquifer. They have been frequently used to evaluate aquifer characteristics and groundwater velocities and to distinguish between different types of springs. They also allow the distinction between event and preevent water (Sklash and Farvolden, 1979) and between the fast and the slow flow component (Dreiss, 1989a). The analysis of the variation of these parameters also provide information on the role of parts of the whole system, e.g. the epikarstic zone (Williams, 1983).

Figure 7 depicts time series of hydraulic and physico-chemical parameters during a four monthly period, commencing with April 1989, observed in region 4 (Sauter, 1990). The spring flow, groundwater hydrographs and the water quality variations suggest that two distinct flow systems exist. Some of the parameters such as spring discharge integrate over the whole system, others are more selective for the fast system (turbidity, electrical conductivity) and some provide information on both systems (groundwater hydrographs, $\delta^{18}O$, temperature).

Spring water turbidity, frequently characterised by a bimodal breakthrough, is indicative of the fast water component, because a minimum hydraulic energy is required for the clay particles to remain in suspension. The first peak is induced by high flow velocities due to the initial pressure pulse, which also is responsible for the rapid increase in discharge, and the second by the actual fast water component arriving at the spring. Similarly, electrical conductivity changes can be used to identify fast water components, because the chemical reactions of the carbonate system are very rapid, so that slow water components cannot be distinguished from preevent water.

Spring water temperature variations are indicative of fast as well as slow water, because the temperature can be considered as a more or less conservative parameter. Temperature variations can be traced back to a particular event even after three months. The relative abundance of oxygen isotopes can be employed as well as a regional tracer and displays a bimodal breakthrough after recharge events. The respective time series however reveal very complex patterns due to the superposition of several events and the variations in input $\delta^{18}O$. Details concerning this particular time series are discussed in Sauter (1990).

Obviously, the scale hierarchy for spring flow chemographs is usually $I << D = L$ for the fissure system and $I <=> D = L$ for the conduit system.

f) Point-source tracer tests

Point source tracer tests reveal information on the hydraulic characteristics of the permeable rock, positioned between input and output location. Depending on the type of test and the scale of investigation, the test data reflect the properties of the conduit system and the fissured system (as the extremes) and a combination of both. If the input location is in the area of a sinkhole (most of the tests), the hydraulic and transport characteristics of the conduits can be examined (neglecting the effect of the transport through the unsaturated zone). With a forced gradient test between two boreholes (probability to encounter a directly connecting fracture is low), the breakthrough curve can be analysed for the parameters of the fissured system. Tracer tests, with the input in boreholes and where the breakthrough is measured at a spring, are influenced by the fissured and the conduit system. The relative proportion of each system is determined by the the distance the dye has to travel from the borehole to reach the draining conduit.

Below, different examples of tracer tests, belonging to the above three categories, are presented.

Conduit System

In region 1 numerous dye-tracer tests were conducted during earlier studies (Villinger, 1969). Due to the intense and deep karstification, a considerable part of the flow is believed to occur in conduits. Figure 8 shows the relationship between the spring discharge and the flow velocities as measured for various artificial dye-tracer tests in the study area. Within the observed range between 20 m/h and 200 m/h, the tracer velocities appear to correlate linearly with the spring discharge. However, the high flow velocities observed for most of the artificial tracer tests are not consistent with the flow regime characterized by the steep slopes of the piezometric surface. As described above, the piezometric levels measured in observation wells are representative for the diffuse-flow as well as for the conduit-flow system, whereas the point-source artificial tracer tests provide information on the tracer flow-path from the injection point to the outlet spring or stream. The scale hierarchy for such an investigation is $I = D = L_n$. There is only one input-output information available for each traced flow-path n. As observed in many

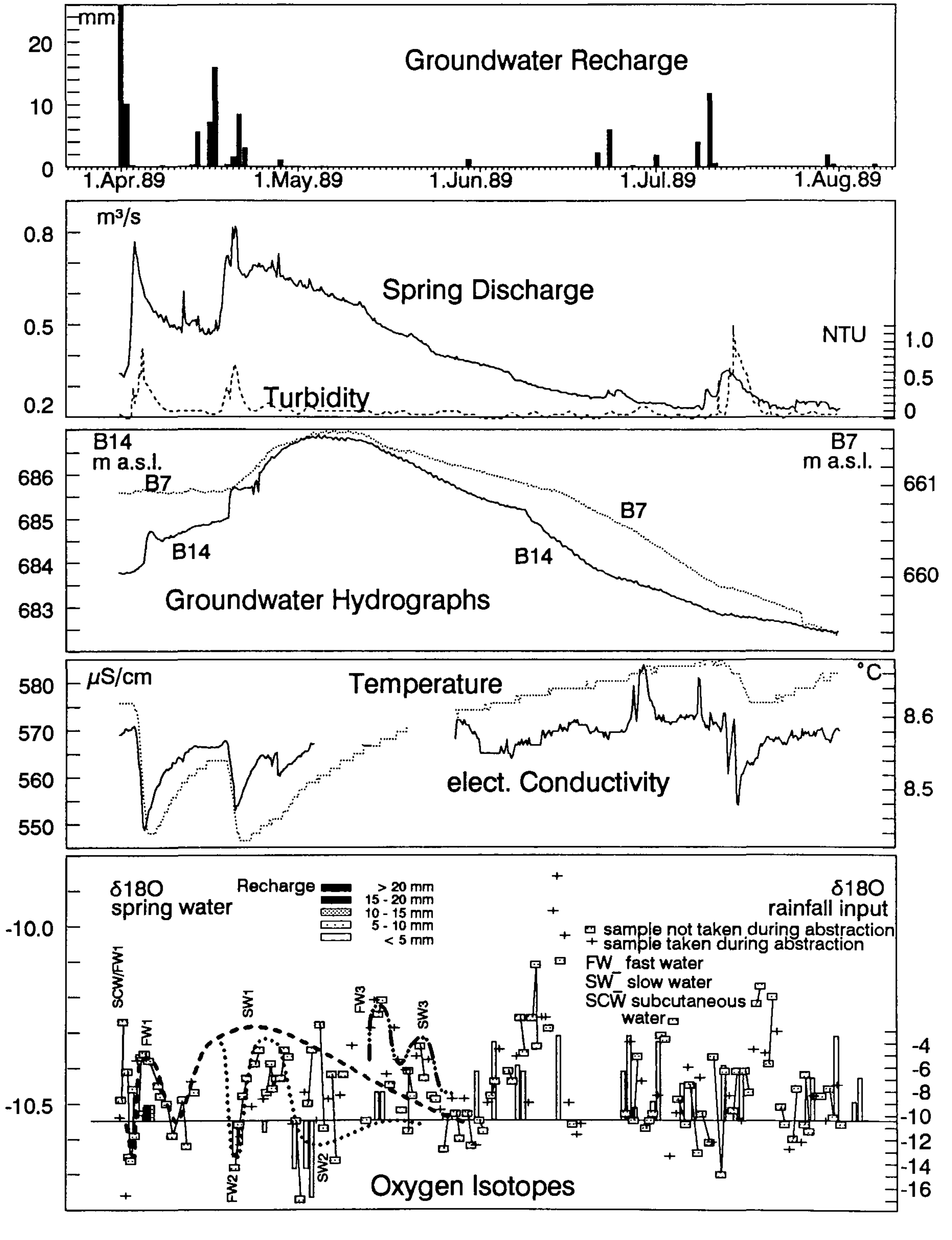

Figure 7: Time series of hydraulic and physico-chemical parameters (Sauter, 1990)

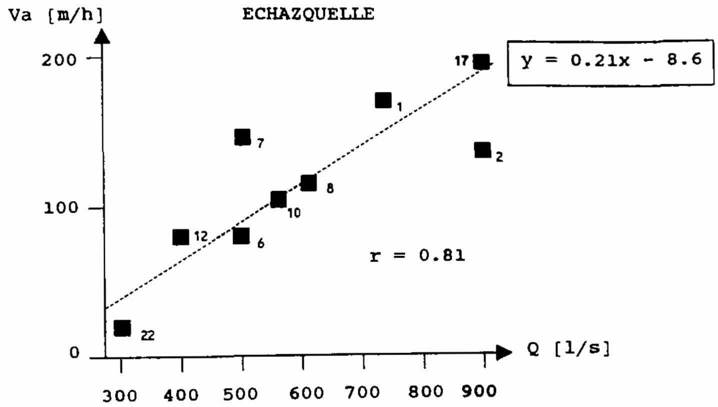

Figure 8: Relationship between spring discharge and tracer velocity for the Echazquelle (Teutsch, 1988)

karst areas, a tracer test from another location within the same catchment may therefore lead to a totally different parameter set.

Diffuse System

Stober (1991) presented a tracer breakthrough of a test, conducted between two geothermal wells, completed in karstified limestones of the Upper Jurassic in the area of Saulgau, situated approximately 50 km south of the Swabian Alb. The wells are 430 m apart. Correcting for interferences from a third well, the breakthrough curve coincides with the theoretical curve for porous media. Because of the low probability of a direct connection between input and output well via a conduit, the test results clearly reflect the characteristics of the fissured system, which obviously can be represented by a simple porous continuum approach. The average pore water velocity was determined at less than 0.2 m/h. The scale hierarchy for this test is $I << D <= L$, where I represents the fissure system only.

Mixed Diffuse-Conduit System

Merkel (1991) performed a number of tracer tests with the input in some of the boreholes of the Gallusquelle catchment (region 4). Samples were taken at the spring, which implies that the breakthrough is a result of the influence of the fissured <u>and</u> the conduit system. The curves are characterised by relatively high peak velocities (35 m/h - 70 m/h) and a long drawn-out tailing. Because the dye was directly injected into the groundwater, the unsaturated zone does have no effect on the breakthrough. The shape of the curve is mainly determined by the properties of the fissured system, whereas the variation in peak velocity depends on the distance between the input location and the point where the dye enters the conduit system.

Some Conclusions

From the field observations described above, it is evident that what is generally described as the characteristics of a karst flow system may depend very much on the type and the scale (D) of the investigation method used and also on the scale (L) of the flow domain. A detection method with a large averaging volume will produce parameter fields which appear to be almost homogeneous, whereas from small scale measurements the same aquifer may appear highly heterogeneous. Similarly, an investigation method might be more selective in the analysis of the high or the low hydraulic conductivity zones.

Consequently, there is not a singular method which should be used to investigate the properties of a karst system. Only the combination of different methods at borehole, intermediate and catchment scale may provide a somewhat complete view of the relevant flow processes.

MODELING APPROACHES

As described above, it may prove useful to analyse and classify the various field observations employing the scale hierarchy concept. However, for any quantitative interpretation the development of an adequate model tool is a prerequisite. There is a whole variety of (physical) model concepts which have been proposed for the simulation of the hydraulic behaviour of fractured rocks. They all employ either a discrete or continous representation of the fracture flow system, together with some algorithm describing the exchange between the fractures and the surrounding rock matrix. The key features of both concepts are briefly discussed below, with respect to their applicability for the simulation of karstified aquifer systems. It can be shown that the selection of the appropriate modeling approach has to be based on the prevailing scale hierarchy of the flow problem.

Figure 9 shows a hypothetical karst aquifer block including fissures and conduits of different size, shape and orientation. This aquifer prototype is translated into five alternative model representations of different type and complexity. For any practical problem, the major model selection criteria are the required investigation effort, the capabilities to simulate karst specific flow characteristics and also the required accuracy of the modeling.

The simplest approach to karst flow is the single-continuum porous equivalent (SCPE) representation shown in Figure 9. It is certainly a crude approximation to the complex variability of the prototype system. Obviously, using this approach individual fractures or conduits cannot be adequately represented. However, in case of a large investigation domain L, individual heterogeneities may not be relevant for the overall system behaviour. To simulate the large scale aquifer behaviour the spatially averaged system properties might be more relevant and have to be investigated at a scale D much larger than the size of the heterogeneities. Therefore, for the resulting scale hierarchy $I << D < L$, the SCPE model could possibly be a valid approximation of the real system. Of course, in a karst terrane these conditions apply only at very large scales and only in cases where the karstification process is not too far advanced. Practical modeling studies where this approach was used for large scale karst studies are described by Baoren and Xuming (1988), Maclay and Land (1988), Teutsch (1988), Tibbals (1990).

A much better representation of the prototype karst system would be achieved if discrete flow-paths could be represented within the model. Two possible representations, the discrete singular fracture set (DSFS) and the discrete multiple fracture set (DMFS) approach are shown in Figure 9. In the DSFS model only one set of fissures and/or conduits is represented, whereas the DMFS model comprises multiple sets of discontinuities representing the whole range of flow systems with small scale fissures and also large scale conduits. The underlying theory of flow through fracture networks has been developed e.g. by Louis (1967). Because of the improved possibilities to represent complex reality, the DSFS and the DMFS models seem preferrable to the continuum models. However, it should be noted that for the discrete fracture approach the location and the geometry of all individual discontinuities must be either exactly known (deterministic model), or has to be described using statistical parameters (stochastic model). For any regional karst aquifer study this leads to an almost unresolvable parameter identification problem, where the scale hierarchy is $I <= D << L$. Bearing in mind that the fracture permeability is proportional to the cube of the fracture width, there is very little chance that any of todays geophysical techniques could provide the required resolution. Therefore, the DSFS and DMFS concepts are applicable only for small scale investigations where adequate fracture statistics might be obtained from densely spaced measurement points. To our knowledge the DSFS or DMFS concepts have not been used so far for any real-world regional study in a karst terrane.

Where appropriate, large scale information is available on the location and the extent of the major conduits, a combination of the SCPE and the DSFS approach might be most adequate. As shown in Figure 9 the diffuse-flow system would be represented by a porous equivalent and the conduit system through discrete fracture elements. The scale hierachy for such a system is $I << D < L$ for the SCPE representation and $I >= D < L$ for the DSFS representation. For such a problem, the parameter identification problem is mostly reduced to the detection of the geometry of the large conduits system. Kiraly (1982) analysed some hypothetical configurations to

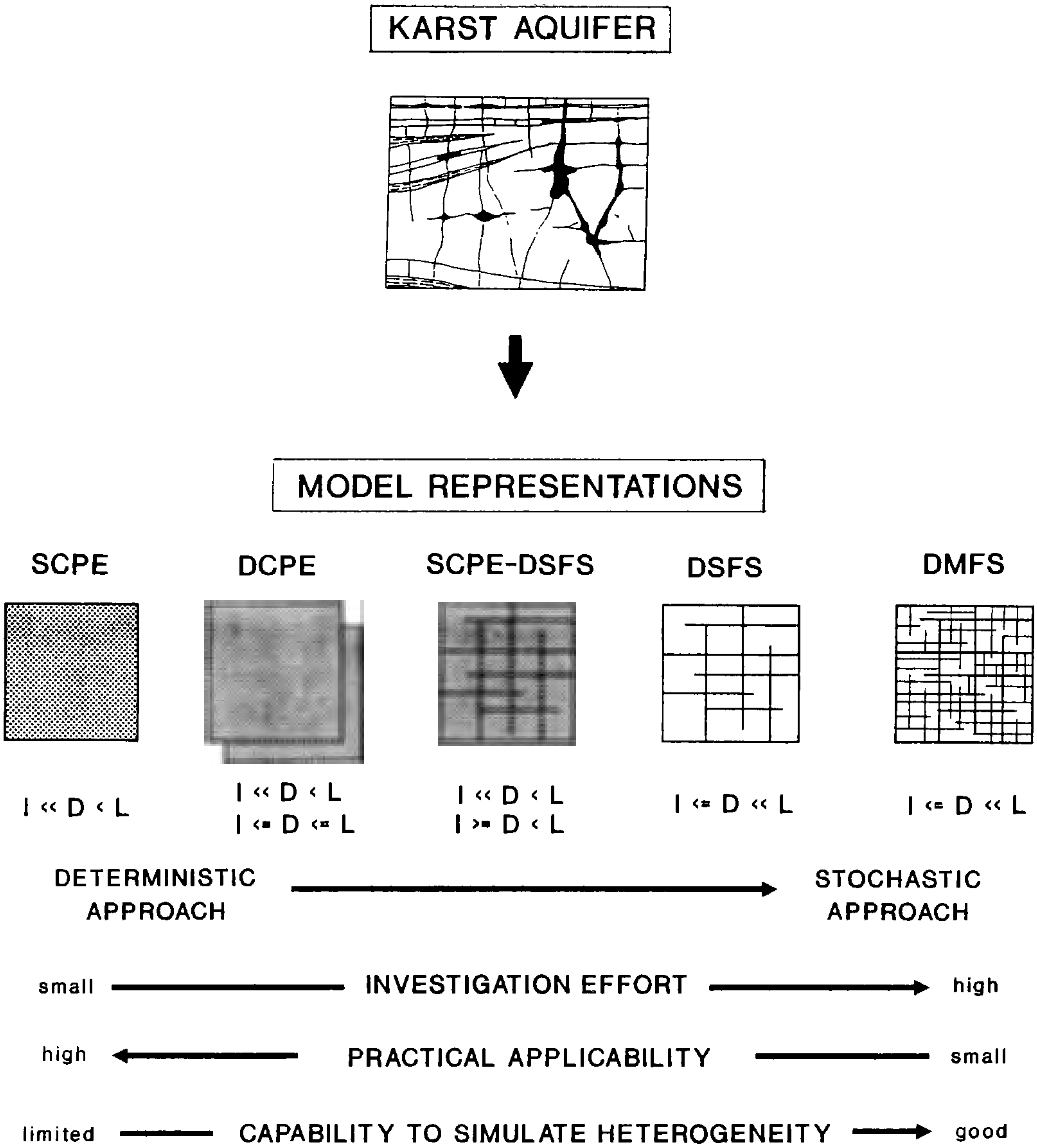

Figure 9: Karst aquifer prototype and five possible model representations (SCPE: single-continuum porous equivalent, DCPE: double-continuum porous equivalent, DSFS: discrete singular fracture set, DMFS: discrete multiple fracture set)

demonstrate the resulting flow effects. Practical field studies were presented by Kiraly (1984) and Yusun (1988).

A compromise between the simple single-continuum and the complex discrete fracture representation is the double-continuum porous equivalent (DCPE) approach shown in Figure 9. Based on the available field evidence from karst terranes, showing high fluctuations of the piezometric levels together with very high groundwater flow velocities, physical reality is approximated using the concept of two overlapping continua. One continuum is represented by the more karstified areas (conduit-flow), yielding high groundwater flow velocities but hardly any water level fluctuations. The second continuum is represented by the moderately karstified aquifer zones (diffuse-flow) with lower hydraulic conductivity and higher storativity. The advantage in using the DCPE model instead of the SCPE is the improved representation of the karst aquifer heterogeneity. On the other hand, the DCPE model does not require the detailed geometrical information which is needed for the discrete fracture approaches. Typically, the scale hierarchy where the DCPE model is applicable would be $I << D < L$ for the diffuse-flow continuum and $I <= D <= L$ for the conduit-flow continuum. The DCPE flow and transport model approach was successfully used in the analysis of several karst systems of the Swabian Alb aquifer (Teutsch 1988, 1989; Sauter 1990, 1991). To our knowledge, the only other attempt to use a DCPE approach in a Karst terrane is reported by Yilin et al. (1988) who simulated the flow in the karst regions of northern China.

Based on the above said, the authors believe that the double-continuum approach is a favourable model concept for many practical karst aquifer studies where data is usually scarce but detailed enough to show that a single-continuum model cannot be applied. One further advantage of the DCPE approach is its simple mathematical formulation described below, which facilitates the implementation into existing program codes.

THE DOUBLE CONTINUUM POROUS EQUIVALENT (DCPE) MODEL

Mathematically, the double-continuum concept is similar to the well-known double-porosity approach which is extensively used in the oil industry (Barenblatt et al. (1960), Warren and Root, 1963; De Swaan 1976; Streltsova 1988). However, flow and transport may occur in both systems with an exchange coefficient desribing the cross-flow between the two. The DCPE model may be visualized as two porous blocks with a common outflow, which are connected through hydraulic resistances at any of the block cells. Such a system is shown in Figure 10.

The equations describing the flow and transport of a non-reactive solute in a double-continuum system are (Teutsch, 1988):

Flow (one-dimensional, depth-averaged):

$$\frac{\partial}{\partial x}\left(K_x^m M \frac{\partial h^m}{\partial x}\right) = S^m \frac{\partial h^m}{\partial t} + (-1)^{m+1} \alpha (h^m - h^{m+1}) \qquad m = 1,2, \qquad (1)$$

Transport:

$$\frac{\partial}{\partial x}\left(D^m \frac{\partial C^m}{\partial x}\right) - v^m \frac{\partial C^m}{\partial x} + (-1)^{m+1} \alpha^* (C^m - C^{m+1}) \qquad \frac{\partial C^m}{\partial t} \qquad m = 1,2, \qquad (2)$$

and

$$\alpha^* = \alpha \frac{(h^1 - h^2)}{n_e M} \qquad (3)$$

where

m = continuum index	h = head potential
K_x = hydraulic conductivity	M = aquifer thickness
S = storage coefficient	D = hydrodynamic dispersion coefficient
C = concentration	v = groundwater flow velocity
n_e = effective porosity	α = exchange coefficient

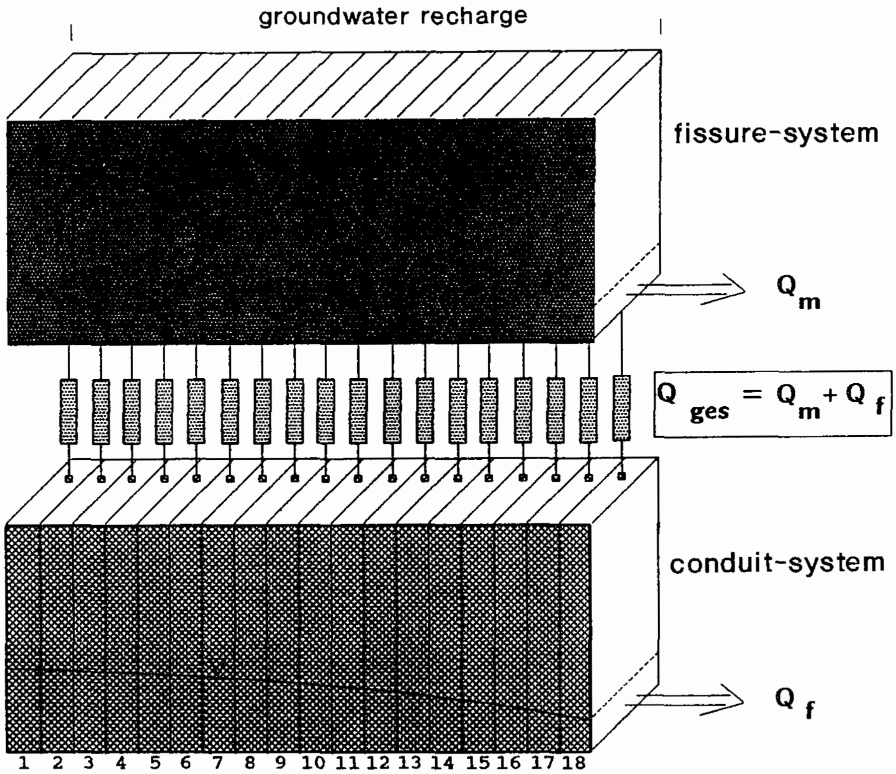

Figure 10: Double-continuum porous equivalent (DCPE) model (Teutsch, 1988)

Two assumptions have to be made in order to apply this concept to a karstified aquifer. Firstly, there is the assumption of a potential continuum in the highly permeable conduit-flow system, at an intermediate scale smaller than the modelled domain. This is probably true for most of the study regions presented, but not necessarily true for those aquifer zones comprising extremly large active conduits. Secondly, it is assumed that the flow in both continua is laminar. This is probably not always true for the fast conduit-system, especially not following strong recharge events. However, due to the lack of information about the discrete geometry of individual conduits, no correction for the nonlinearity of the flow can be introduced.

Based on the well known MODFLOW groundwater flow program (McDonald and Harbaugh, 1984) a three-dimensional double-continuum model was recently developed at the University of Stuttgart. This program is presently used for the analysis of the regional karst flow in region 5, where a railway tunnel with a length of 13 km is planned to cut through the Swabian Alb escarpment.

CASE STUDY

The applicability of the DCPE model has been tested on the Gallusquelle groundwater catchment (region 4), an area for which longterm (25 years) of flow and hydrograph data were available, as well as numerous tracer tests, small scale and large scale hydraulic tests and two years of water quality data. These were good conditions to calibrate and test the model approach on actual field measurements.

Figure 11 and 12 demonstrate that the flow and the hydraulic characteristics could be reproduced very well, considering the assumptions in recharge input and its respective distribution with time and space (Sauter, in prep.). Observed differences can be explained by the absolute depth of recharge and by the fact that the hydraulic parameters vary with depth. In Sauter (1991 - this issue) it is shown how the aquifer geometry, the hydraulic conductivity and the storage coefficients have been determined for each part of the system and at the respective scale, consistent with that used for model calibration.

Without any further modifications, the regional transport of an inert areal tracer, such as $\delta^{18}O$, was simulated with a double-continuum random-walk model (Teutsch, 1988) (Fig. 13). It is interesting to note, that as opposed to the unimodal breakthrough of point source tracer tests, areal source tracers may produce a bimodal breakthrough, which, as shown in Figure 7, is modified due to the interference of several other recharge events (dilution). The result is a highly complex signal, which compares well with the isotope breakthrough observed in the field, which would be very hard to understand without any model assistance.

Further work will focus on the analysis of breakthrough signals resulting from point and areal sources modified by the various physical and chemical processes. The model is a tool which can be employed to analyse specific scenarios concerning resource development as well as contaminant migration. Furthermore, it allows the testing of various hypothesis such as autogenic and allogenic recharge mechanisms and also the simulation of chemical changes due to e.g. dissolution.

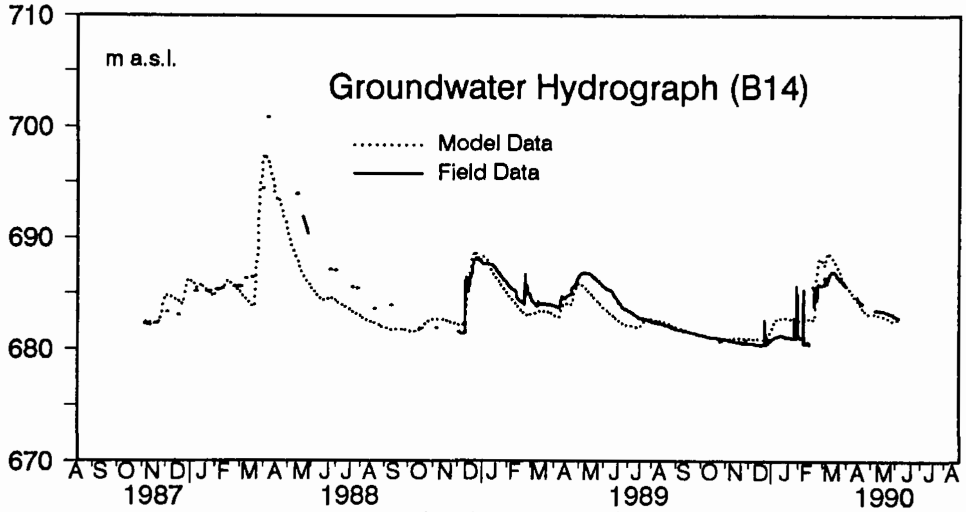

Figure 11: Measured and simulated groundwater hydrograph

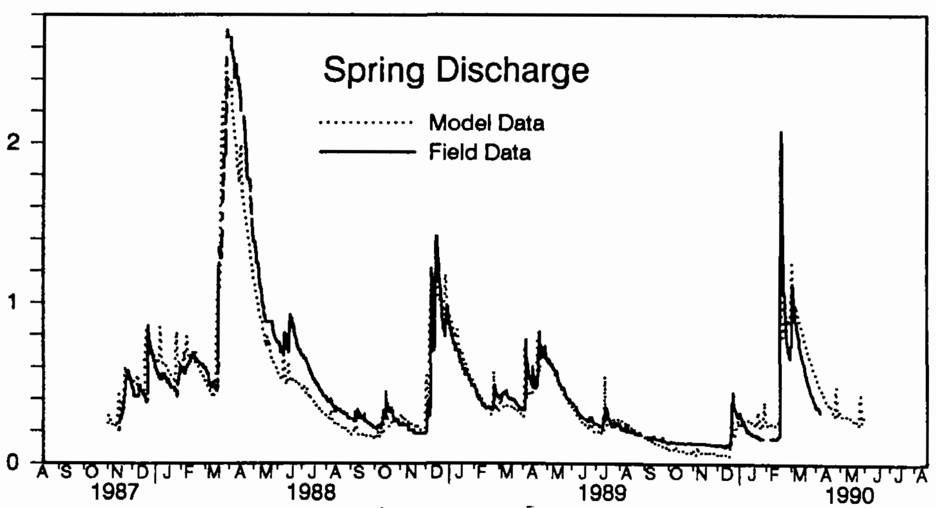

Figure 12: Measured and simulated spring flow hydrograph

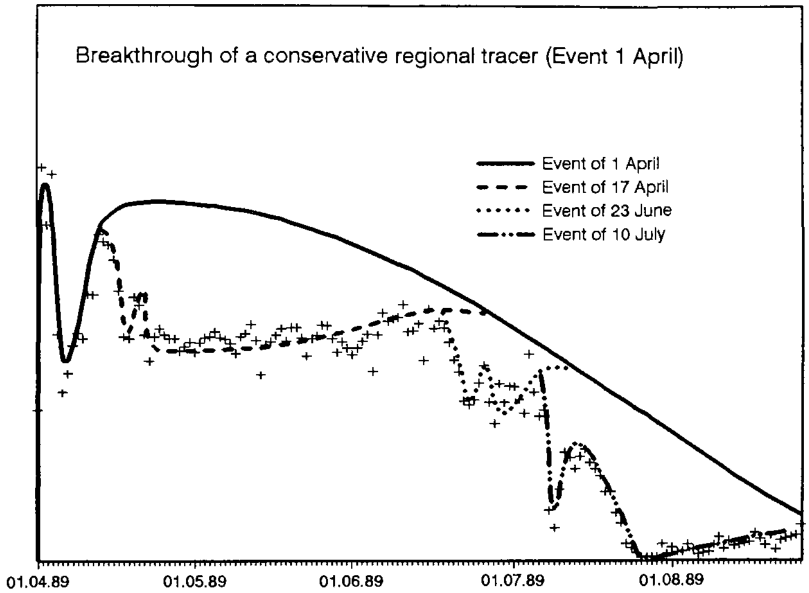

Figure 13: Simulated breakthrough of an areal-source tracer

SUMMARY AND CONCLUSIONS

The aim of this paper is to present a consistent framework for the classification of flow and transport phenomena frequently observed in karst terranes, which in many cases appeared to be incompatible in a single flow system. This approach is based on scale hierarchy concepts which are illustrated on various field observations. It is shown that the classification of the flow system in a heterogeneous aquifer like the karst may depend strongly on the type and the scale of the investigation method used. Furthermore, the relationship between the model conceptualisation and the scale hierarchy is discussed. Various experiences from the Swabian Alb karst aquifer system show that the double-continuum porous equivalent (DCPE) model approach is superior to other concepts for most karst aquifer situations, which require an adequate representation of the heterogeneities on a very limited data base.

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BIOGRAPHICAL SKETCHES

Georg Teutsch: He received his M.Sc. in Hydrogeology in 1980 from the University of Birmingham (UK) and his Ph.D. in 1988 from the University of Tübingen (FRG). He has been recently appointed as an associate professor of geohydrogeology at the Institut für Wasserbau of the University of Stuttgart where he conducts a major part of the groundwater research since 1986. After various assignments in the field of groundwater resources evaluation and exploration, his current interests focus on transport processes in heterogeneous media. He has authored more than 30 scientific publications in the field of practical and theoretical geohydrology with an emphasis on groundwater modeling.

Martin Sauter: He is a research associate at the Chair of Applied Geology at the University of Tübingen (FRG). In 1980 he obtained a diploma in Geology from Tübingen University and in 1981 an M.Sc. in Hydrogeology from Birmingham University (UK). Between 1982 and 1987 he was involved in various projects in Germany and overseas, covering areas of groundwater contamination, geophysical well logging and water resources development. He is presently finalizing his Ph.D. project on regional flow and transport modeling and model validation in karst terranes. His main interests lie in the area of groundwater flow and transport modeling in karst, hydraulic testing methods and hydrochemistry.

Questions-and-Answers:

Q1: You identified your model used as a modification of MODFLOW. Is this modified MODFLOW available in either draft form or final form ? If so, from whom ?

A1: The double-continuum model presented in the paper is based on the well-known USGS MODFLOW package. However, the structure of the program is different because of two instead of one continuum and the cross-flow calculation. Furthermore, a routine has been developed which allows the reactivation of cells which fall dry during the simulation. A documented and tested version of the program will be available from the Univ. of Stuttgart, probably by mid of 1992.

Q2: You briefly mentioned that the northern aquifer system is dominated by diffuse flow [Remark: here the correct question should probably read "conduit flow" instead of "diffuse flow"] and the southern aquifer is dominated by diffuse flow. What are the geologic controls on [Remark: here the correct question should probably read "and" instead of "on"] which aquifer type is dominant ? How important are facies changes within the carbonate rocks.

A2: In the northern part of the Swabian Alb karst aquifer the piezometric levels are controlled by a series of springs which developed at the aquifer base during the Quaternary as a result of tectonic uplift and tilting of the limestone plateau. Due to the spatial concentration of flow and the increased hydraulic gradient, the original diffuse flow conditions therefore matured to a conduit flow system. In the southern part of the aquifer, the tilting of the limestone plateau caused a decrease in the hydraulic gradient and therefore the diffuse flow conditions were maintained. Within this regional flow pattern, the local flow conditions are of course highly dependent on the type of carbonate facies and therefore highly variable.

Q3: The fit between the DCPE model and actual isotopic values is significantly worse than would be the case if a simple 2-point moving average had been used. What then is the utility of the DCPE model ?

A3: It should be noted that the isotope data was not used for the calibration of the DCPE model. The model was calibrated on the base of spring hydrographs, groundwater levels and a few point-input, artifical dye-tracer test data only, whereas the isotope data represent an areal-input, natural tracer transport. In fact, a recently conducted analysis of the isotope breakthrough curves has demonstrated the high sensitivity of the model outcome on the double-continuum cross-flow coefficent. Referring to the proposed 2-point moving average approach, I don't see how this could help improve the model fit in a physically meaningful way.

Session I:

Ground Water Modeling and Hydrogeology

ASSESSMENT OF HYDRAULIC CONDUCTIVITY IN A KARST AQUIFER AT LOCAL AND REGIONAL SCALE

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ABSTRACT

The hydraulic parameters in a karstified limestone aquifer are usually only available for single boreholes. There is a high density of information on wells in valleys, which are not necessarily representative for the aquifer due to an increased degree in karstification close to the regional outlet. As a result of these circumstances, the values obtained cannot be directly used as input parameters for a regional model. The selection of the appropriate hydraulic parameters requires a careful analysis of the scale, the values are applicable at, and the allocation of the parameters to the respective part of the system, which consists of a fast transit (conduits) and a slow flow (fissured) system. Slug tests, injection, packer and pumping tests were used to obtain parameter estimates of the low and intermediate range of the hydraulic conductivity spectrum, which reflect the response of the aquifer at local scale. Regional parameters could be evaluated using several approaches. Applying Darcy and with the information on discharge and head gradient, average transmissivities can be calculated, representing the slow regional system. Taking the concept of Rorabaugh (1964) for flow from bank storage, regional parameters were computed, using the recession coefficients for the fast and slow flow system. The necessary storage coefficients for the fast system could be derived from tracer tests and for the slow system from the quotient between the discharged volume of water and the drained rock volume. The regional values were checked for plausibility, using the local values and taking into account the different proportions of the fast and slow system. The resulting calibrated model parameters compared reasonably well with the observed regional values.

INTRODUCTION

During the last few years, double-continuum models have become a recognized tool for the prediction of flow and transport in karstified limestone aquifers (Teutsch, 1988, Yilin et al, 1988, Sauter, 1991, Sauter, 1992). This type of modeling approach manages to accomodate both, a regional flow system, that is slowly depleted and at the same time a fast transit system to simulate the rapid response after a rainfall event. It also simulates tracer breakthrough with the observed high average velocities. Double-continuum models are also based on the physics of groundwater flow and capable of simulating spatial variations in flow and transport phenomena. These advantages however require on the other hand the knowledge of flow and transport parameters of the fast and the slow system at the appropriate spatial resolution scale, which are frequently difficult to obtain. Depending on the scale of the problem, different hydraulic conductivities will apply as illustrated by Kiraly (1975) (figure 1). The hydraulic parameters of slow and fast system will be different whether for example a borehole to borehole tracer test (several tens of meters) or regional groundwater flow (several thousands of meters) is being analysed (Teutsch & Sauter, 1991, this issue).

In the present paper, the terms conduits and fractures are used interchangeably for the fast transit system and fissures and matrix are used to describe the slowly draining part of the aquifer, in order to show the connection to double-porosity terminology and its applicability to karst aquifers.

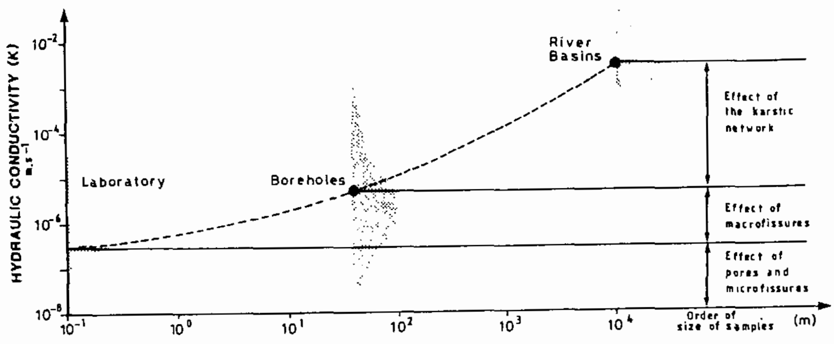

Figure 1: Effect of scale on the hydraulic conductivity of karst aquifers (Kiraly, 1975)

This paper attempts to demonstrate how the hydraulic parameters can be obtained from various tests applying different analytical methods and how the appropriate values can be found for the respective scale at which information is sought. The idea is to evaluate the hydraulic conductivity at the lower scales in order to be able to derive the resulting hydraulic conductivity at the next larger scale, together with a knowledge of the various proportions of fast and slow flow system, which exist at every scale. The parameters obtained can be compared with the calibrated values of a regional double-continuum groundwater flow model.

The results presented below are part of a karst research project, investigating flow and transport processes in a spring catchment of a karstified limestone aquifer in south Germany. The objectives are the numerical simulation of these processes and the quantification of the input parameters and groundwater recharge.

SCALE PHENOMENA IN KARSTIFIED LIMESTONE AQUIFERS

In highly heterogeneous systems such as karst aquifers, parameters like porosity and the related storage coefficient are highly dependant upon the scale of the investigation method (Teutsch & Sauter, 1991, this issue). Castany (1984) demonstrates how the porosity values obtained vary with the volume of rock sampled, expressed as the radius of investigation. Starting at a micro (laboratory) scale, where the porosity can vary from 0 (tight rock) to 1 (void), homogeneous values can be measured if a sufficiently large number of heterogeneities (fissures, conduits) are integrated within the sample volume. Contrary to the porosity, the hydraulic conductivity increases at a larger scale (Kiraly, 1975, figure 1, Department of Energy, 1986), which is plausible because of the added effect of the high permeability conduits. The porosity can also be assumed to increase but only very slightly, which however cannot be measured. This different behaviour could be explained by the cubic law (Snow, 1965), which describes the relationship between permeability and the fracture aperture.

RELEVANCE OF SCALE EFFECTS IN GROUNDWATER MODELLING

In a numerical model, simulating groundwater flow and transport, the hydraulic parameters dedicated to the respective discretised unit (element/cell), is a result of the sum (arithmetic/harmonic/geometric) of the parameters of the identified sub-systems, weighted by their respective proportions within the particular model unit. The problem of accomodating a number of flow sub-systems in a numerical model can be approached by either employing models, incorporating multiple continua such as MINC (Narasimhan et al, 1988) or by determining the representative parameter at the modelled scale of a double-continuum model. The structure of the MINC model, applied to a karst aquifer, could be visualised as an intertwined system of fast (fracture, conduit) and slow (matrix, fissure) flow, i.e. double-continuum systems, whereby each slow system can again be split into a conduit and fissure flow continuum.

The fractured continuum of the lowest manageable double-continuum sub-system could be visualised as the joints, spaced at decimeter intervals (10 cm - 30 cm) and frequently measured in quarries. The complimentary parameters of the matrix (fissure) continuum can be measured in the laboratory on cores with dimensions of several centimeters. The above described double-continuum system could be regarded as the lowest in the hierarchy of multiple interacting continua. It forms the matrix continuum of the next higher double-continuum system, where the higher permeable set of fractures and the solution widened fissures are spaced apart between one and several meters. The double-continuum system can be investigated at this level in the hierarchy by double packer tests, where the packers enclose a test interval of less than one meter.

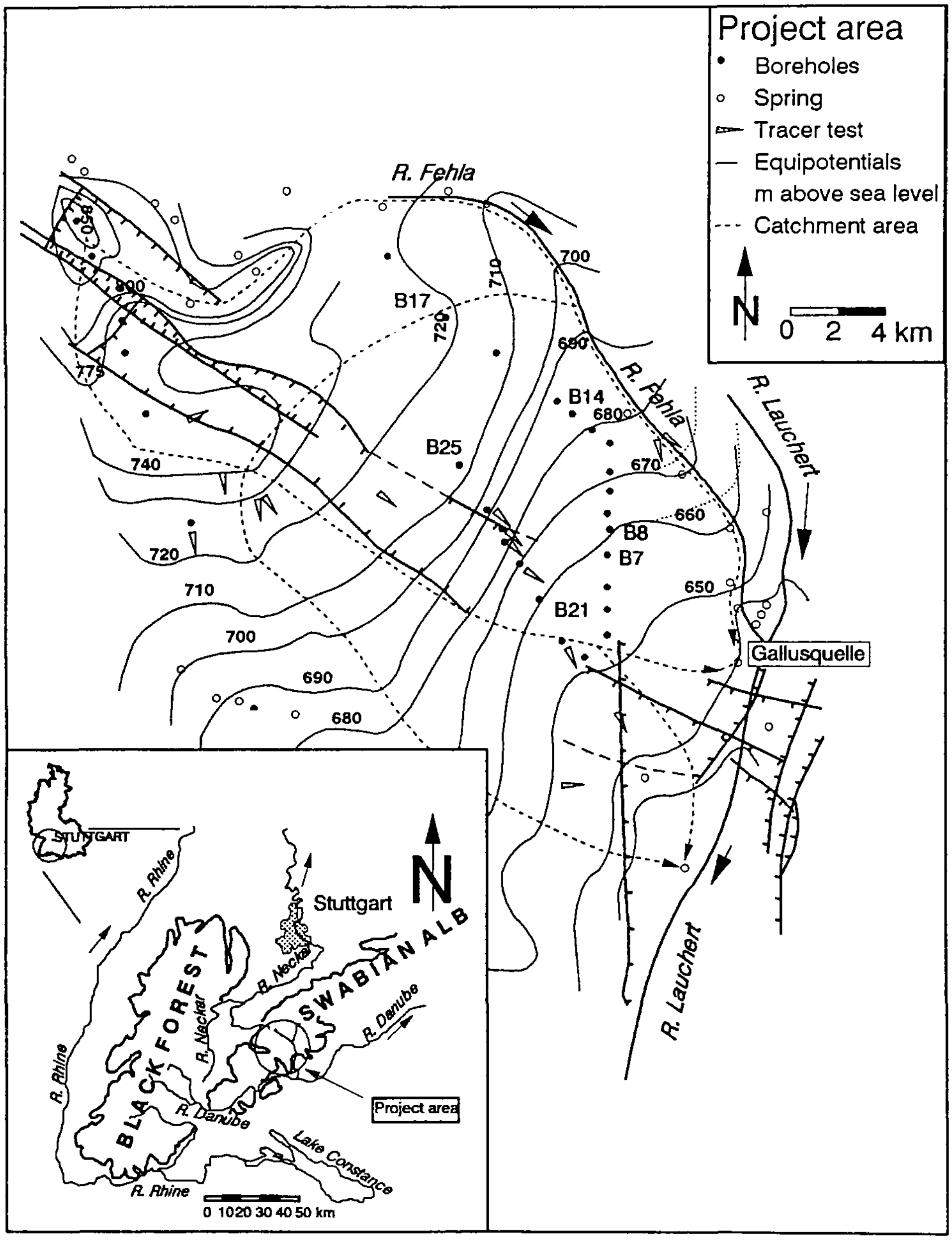

Figure 2: Gallusquelle karst aquifer catchment

Slug and injection tests with a test interval and radius of investigation of several tens and pumping tests with a scale of investigation of several hundreds of meters can identify the double-continuum systems at the respective scales.

At the regional, i.e. catchment scale, where conduits and caves represent the fracture continuum, the hydraulics of the matrix continuum could possibly be derived from pumping tests, which are, as outlined above, the result of the hydraulic characteristics and the interaction of all the lower scale double-continuum flow systems.

For practical purposes, if the problem in question asks for answers at regional scale, no information is required on e.g. the potential change over a distance of several meters. It is also difficult to obtain all the relevant input data at every scale. Therefore, the flow system can be conveniently simplified to a single double-continuum model, assuming that the parameters can be determined at the modeling scale.

Within the context of this study, the model was designed to simulate regional flow and transport and the breakthrough of tracer tests with travel distances of several thousands of meters. Therefore, the nodes of the model grid were spaced 500 m apart, which was a compromise between the lowest resolution of information on aquifer geometry, and the radius of investigation of a large scale pumping test.

PROJECT AREA

For the investigations, the spring catchment of the Gallusquelle was selected. It is situated in south-west Germany on the Swabian Alb, a small mountain range, that stretches in an approximate south-west north-east direction for roughly 200 km (figure 2). Morphologically, the project area dips gently from an escarpment (1000 m a.s.l.) in the north-west down to about 600 m in the region of the spring. The Gallusquelle groundwater basin forms a part of the catchment (450 km^2) of the Lauchert river, which is fed mainly by karst springs. The river Fehla represents the boundary to the north-east.

The area is well suited for the intended measuring program. More than 25 years of continuous records of discharge, waterlevel fluctuations (weekly readings), and climatic data were available, as well as a number of about 20 wells that allow the construction of a water level map, even in areas far removed from the discharge point. Fifteen tracer tests help to delineate fairly accurately the catchment boundaries, which cover an area of approximately 45 km^2.

GEOLOGY

Geologically, the area is composed exclusively of carbonate rocks of the Upper Jurassic (figure 3). At the surface, predominantly the massive limestones of the Kimmeridge 2/3 are exposed, which reach a maximum thickness of between 90 and 140 m. They are underlain by the Kimmeridge 1, a marly limestone sequence, with more or less expressed bedding and a thickness of approximately 50 m. The lowest relevant geological unit consists of well bedded limestones, the Oxford 2. The whole stratigraphical succession dips south-east. The south-west border of the catchment is formed by the Hohenzollern graben, which is tectonically still active. Another fault zone strikes north-south and borders the project area in the south-east.

HYDROGEOLOGY

As shown in figure 3, the aquifer is formed by three geological units, the massive limestones (ki2/3) in the south-east, the marly limestones (ki1) in the centre and the well bedded limestones (ox2) in the north-west. The aquifer base does not follow any stratigraphical boundary. Figure 3 shows highly transmissive and permeable zones of the Gallusquelle groundwater catchment. The boreholes were all projected onto a profile line, assuming no variations vertical to the profile, and the borehole logs were interpreted in terms of tight rock, fractured and slightly karstified and highly karstified. Further evidence, regarding the differentiation in hydraulic characteristics could be provided by the analysis of long and short-term records of groundwater hydrographs (Sauter, in prep.).

Saturated thicknesses are estimated to reach approximately 30 m, in exceptional cases 50 m, depending on the season, although the limestone sequence may be in excess of 150 m. An increased gradient of the piezometric surface can be observed in the centre of the catchment (figure 4), where the water table cuts across the less permeable marly limestones. Further upgradient, the aquifer is formed by the Oxford limestones. The water table constitutes the top of the aquifer and unconfined conditions prevail in the entire catchment. The unsaturated zone is highly karstified and reaches thicknesses of between 90 and 120 m.

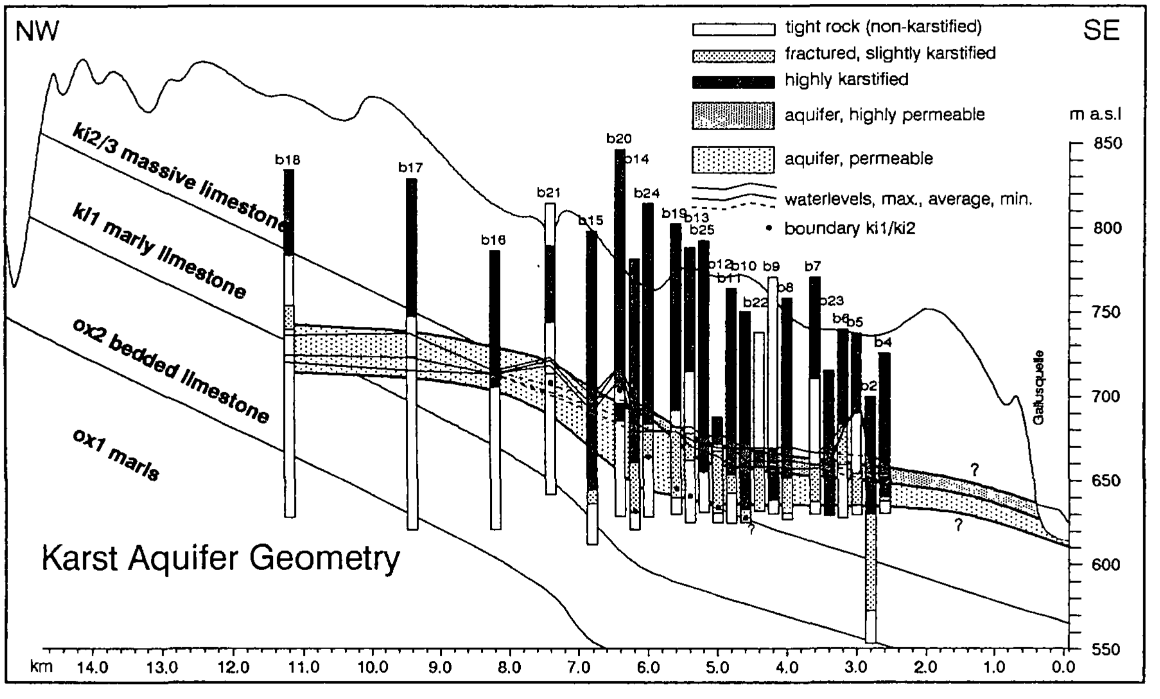

Figure 3: Geological and hydrogeological crossection of karst aquifer

Closer to the spring, annual water level fluctuations range between 5 and 15 m and further upgradient, between 10 m and 30 m, reflecting the decrease in transmissivity and storage coefficient away from the point of discharge.

DETERMINATION OF HYDRAULIC PARAMETERS AT THE RESPECTIVE SCALE

The evaluation of hydraulic parameters was subdivided into a regional assessment, taking into account the existence of the fast (conduit) and the slow flow (fissure) system, local and sub-local scale tests in boreholes and laboratory measurements derived from the literature. Particular emphasis has been put on the separate evaluation of fast and slow system characteristics.

REGIONAL PARAMETERS

Common hydraulic testing methods such as slug tests, packer tests and pumping tests, have a limited testing radius, which is usually in the order of several tens to hundreds of meters, depending on the type of aquifer material. The only signal, exciting larger regions of the aquifer system are natural pulses, such as rainfall events, which can be evaluated by applying recession analysis to the output signal, i.e. the spring discharge. This method has the advantage that information can be gained over a large area, but because the registration of the output signal can only be measured at the spring, a differentiation into regionally varying parameters is generally not possible. If however the regional hydraulic conductivity varies laterally and with depth (figure 3), the value of the parameter obtained depends on the intensity of the signal (recharge depth) and only reflects the hydraulic characteristics of that portion of the aquifer, dominating the flow.

The approach taken for the evaluation of the regional hydraulic conductivity was first to measure average regional gradients in the south-eastern, central and north-western part of the Gallusquelle catchment during three different flow conditions, low (0.12 - 0.36 m^3/s), intermediate (0.47 - 0.99 m^3/s and high flow (1.25 - 2.66 m^3/s). Assuming that there is no more recharge to the aquifer together with a knowledge of the respective discharge and the saturated thickness, values can be calculated for T and K.

Secondly, the recession coefficients α, derived from spring discharge measurements were used to calculate average transmissivities at different fill levels of the aquifer, using a relationship, developed by Rorabaugh (1964) for groundwater discharge from bank storage.

Hydraulic Conductivity

Gradient (Darcy) approach

Figure 4 shows a block diagram of the aquifer, subdivided into three different areas, a highly conductive one in the south-east, low conductive in the central and intermediately conductive in the north-western part. The respective discharge is allocated as a fraction of the total, according to the three different surface areas. The base of the aquifer was taken from figure 3 and the saturated thickness varied according to the waterlevel in the boreholes, taken for evaluation of the gradient.

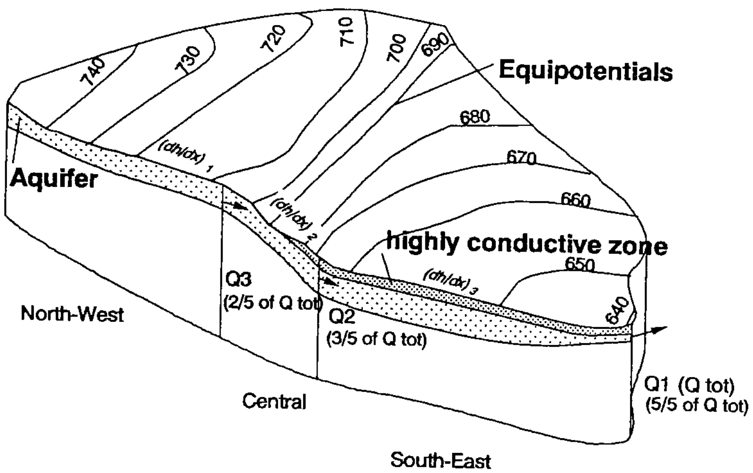

Figure 4: Evaluation of regional hydraulic conductivity using throughput analysis

The slope of the watertable was determined using the waterlevels in boreholes B7 and the spring for the south-east and that in B25 and B21 for the central region. Because of the lack of borehole information, an average regional gradient was taken from Villinger (1977) and figure 2 for the north-west.

The hydraulic conductivities varied between 2 and $13*10^{-4}$ m/s for the south-east and $2*10^{-5}$ and $3*10^{-4}$ m/s for the central region. An average hydraulic conductivity of approximately $1.5*10^{-4}$ m/s was determined for the north-western region. The lower value was generally calculated during low flow conditions, which is believed to represent "regional matrix", i.e. regional fissure hydraulic conductivity. The more conduits are included at higher saturated thicknesses (high hydraulic conductivity zone), the higher the resulting K. During intermediate flow conditions, the higher permeable zone, identified in figure 3 is partly saturated, derived from the water level in B7 (above 660 m), and is therefore contributing to the flow. Above a level of 663 m, permeability is reduced again. As could already be expected from the information on waterlevels and aquifer geometry, the hydraulic conductivity is lowest within the high gradient area and highest within the south-eastern area close to the spring.

		Gradient:	low			Gradient:	high		
		low	interm.	high		low	interm.	high	
	Va		110	250	m/h		110	250	m/h
	flow:	100	600	1500	l/s	100	600	1500	l/s
South-East	K regcond:		7.6	17.4	m/s		5.1	9.9	m/s
	%frac		0.00007	0.00006			0.00010	0.00011	
	K reg	0.0002	0.0007	0.0013	m/s	0.0002	0.0007	0.0013	m/s
	Gradient		0.004	0.004			0.006	0.007	
Central	K regcond:		3.1	6.9	m/s		1.3	3.0	m/s
	%frac		0.00003	0.00004			0.00006	0.00009	
	K reg	0.00002	0.0001	0.0003	m/s	0.00002	0.0001	0.0003	m/s
	Gradient		0.01	0.01			0.023	0.023	
North-west	K regcond:				m/s				m/s
	%frac				%				%
	K reg		0.00015		m/s		0.00015		m/s
	Gradient								

Table 1: Regional hydraulic parameters (Darcy approach)

The main source of error lies in the estimation of the saturated thickness. It was however not possible to use transmissivities because of prevailing unconfined conditions and because the waterlevel fluctuations amount to a considerable fraction of the total saturated thickness at average flow conditions.

The hydraulic conductivities obtained are a result of the varying contributions of fracture (conduit) and regional matrix (fissure) system. At low flow conditions it can however be assumed that the effect of the regional conduit system is minimal, because all the flow stems from the matrix blocks and that the measured gradients are mainly determined by the regional matrix. At higher flow and flood conditions the hydraulic conductivities are dominated by the conduits.

Evaluation of Fracture Contribution to the Hydraulic Conductivity

With a knowledge of the hydraulic conductivity of the regional fissured continuum (K_{regmat}), together with an estimation of the hydraulic conductivity of the total system (K_{reg}), a volume percentage of the conduits per unit volume permeable rock (%frac) can be determined (eq. 1, 2).

$$K_{reg} = K_{regfis} (1 - \%frac) + K_{regcond} \cdot \%frac \qquad (1)$$

with

$$K_{regcond} = \frac{v_a}{n_e} \cdot \left(\frac{\partial h}{\partial x}\right)^{-1} \qquad (2)$$

This approach however requires information on the hydraulic conductivity of the conduit system, which can be derived from the average velocity v_a of tracer breakthrough curves. The arithmetic average of the relative contributions of the two systems was used for the calculations (eq. 2). Because the effective porosity (n_e) is unity in fractures, the only unknown in the calculation of $K_{regcond}$ is the hydraulic gradient in the fractures. It was attempted to to estimate the range of gradients, likely to occur, using various approaches. The maximum gradients within the fractures are those of the regional matrix and the minimum gradient is parallel to the aquifer base, derived from figure 3. Lower gradients are unlikely to occur. With the approach described above, $K_{regcond}$ can be narrowed down to one order of magnitude. Table 1 lists the results of the calculation and it becomes apparant that the hydraulic conductivity of the conduits is likely to range between a few and approximately 10 m/s, depending on the gradient and the flow conditions. Correspondingly, the volume percentage of the fracture contribution to the flow can be estimated to range between 0.005% and 0.01%, which compares well with the conduit storage evaluated below.

To further support the $K_{regcond}$ values obtained, they were compared with those of Gale (1984). The karstification in the project area however is much less developed, than in the catchments discussed in Gale (1984). The author analysed the morphology of the conduit-bottom beds (e.g. scallops), composed of hydraulically transported sediments and the solutionally developed bedforms of accessible conduit walls. The author was able to derive mean values of flow velocity and other hydraulic properties for turbulent flow. Mean conduit flow velocities, which were partly derived from literature varied between 0.43 m/s and 0.03 m/s. The flow velocities derived from tracer tests and used for the evaluation of hydraulic conductivity (table 1) range at the lower end of this spectrum, i.e. between 0.07 m/s and 0.03 m/s. These flow velocities are however an underestimate of the actual mean velocity in the channels and conduits, because of the tortuous nature of the flow paths. There is also a certain bias towards higher flow velocities, because the conduits have to have a certain size, in order to be accessible.

Baseflow Recession Method (Rorabaugh, 1964)

Rorabaugh (1964) showed that the slope of base flow recession curves of water released from bank storage after flood events plotted on a log-lin diagram (discharge - log) is proportional to aquifer diffusivity. With a knowledge of the groundwater basin geometries (L average distance to the groundwater devide) and the storage coefficient, transmissivities and hydraulic conductivities can be evaluated according to the following eq. 3

$$T = \frac{\alpha \; 4 \; S \; L^2}{\pi^2} \tag{3}$$

with α representing the base flow recession coefficient.

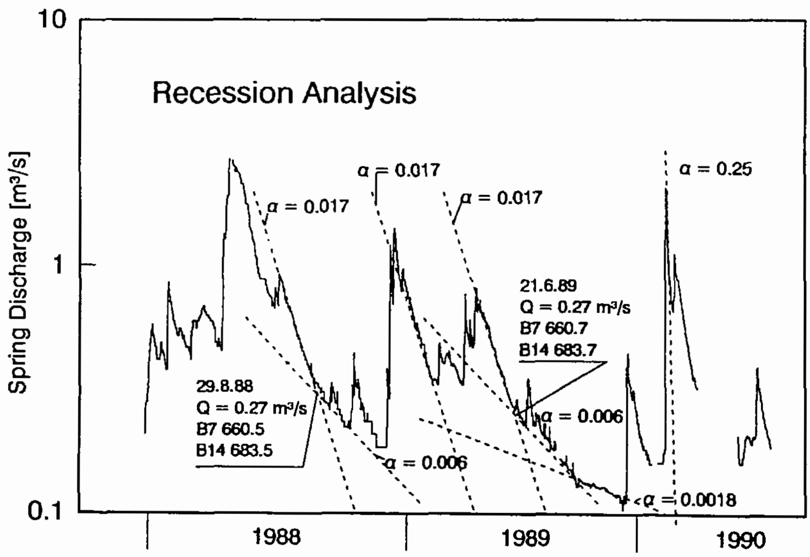

Figure 5: Discharge recession of the Gallusquelle spring

Atkinson (1977) and Trainer et al (1974) successfully applied this method to evaluate regional average transmissivities for limestone aquifers. As already discussed, horizontal variations in aquifer characteristics cannot be assumed to be detected with this method, because the dependant variable - discharge - can only be measured at one single point. However, because of the decline in water level, a recession curve reveals information on lower zones of the aquifer with decreasing discharge.

Figure 5 displays the recession characteristics of the Gallusquelle groundwater catchment over two and a half years. It can be observed that the recession coefficient decreases from ca. 0.25 d^{-1} to 0.0018 d^{-1}. It is believed that the value of 0.0018 represents the "pure fissure" and the value of 0.25 the "pure conduit" recession, because in both cases, the other flow component can be considered negligible. The values of 0.017 and 0.006 can be assumed to be a combination of K_{cond} and K_{regfis} with varying proportions. The change in slope between $\alpha = 0.017$ and $\alpha = 0.006$ occurs synchroneously with the water level dropping below the highly conductive horizon, identified in figure 3, during two consecutive years. It is not surprising that the increase in K is more prominent than in storage, considering the above explanation.

	alpha	S	L [m]	T [m2/d]	sat. thick [m]	K [m/s]
K regfis	0.0018	0.010	11000	8.83E+02	50	0.0002
"mixed"	0.0060	0.016	11000	4.71E+03	30	0.0018
"mixed"	0.0170	0.012	11000	1.00E+04	30	0.0039
K regcond	0.25	1	11000	1.23E+07	15	9.0

Table 2: Regional hydraulic conductivities (Rorabaugh, 1964)

Table 2 summarises the results and lists the respective hydraulic conductivities. The matrix value of 0.0002 m/s corresponds to the matrix value of the highly permeable south-east region, which is plausible, considering that during low flow conditions that particular region is dominating the flow due to its higher storage and transmissivity. Further upgradient it can be assumed that the actual permeable zone of the aquifer is completely drained. The conduit hydraulic conductivity also corresponds to values determined with the gradient method. The value for the recession coefficient was determined in February 1990 during a time period, when the contribution of the fissures could be assumed to be negligible. This particular event was ideal for such an evaluation because of its clean and sharp input function. The mixed values are about twice as high as the hydraulic conductivities, determined with the gradient method. This might be caused by the fact that a fixed value of saturated thickness is used to calculate K. Transmissivities compare somewhat better. Another source for differences could be that with the gradient method, hydraulic conductivities are determined for a fixed discharge, whereas the hydraulic parameters, calculated using the recession method, apply to a range of discharges.

Storage

Storage at the regional scale is generally subdivided into conduit storage and diffuse storage. Conduit storage could be evaluated with the approach suggested by Williams (1983). The values obtained varied around 0.01% ± 0.01%. Another method employed tracer tests, whereby the dye was injected into sinkholes, connected to preferential flow paths. The volume of the groundwater, discharged from the time of injection until the arrival of the dye, devided by the volume of the saturated rock produced similar values. Similarly, diffuse storage was determined by deviding the water volume discharged by the aquifer volume drained (regional changes in water levels). Diffuse storage can be assumed to vary between 1% and 2%.

Summary

Combining the above results, regional hydraulic conductivities of the fissured system ranges between $2*10^{-5}$ m/s and $2*10^{-4}$ m/s and of the fractures between a few and approximately 10 m/s. The relative proportion of the conduits per unit rock volume most likely ranges somewhere around 0.01% ± 0.01%. Total conduit and fissure storage ranges between 1% and 2%.

LOCAL PARAMETERS

Depending on the type of aquifer and the test duration, the range of investigation of pumping tests can be expected to be in the order of several hundreds of meters, i.e. the local range. Only a small number of full scale pumping tests are reported (Archive of the Geologisches Landesamt, Baden-Württemberg), that had been conducted in or close to the project area. The tested boreholes are usually located close to the river valleys, where most of the settlements are located, the risk is lowest to drill a dry hole and where also the drilling costs are lowest because of the thinner unsaturated zone.

Hydraulic Conductivity

The available pumping test data, which frequently consist of single readings were analysed, using the iterative method of Walton (1970) for the evaluation of aquifer transmissivity and making some assumptions for storage and welloss. The resulting hydraulic conductivity (K_l) ranged between $4*10^{-5}$ m/s and $1*10^{-3}$ m/s, which is in the range of regional hydraulic conductivities, determined for the fissures (K_{regfis}) of the central and the highly permeable south-eastern area of the Gallusquelle catchment. Test results of a pumping test in compact non-karstified limestone at Meßkirch yielded a hydraulic conductivity of approximately $1*10^{-6}$ m/s (K_{lmat}).

Villinger (1988) determined hydraulic parameters for two wells, drilled in the Lauchert valley near Veringenstadt and obtained hydraulic conductivities, that varied by more than an order of magnitude although the wells were only 25 m apart. The hydraulic conductivity of the first well corresponds to the values generally determined in valleys ($1*10^{-4}$ m/s) and that of the second well to the K_{regfis} of the central low permeable area of the catchment ($2*10^{-5}$ m/s). A pumping test was performed in borehole Bitz (north-west area of Gallusquelle catchment), and a hydraulic conductivity of $1*10^{-5}$ m/s could be evaluated, employing a double-porosity groundwater flow model (TRAFRAP-WT, Huyacorn, 1983).

Storage

The pumping test data from Veringenstadt allowed the determination of a storage coefficient of 0.01%, which is interpreted as the conduit storage because the highly permeable fractures dominate the flow towards the well. The storage of the fissured system at local scale could be determined from tracer breakthrough data of a forced gradient borehole-borehole test (Stober, 1991) to range between 1% and 3%, depending on the method employed for evaluation.

Summary

The values determined by large scale pumping tests are believed to represent the hydraulic characteristics of the "regional matrix" (K_{regfis}) ($1*10^{-4}$ m/s - $1*10^{-5}$ m/s), because their radius of influence, given a sufficiently long pumping period, is estimated in the order of several hundreds of meters using various steady state approximations. The borehole Bitz, situated in the north-western area of the Gallusquelle catchment reflect the hydraulic characteristics (K_{regfis}) of the central area ($1*10^{-5}$ m/s) and the results from the wells in the valleys, those K_{regfis} of the south-eastern area. The storage values correspond to those, determined at regional scale, i.e. 0.01% for the conduits and 1% - 3% for the fissure system.

SUBLOCAL PARAMETERS

The low conductivity end of the spectrum, not taking into account laboratory measurements, could only be evaluated from boreholes further upgradient. Because the unsaturated thickness is very high (≈100m) and the 2.5" casing could not accomodate powerful pumps, it was impossible to conduct full scale pumping tests. Considering the low hydraulic conductivities that had to be expected within the lower part of the aquifer, tapped by the piezometers, and in view of the the difficulties involved in assessing high well losses, it was decided to conduct slug tests and injection tests.

Slug Tests

In the present study, slug tests were conducted by displacing the water within the borehole by compressed air. The compressed air was supplied via a tank and after the well was put under a predetermined pressure, no more air was added until equilibrium was reached between well and formation head. This could take up to 45 minutes, depending on the initial change in water column. Progress of the equilibration could be monitored with two pressure transducers, the first one was placed below the lowest expected water level, the second measured gas pressure. The compressed air within the well was allowed to suddenly escape via a large valve, which did not take longer than approximately 2 seconds, and the recovery of the water level was digitally recorded. The tests were repeated at several different pressures in order to test the dependance of the test on the particular conditions of the well (Streltsova, 1988, p. 367).

Well	B8			B17				Unit
Test	Test 1	Test 2	Test 3	Test 1	Test 2	Test 3	Test 4	
Displacement [m]	1.67	4.61	11.2	1.1	2.9	6.95	8.74	m
K sl (Cooper, 1967)	1.2E-04	6.5E-05	bad fit	6.0E-05	7.8E-05	3.2E-05	3.2E-05	m/s
K sl (Ramey et al, 1975)	2.4E-05	1.9E-05	1.4E-05	5.7E-05	4.6E-05	4.1E-05	4.1E-05	m/s
Skin factor	4	9	17	45	45	45	45	
K sl (Bouwer et al, 1973)	2.3E-05	7.8E-06	2.9E-06	1.8E-05	4.2E-06	3.8E-06	3.7E-06	m/s
Model Results (TRAFRAP-WT, Huyacorn, 1983)								
K slfis	5.0E-06	3.0E-06	1.0E-09	1.0E-09	1.0E-09	1.0E-09	1.0E-09	m/s
K sl	3.0E-05	1.0E-05	5.0E-06	1.0E-05	1.0E-05	7.0E-06	7.0E-06	m/s

Table 3: Hydraulic conductivities, derived from slug test data using different analytical methods

Examples are presented for the tests in borehole B8, because they display the effects of the double-continuum system. The results of some typical tests, employing various evaluation methods, are compiled in table 3.

The data obtained from the tests were first analysed for homogeneous aquifer conditions using the Cooper et al (1967) method. Reasonable fits could be obtained for all tests, apart from B7, from about $H/H_0 = 0.6$ down to $H/H_0 = 0.1$. At the beginning, the measured curve systematically stays below the theoretical one, towards the end of the tests, the drawdown within the aquifer might not be negligible anymore. The values obtained varied from $1*10^{-4}$ to $1.3*10^{-5}$ m/s; storage coefficients were not determined because they were totally unrealistic. Barker et al (1983) also point out that they are usually a gross underestimate, even by a factor of 10^6.

The same data were also analysed with the method presented by Ramey et al (1975) and the fit appeared to be somewhat better, with however the same problems, concerning early and late data, as observed before. The hydraulic conductivity varied between $1*10^{-5}$ and $6*10^{-5}$ m/s, the skin factors between 4 and 45. Dougherty et al (1984) presented a sensitivity analysis of the effect of inner boundary conditions as there are, well storage, skin effect and partial penetration. Well storage and skin effect are accounted for by the Ramey method. The neglect of partial penetration tends to overestimate hydraulic conductivity by about a factor of 6 - 7, assuming that only about a quarter of the aquifer thickness is open to the well, which is probably the case in the tested aquifer.

The data were therefore analysed with the procedure described by Bouwer et al (1973) and Bouwer (1989), figure 6a. The tests in boreholes B8 and B7 were corrected for the initial rapid increase, which is assumed to be caused by the fast drainage of the highly permeable area immediately surrounding the well (Bouwer, 1989). By evaluating the volume of water, that suddenly entered the well, an equivalent corrected casing radius can be calculated for the initial head. The hydraulic conductivities, evaluated with the Bouwer and Rice method (table 3) were as expected much lower and varied from $1*10^{-6}$ to $2*10^{-5}$ m/s.

In order to be able to account for the geometry of the well and its boundary condition, a numerical finite element model (double-porosity) was run, to simulate the recovery data (TRAFRAP-WT, Huyacorn, 1983). The fluid exchange between fractures and matrix can be modelled as transient flow (de Swaan, 1976) and a skin surrounding the matrix blocks (interporosity skin) is integrated as well (Moench, 1984). Frequently, a pseudo steady state matrix to fissure flow (Barenblatt et al, 1960) represents the field data better, which can be achieved by setting the interporosity skin to a high value. The important calibrated model parameters are displayed in table 3. The model results vary for the tests with the highest initial displacement within a very narrow range, i.e. total hydraulic conductivities (K_{sl}, fissures + conduits) between $0.3*10^{-5}$ m/s (B14) and $0.7*10^{-5}$ m/s (B17). The fit between model and field data is generally good (e.g. figure 6b). Initial fluctuations of the test data could be either caused by the system itself that reacts underdamped or by the test procedure. The compressed air requires a finite time to escape and a standing wave might develop with compression and decompression cycles. Most of the tests could be simulated without any major matrix contribution ($K_{slfis} = 1*10^{-9}$ m/s, sub-local fissure hydraulic conductivity). Only in tests B8_1 and B8_2, where the initial displacement was lower, higher K_{slfis}-values ($5*10^{-6}$ m/s - $1*10^{-9}$ m/s) were used to improve the fit at early and late times.

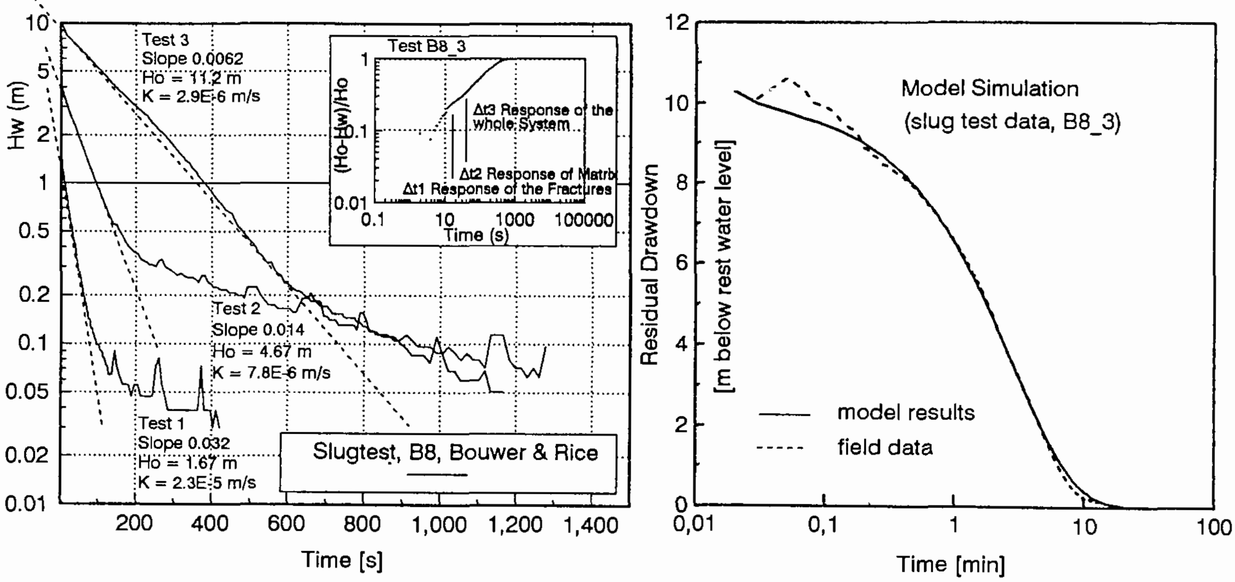

Figure 6: Estimation of hydraulic conductivity from slug test data

In trying to assess how representative the above tests are for the aquifer in question and to assist in allocating an appropriate scale of investigation to slug tests, it is important to determine a radius of influence. Ramey et al (1975) and Sageev (1986) investigated the radius of influence of slug tests and developed appropriate diagrams. A critical parameter in this assessment is the scale of resolution of the recovery measurements, which however can be easily dealt with by using pressure transducer readings. For resolutions of $H/H_o < 0.01$ and a C_D value (dimensionless well storage, Sageev, 1986, p1328) of 100, which the result is not very sensitive to, the radius of investigation is 1000 times that of the well radius, i.e. approximately 10 - 20 m.

It was mentioned earlier on that the hydraulic conductivity of the tests is dependant on the displacement depth, i.e. the duration of the test. This feature, which is attributed to the heterogeneous characteristics of the matrix/fracture system, has also been described by Streltsova (1988), who observed it whenever "the formation volume, influenced by the test is smaller than the representative formation volume required for fracture pattern replication" (p.366). The following conceptual model might explain this hydraulic behaviour. If the well is drilled within a compact matrix block, the hydraulic response of an aquifer section is mainly determined by the low matrix conductivity near the well. If however the well is connected to the fracture network, short tests will only excite the highly conductive fractures and the longer the duration of the test, the more the less conductive matrix will contribute to the test result. Therefore a systematic decrease in the hydraulic conductivity with increasing test duration (displacement) can be observed.

Injection Tests

Two kinds of injection tests were used for parameter estimation. The first ones, conducted within the context of the study consisted of the injection of water into cased boreholes (B7, B25, B14, B8). It was attempted to obtain steady state conditions and in order to circumvent the inconsistencies in flow during the injection phase, only the recovery period was used for analysis. The second kind of test are double packer tests in an uncased hole near the village Bitz (unpublished data, Landesanstalt für Umweltschutz, Karlsruhe), investigating vertical hydraulic conductivity distribution. The profile within the saturated zone was not complete because of packer leakage and therefore did not sample the sections with high hydraulic conductivities. The Deutsche Bundesbahn also kindly gave permission to use their data from double-packer tests, performed in a series of boreholes in the eastern Swabian Alb, drilled for investigatory purposes for a planned tunneling project.

The tests in boreholes B7, B8, B14, B25 were analysed first using the traditional evaluation method for Lugeon tests (e.g. Houlsby, 1976). Another standard method is the Theis recovery or Horner plot, where the section of the time drawdown curve is used for the analysis when radial flow conditions can be assumed. Barker (1981) developed an analytical technique, specifically designed for tests in fractured rocks. Transmissivity is evaluated by an iterative method using eq. 4

$$T^{(n)} = \frac{Q}{2\pi h} \ln \frac{T^{(n-1)}}{C\, r_w \sqrt{K_v K_h}} \tag{4}$$

with C as a constant (1.781), K_h and K_v the horizontal and the vertical hydraulic conductivity of the matrix.

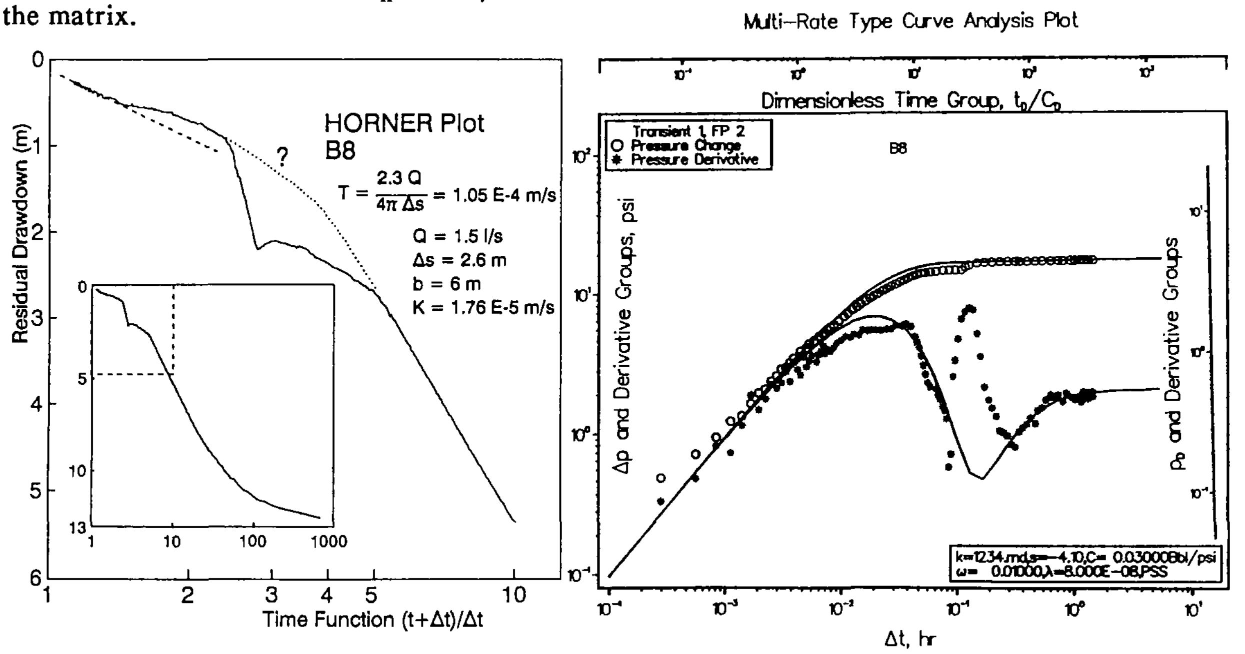

Figure 7: Estimation of hydraulic conductivity from injection test data

Figure 7a displays the Horner plot for borehole B8. The test B8 displays a delayed recovery, which is caused by water running down the side of the steel casing. By the time the drop of the water level in the borehole is slower than the speed of the water seeping down the casing, the rate in head drop is inversed. The results are however not influenced to a major degree, because the effect ceases before the onset of the infinite acting radial flow period. The S-shaped curve, a typical characteristic of double porosity aquifer tests, also becomes apparant in test B8.

Test B8 was also analysed employing the well test analysis program STAR, developed by Schlumberger. The field data could be fitted without using any skin, on the contrary, a negative skin of -4 was obtained from the model fit (figure 7b). The peak observed in the pressure derivative at 0.1 hr is caused by the above described delayed seepage down the side of the casing.

In sum, hydraulic conductivities for the injection tests in B7, B8, B14 and B25 vary between $5*10^{-6}$ m/s and $5*10^{-5}$ m/s, depending on the borehole tested. The values are generally higher than those obtained from slug tests. Assuming a regular fracture network in three dimensions and a distance between fractures of 0.2 m, the matrix block shape factor α (Warren et al, 1963) can be calculated. Together with the knowledge of λ, the interporosity flow parameter and the hydraulic conductivity of the fractures, which could be obtained from the tests, the matrix hydraulic conductivity (K_{slfis}) can be estimated to range between $5*10^{-7}$ m/s and $5*10^{-8}$ m/s.

The radius of investigation can be calculated according to Streltsova (1988, p79, eq. 6), with R_i as the radius of investigation η as the diffusivity and t the flow period. A and c_2 are constants, that depend on the definition of the radius and the system of units used. The radius of influence calculated with the above method, with c_2 being unity and A equal to 4.781, can be determined between 10 and 20 m depending on the diffusivity (m^2/d) and the time (d) used.

$$R_i = A \sqrt{c_2 \eta t} \tag{5}$$

Test Results and Hydraulic Parameters of Double Packer Tests

The test sections in borehole Bitz were preferentially located in horizons where fractures and solution channels could be detected, using a selection of borehole logs and borehole television inspection. However, highly productive zones could not be tested because of packer leakage and the limited injection rate of the testing equipment. This explains hydraulic conductivities ranging between $1*10^{-5}$ m/s and only $3*10^{-4}$ m/s.

Double packer tests in the eastern Swabian Alb yielded hydraulic conductivities between $5*10^{-3}$ m/s and $1*10^{-8}$ m/s with a maximum located between $1*10^{-6}$ m/s and $5*10^{-5}$ m/s.

Summary

At a sublocal scale, the system (conduits + fissures) hydraulic conductivity varies within a very narrow range ($1*10^{-5}$ m/s and $1*10^{-6}$ m/s). The slug and injection tests deliver fissure hydraulic conductivities, ranging between 10^{-7} m/s and 10^{-9} m/s, which correspond to the low range K_{slfis} of the double packer tests. Because of the specific test setup, no storage values could be determined.

LABORATORY SCALE MEASUREMENTS

In order to complete the spectrum of hydraulic conductivities, measured at various scales, some data could be found in Weiß (1987), which however only cover the bedded facies of the carbonate series of the Upper Jurassic (Franconian Alb). A very definite maximum was measured between 10^{-8} m/s and 10^{-9} m/s for bedded limestone. For dolomite beds, the hydraulic conductivities varied from 10^{-4} m/s to 10^{-8} m/s at the measured scale of 3 cm diameter cores. It is however difficult to take samples of the massive karstified limestone. Hydraulic conductivities can be assumed to vary by many orders of magnitudes, in the extreme case no porous or fractured medium is encountered at all, if e.g. a "sample" is taken in the centre of a cavity. Storage values were determined to range in the order of approximately 3%.

CONCLUSIONS

Figure 8 displays the karst system hydraulic conductivities of all the hydraulic tests and the regional evaluation. It is apparant that the hydraulic conductivity increases with an increasing scale of investigation. Lowest values are determined in the laboratory and the highest on a catchment scale. The values are categorised into a socalled "common range", i.e. hydraulic conductivities, that are most frequently measured and "low conductivities", which usually give an indication of the hydraulic conductivity of the fissure system at the respective scale. Among double packer tests, the higher values are usually underrepresented, which is generally a result of the testing equipment with its limited injection rate.

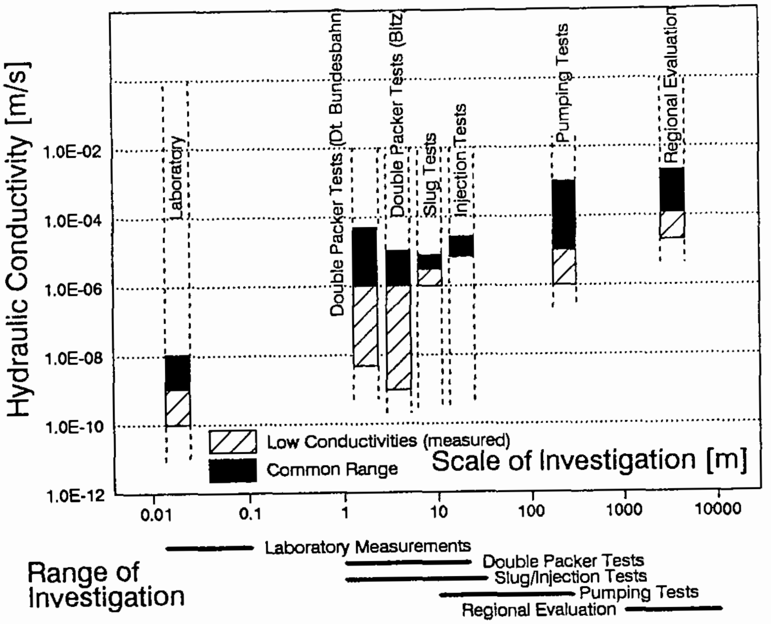

Figure 8: Relationship between hydraulic conductivity and scale of investigation

It is interesting to note that measurements at the next lower scale seem to reflect the hydraulic characteristics of the fissures at the next higher scale. Slug tests and injection tests produced fissure hydraulic conductivities of between 10^{-6} m/s and 10^{-9} m/s, which correspond approximately to the total hydraulic conductivity measured in the laboratory. The "common range" of the laboratory tests are probably an underestimate, because samples have only been taken from the bedded facies. This range can probably be extended by one order of magnitude to 10^{-7} m/s. The fissure system values, resulting from slug and injection tests (K_{slfis}) correspond to the permeabilities, measured in less karstified, i.e. fissure zones of the double packer tests. Average hydraulic conductivities measured between $1*10^{-5}$ m/s and $1*10^{-6}$ m/s by slug tests correspond to fissure system values of pumping tests (K_{lfis}), identified as hydraulic conductivities of unkarstified limestone. A similar relationship applies, if the pumping test results are compared with the regional fissure system values (K_{regfis}).

Figure 9 graphically summarises the above observations, and shows that hydraulic conductivity (fissures + conduits) of the lower scale can serve as input for fissure hydraulic conductivity at the next higher scale. Storage values do not vary to a major degree and can be estimated at 0.01% ±0.01% for the conduits and at 1% to 2% for the fissures.

These findings have been used as input for a regional double-continuum flow and transport model, which demonstrates its usefulness in problems concerning groundwater resource evaluation and contaminant risk assessment.

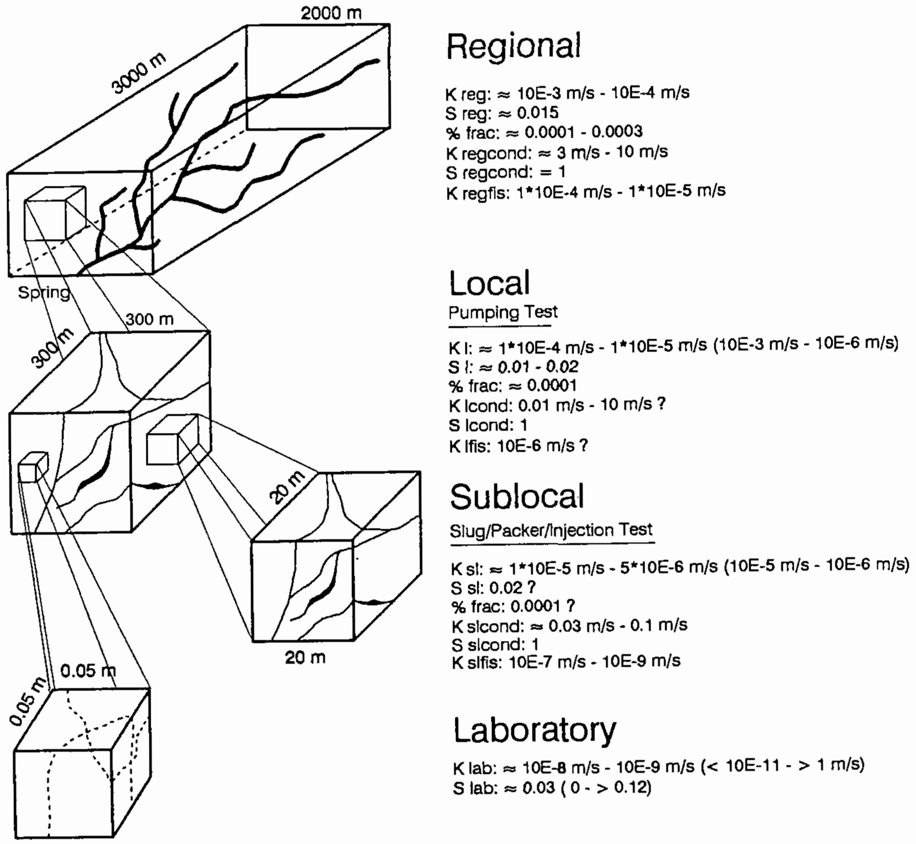

Figure 9: Geometrical relationships and hydraulic conductivities at different scales

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ASSESSMENT OF HYDRAULIC CONDUCTIVITY IN A KARST AQUIFER AT LOCAL AND REGIONAL SCALE

Martin Sauter

Question: Rane Curl has determined that many caves (perhaps most) are fractal objects which have a determinable fractal dimension. Caves are permeability (conductivity) paths of a scale. Your work seems to be approaching the problem of scale-variant parameters from a broader perspective. Could you comment on possible connections between Curl's work and yours?

Curl, R.L. 1986, Fractal dimensions and the geometry of caves: Mathematical Geology 18:765-784.

Answer: The fractal approach assumes that the same fractal dimension exists everywhere within the investigated domain and that the principle of self-similarity applies at a small and at a large scale. Fractals are typically scale-invariant. In order to achieve this self-similarity at all scales of permeable pore space, the process, generating the hydraulic conductivity should not vary in space and time. As far as caves are concerned, they usually form within a geologically short period and are the result of a single process. The aquifer studied, formed over a period of millions of years since the Miocene with the solution process modified by the glacial period, valley erosion and backfill and global and small scale tectonics, which makes self-similarity in porespace geometry and therefore hydraulic conductivity unlikely. The scale hierarchy approach does not assume self-similarity and scale-invariant behaviour but attempts to recognize the different genetic history of small and large scale permeabilities (cf fissured basalt flows and highly permeable flow tops, in Dept. of Energy, 1986, DOE/RW-0070)

APPROACH FOR DELINEATING THE CONTRIBUTING AREAS OF A WELL FIELD IN A CARBONATE-VALLEY AQUIFER

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ABSTRACT

The contributing area of a well that penetrates saturated sediments and some sedimentary rocks can be generally estimated by use of principles of flow in granular porous media. Unfortunately, ground-water flow in bedrock aquifers commonly occurs in fractures and conduits that poorly approximate granular porous media at the well-field scale. The most common methods that can be used to evaluate whether flow in a fractured-rock aquifer approximates flow in a granular porous medium include (1) borehole geophysical logging and mapping, (2) monitoring of ground-water levels, (3) slug tests, (4) dye tracing, (5) monitoring of ground-water-quality fluctuations, and (6) constant-discharge aquifer tests. If conduit flow exists, the same methods can aid in estimating the contributing area to a well or well field.

The six methods were used to show that flow in a well field in a fractured-rock aquifer in Nittany Valley, Pa., approximates flow in granular porous media under nonpumping and pumping conditions. The well field contains 12 wells in the Nittany Dolomite and Axemann Limestone of Ordovician age. Fractures and conduits are approximately evenly distributed in all directions throughout the well field and are interconnected.

The six methods show that delineating contributing areas to a well field in fractured rock can be based on methods that assume a granular porous medium. However, delineating the contributing area to the well field is difficult because of the absence of observation wells located upgradient of the field, the heterogeneity of the aquifer, and poor definition of ground-water and surface-water interactions. Because the methods used in this study were based on fixed-radius assumptions, they do not provide realistic contributing areas as do the uniform-flow and semianalytical equations and ground-water-flow modeling which integrate hydraulic gradient and aquifer heterogeneity into the solution.

INTRODUCTION

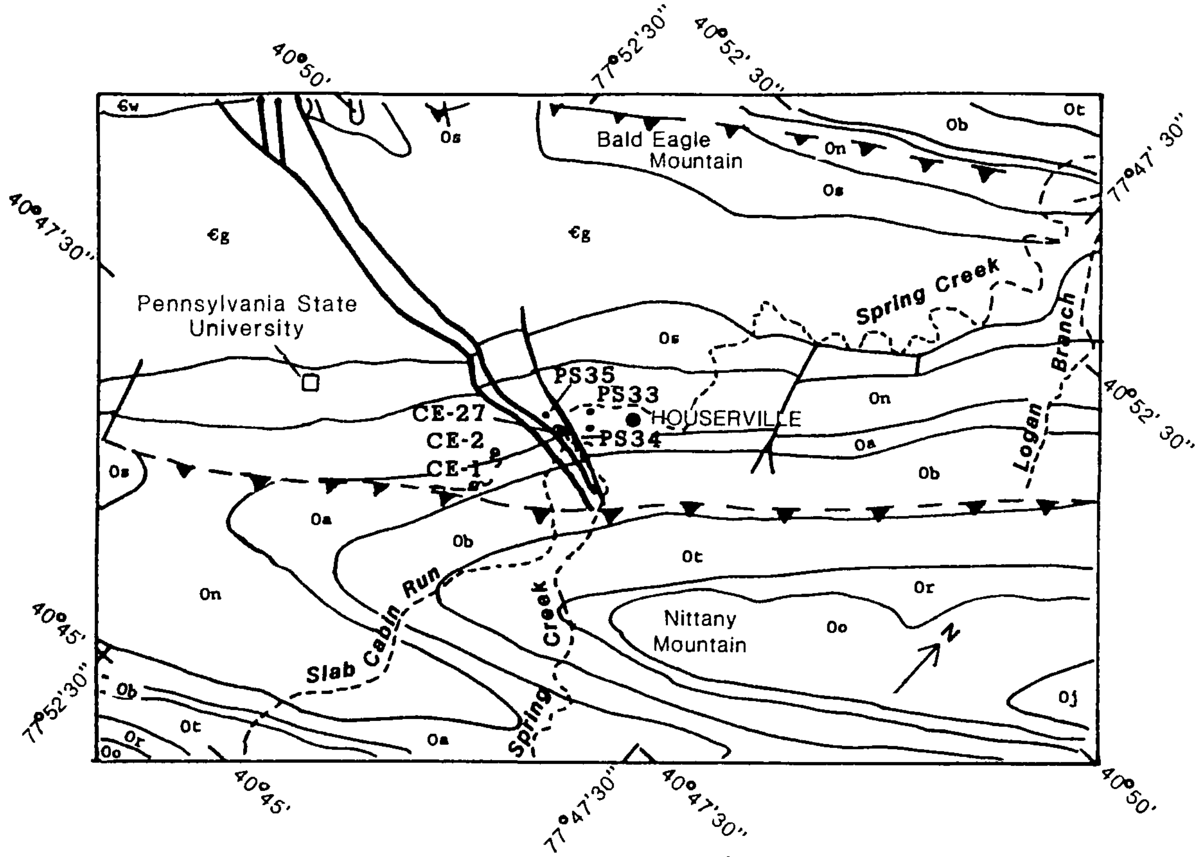
As part of the Pennsylvania Department of Environmental Resources's (PaDER) wellhead-protection program, the U.S. Geological Survey (USGS) evaluated methods for delineating the contributing area of supply wells in typical hydrogeologic settings throughout Pennsylvania. This report provides an approach for delineating the contributing area of a well field in a fractured carbonate-valley aquifer and evaluates the fixed-radius methods that can be used to delineate contributing areas to a well field within the valley. The aquifer is in the Spring Creek drainage basin within the Nittany Valley, central Pennsylvania. Topography at the well field is hummocky with vertical relief of approximately 70 ft (feet).

HYDROGEOLOGY

The Nittany Valley is located in the Appalachian Mountain section of the Valley and Ridge physiographic province. Topography is characterized by a succession of prominent northeast-trending ridges and valleys. Hydrogeologic settings similar to that of the Nittany Valley are common throughout the folded Appalachian Mountains. Ten formations, which range from late Cambrian to late Ordovician age, crop out in the study area (fig. 1). Upland areas are synclinal and consist primarily of resistant shale, sandstone, and quartzite. The Nittany Valley is anticlinal, and soluble carbonate rock crops out in the valley floor (Parizek, 1971).

The Houserville well field is located in the central part of the Spring Creek basin. Basin area is approximately 175 mi^2 (square miles). Three large-capacity supply wells completed in the Nittany Dolomite are owned and operated by Pennsylvania State University and comprise the well field. These wells are 1,250 to 1,950 ft apart (fig. 2). Nine wells, completed in the Nittany Dolomite and the Axemann Limestone, are within 2,400 ft of the supply wells (fig. 2). The Nittany Dolomite is 1,200 ft thick, and the Axemann Limestone is 400 ft thick (Wood, 1980, pl. 1). Both formations strike approximately north 45° east and beds are tilted 15° to the southeast. Transmissivity of the Nittany Dolomite is roughly one- to two-orders of magnitude higher than that of adjacent formations (table 1). No obvious relation exists between topographic position of the wells and horizontal hydraulic conductivity. A cross fault extends through the center of the well field and offsets the Nittany Dolomite and Axemann Limestone by 700 ft. Supply and observation wells are located on both sides of the fault.

Most ground water in the well field flows to the northeast along strike. The hydraulic gradient of the water table on May 16, 1990, was 0.001. In the broader area surrounding the well field, 4 mi (miles) upgradient and 2 mi downgradient of the well field, the hydraulic gradient was 0.003 in April-May, 1985 (Albert Becher, U.S. Geological Survey, written commun., 1991). The water table at the well field does not reflect topography. At the well field, Spring Creek and Slab Cabin Run are perched above the water table (fig. 1). Average discharge of Spring Creek is approximately 60 ft^3 (cubic feet per second) at gaging station 01546400 located 1.0 mi downstream from the well field (U.S. Geological Survey, 1991).



EXPLANATION

GEOLOGIC UNITS

Ob - Bellefonte Dolomite	€w Warrior Limestone
Oa - Axeman Limestone	Oo - Oswego Sandstone
Ot - Trenton Limestone	€g - Gatesburg Formation
On - Nittany Dolomite	Oj - Juniata Formation
Os - Stonehenge Limestone	Or - Reedsville Shale

.PS35 SUPPLY WELL AND LOCAL IDENTIFIER

— — — PERENNIAL STREAM

CE-2 SPRING AND IDENTIFIER

THRUST FAULT WITH TEETH ON SIDE OF UPPER PLATE. DASHED WHERE INFERRED

Figure 1.--Geology and the Houserville well field in Spring Creek basin. (From Parizek, 1971, figure 6.)

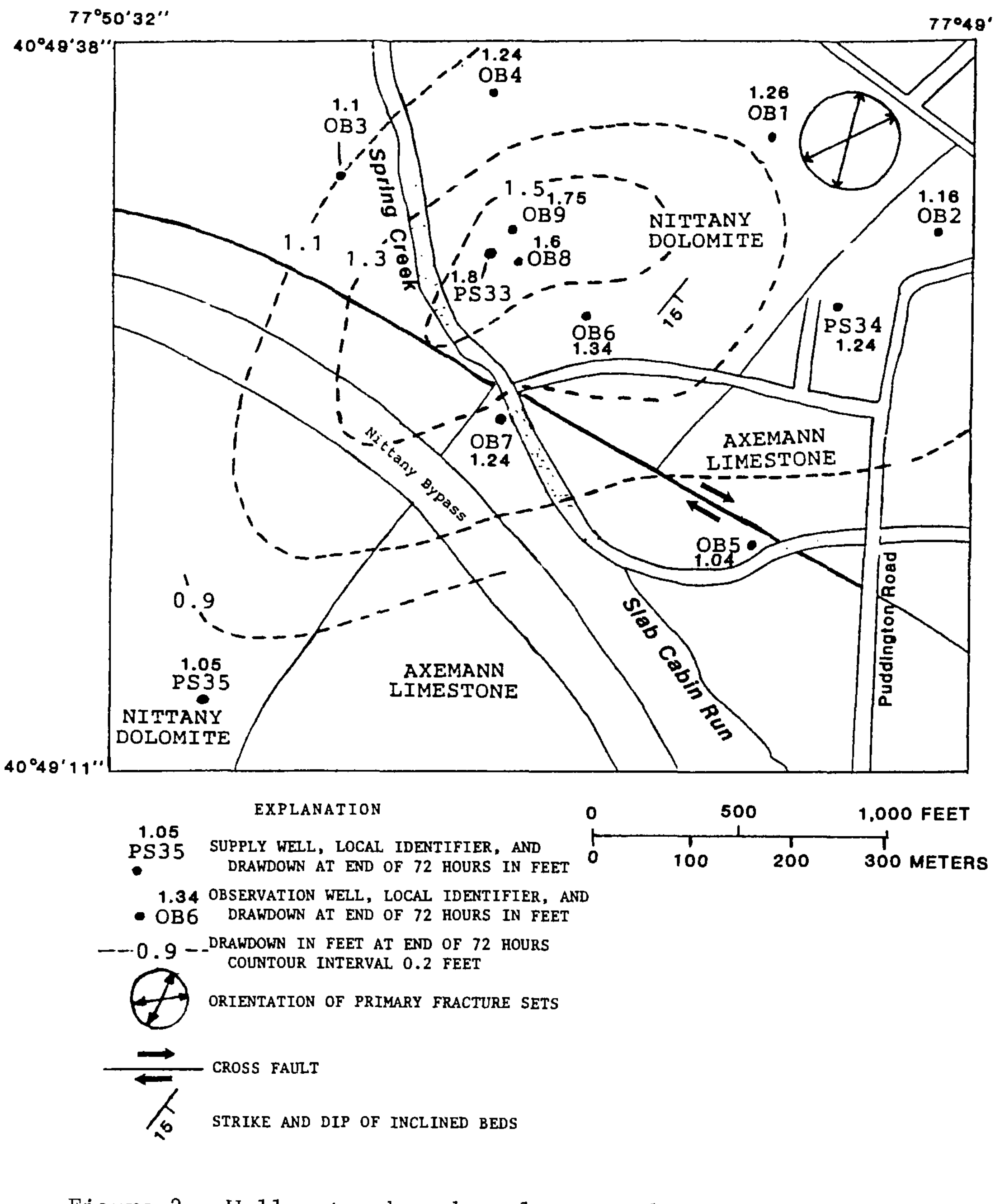


Figure 2.--Well network and geology at the Pennsylvania State University Houserville well field and drawdown at the end of a 72-hour aquifer test pumping well PS33 at 630 gallons per minute. (Geology by Parizek, 1971, figure 6.)

Table 1.--Summary of hydraulic properties
[ft^2/d, square feet per day; gal/min, gallons per minute; --, no data; NA, not applicable]

Geologic unit(s)	Transmissivity (ft^2/d)	Storage coefficient	Pump discharge (gal/min)
Trenton	[1,2] 15	[3] .015	10
Limestone	[1,4] 560	[3] .015	100
Bellefonte	[1] 40	--	50
Dolomite	[5] 240	[3] .015	--
	[6] 196 - 1,320	[6] .00027 .00062	--
Trenton and Bellefonte Limestones, and Reedsville Shale, undivided	[7] 1,000	[7] .003 .009	NA
Axemann	[1] 200	[3] .015	100
Limestone	[8] 3,700 and 7,100	--	NA
	[7] 5,000	[7] .003 .009	NA
Limestones Ordovician Age	[5] 860	--	--
Nittany	[9] 3,800	[9] .008	200
Dolomite	[1] 5,200	[3] .015	500
	[9] 120,000	[3] .015	1,000
	[8] 23,000 160,000	--	NA
	[5] 11,000	--	--
	[6] 44,919 247,059	[6] .003 .05	--
	[7] 100,000	[7] .003 .009	630 2400
Stonehenge	[1] 80	[3] .015	50
Limestone	[9] 7,600	[9] .08	500
	[7] 2,000	[7] .003 .009	NA
Gatesburg	[1] 2,000	[3] .015	200
Formation	[9] 2,700	[9] .04	500
	[9] 5,000	[9] .04	1,000
	[5] 11,000	--	--
	[6] 37,032 56,486	[6] .0003- .0006	--
	[7] 50,000	[7] .003 .009	NA

[1] Transmissivity based on specific capacity according to the method of Meyer, (1963) (Becher, U.S. Geological Survey, written commun., 1991).
[2] Data for the Coburn through Nealmont Formations (Wood, 1980, pl. 1), which are equivalent to the Trenton Limestone (Parizek, 1971, fig. 6).
[3] Transmissivity based on specific yield (Becher, U.S. Geological Survey, written commun., 1991).
[4] Data for the Benner through Loysburg Formations (Wood, 1980, pl. 1), which are equivalent to the Trenton Limestone (Parizek, 1971, fig. 6).
[5] Transmissivity based on specific capacity (Siddiqui and Parizek, 1971).
[6] Transmissivity from Rauch and White (1971, table 2).
[7] Data from transient simulation of three-day constant-discharge aquifer tests.
[8] Transmissivity based on slug tests at the Houserville well field.
[9] Transmissivity based on aquifer-test data (Becher, U.S. Geological Survey, written commun., 1991)

GROUND-WATER DISCHARGE AT AND NEAR THE HOUSERVILLE WELL FIELD

Well PS35 (fig.2) is the only active well at the Houserville well field and is pumped intermittently at 1,000 gal/min (gallons per minute); average pumping rate during 1990-91 was 254 gal/min (John Gaudlip, Pennsylvania State University, oral commun., 1991). Wells PS33 and PS34 are a permitted source of supply with a maximum withdrawal rate of 1,200 gal/min per well. Springs CE-2 and CE-27 are 4,200 and 3,800 ft (fig. 1) south-southwest of the well field in the Nittany Dolomite. Springs CE-2 and CE-27 discharge about 540 and 100 gal/min, respectively, and are a permitted source of supply at 175 and 35 gal/min. Spring CE-1 in the Axemann Limestone is 7,200 ft southwest of the well field and discharges about 4,000 gal/min.

METHODS FOR EVALUATING THE HYDROLOGIC RESPONSE OF THE AQUIFER TO NONPUMPING AND PUMPING CONDITIONS

One of the major issues in fractured-rock hydrology is whether the ground-water flow through fractures in bedrock approximates flow in a granular porous media. Hydrologic testing was conducted to address this issue at the scale of the Houserville well field. Results are used to determine whether (1) flow occurs through multiple water-bearing fractures and conduits that are sufficiently interconnected to approximate a granular porous media (referred to as an interconnected fracture-conduit flow system), or (2) flow occurs through a single or a few fracture(s) and conduit(s) that are poorly connected and do not approximate a granular porous media (referred to as a conduit-flow system). Conduits are solution-enlarged fractures that are capable of transmitting large volumes of water.

Borehole Geophysical Logging and Mapping

Fluid flow in fractures and conduits is controlled by several factors, including the density, length, orientation, aperture, surface roughness, spatial distribution, and connectivity of the fractures and conduits (Barton and Hsieh, 1989). The relation of these factors to fluid flow in a fractured-bedrock aquifer is complex and difficult to quantify (Gale, 1982). However, these factors are typically and most thoroughly measured by use of borehole geophysical logging and mapping. Measurements in many cases are limited to a fraction of the entire fracture and conduit network. Information obtained about fractures and conduits by use of conventional borehole geophysics (Paillet and Keys, 1984) is generally limited to less than 5 ft beyond the borehole annulus. The highly experimental downhole ground penetrating radar has been used to successfully map fractures and their orientations at more than 300 ft from the borehole and tomographically between boreholes (John Williams, U.S. Geological Survey, oral commun., 1991). Information obtained by mapping is limited to the surface of a rock exposure. Despite limitations of measurement and analysis, conventional borehole geophysical logging and mapping are robust methods when used to characterize the control that fractures and conduits impart on fluid flow.

Numerous fractures and conduits detected in boreholes by caliper logs, along with abundant measurements of fractures where bedrock crops out indicates that the Houserville well field is a connected fracture-conduit aquifer system. Borehole geophysical logging was conducted in all wells, except in wells PS35, OB4, and OB9, chiefly to identify fractures and conduits and to estimate their apertures. Caliper, single-point-resistance, gamma-ray, fluid-temperature, fluid-resistance and brine-trace logs, and borehole video tapes were made (Paillet and Keys, 1984). Numerous fractures and conduits were identified in all production and observation wells. Caliper logs from all 9 wells show 7 to 22 fractures per 100 ft of borehole. Fracture aperture ranged from 0.1 to greater than 1.5 ft. Fractures on bedrock that crop out in the well field were mapped, chiefly to identify the orientation of fractures. Two major sets of fractures were identified and trend approximately 30° to the strike of bedding (fig. 2) along with some minor sets of fractures with different orientations. The distance between individual fractures of each major set is approximately 1 to 2 ft. Some of these fractures extend below land surface and into the aquifer. Assuming two major sets of fractures intersect each other at many points, the connectivity of the fractures and conduits is expected to be high.

Monitoring of Ground-Water Level

A comparison of water-level fluctuations in the 11 wells monitored could indicate the relative extent of connections between fractures and conduits within the aquifer. An interconnected fracture-conduit network is likely to produce water-level fluctuations similar to those that could be expected in a granular porous media aquifer. Anomalous fluctuations in some wells could indicate a fracture-conduit network in which parts of the aquifer are poorly connected or isolated from each other. Seasonal water-level fluctuations in the well field were nearly identical in all 11 wells. Similarity in the phase, period, and amplitude of rises in water level caused by recharge suggests that all wells intercept fractures and conduits that are interconnected (Rorabaugh, 1960). The similarity in slope of water-level recessions in all wells indicate that the aquifer's hydraulic diffusivity is about the same throughout the well field. The response time of water levels to recharge can help evaluate whether the fracture-conduit network acts as an interconnected fracture-conduit flow system or a conduit-flow system (Shuster and White, 1972). In response to 2.8 in. (inches) of precipitation, water-levels in all wells rose about 8 ft over a 20-day period. This water-level fluctuation is not flashy as is typical of a conduit-flow system, but gradual and more like an interconnected fracture-conduit flow system. Some water-level fluctuations are caused by the cyclic pumping of supply well PS35 in the Houserville well field (fig. 3). Water levels in all wells respond rapidly to pumping and are most rapid and greatest near the pumped well.

Slug Tests

Slug tests provided a rapid and inexpensive method of estimating the heterogeneity of aquifer hydraulic properties. The tests were analyzed by methods that account for the inertia of the mass of water in the well and aquifer (van der Kamp, 1976; Kipp, 1985). The median transmissivity is at 79,000 ft^2/d (feet squared per day), and ranges from 23,000 to 160,000 ft^2/d wells completed in the Nittany Dolomite.

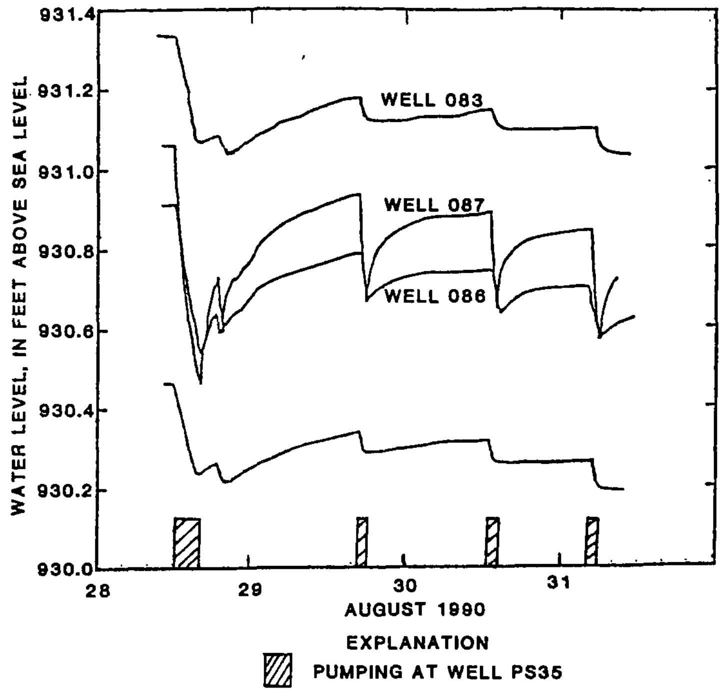


Figure 3.--Water-level flucuation in four observation wells caused by intermittent pumping at well PS35.

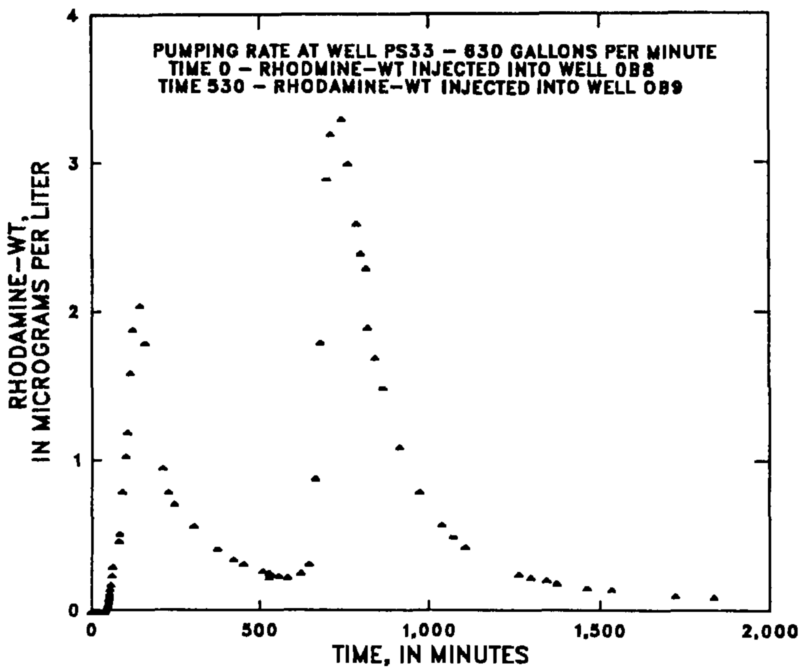


Figure 4.--Dye-recovery curve for forced-gradient dye-tracing tests.

Because the slug tests were conducted at wells having different open-hole lengths, horizontal hydraulic conductivity is a better indicator of the heterogeneity of the fracture system in the aquifer than transmissivity. Horizontal hydraulic conductivities at five wells completed in the Nittany Dolomite range from 800 to 4,500 ft/d (feet per day). The horizontal hydraulic conductivities of the Axemann Limestone at wells OB2 and OB5 are 100 and 200 ft/d, respectively. The moderate to large hydraulic conductivities at the seven wells tested indicate that these wells are capable of yielding substantial amounts of water that can only be supplied by water-bearing fractures intercepting the well bores. Therefore, slug tests indicate that fractures are approximately evenly distributed throughout the well field. The variability of hydraulic conductivity in the well field indicates that the number and spacing of fractures and conduits that intercept the wells are variable.

Dye Tracing

Forced-gradient dye-tracing tests (Mull and others, 1988; Quinlan, 1989) were conducted during a 72-hour constant discharge aquifer test in which well PS33 was pumped at 620 gal/min. Rhodamine-WT dye was injected in observation wells OB8 and OB9 equidistant (120 ft) from the pumped well (fig. 2). The mean velocities for dye traveling from wells OB8 and OB9 are 1,030 and 720 ft/d, respectively. Dye break-through curves (fig. 4) show about the same time-of-travel for dye during both tests, and the curves are smooth, bell-shaped, and positively skewed. Therefore, water-bearing fractures and conduits appear to be well connected within 120 ft of the pumped well.

Monitoring of Ground-Water-Quality Fluctuations

Variations in ground-water quality can distinguish interconnected fracture-conduit flow systems from conduit-flow systems in carbonate-valley aquifers. Ground-water specific conductance during recharge events in interconnected fracture-conduit flow systems is typically constant or gradually increases and decreases with discharge, but in a conduit-flow system, specific conductance fluctuates rapidly during and after recharge (Shuster and White, 1972, p. 1069; Jacobson and Langmuir, 1974, p. 261). Ground-water specific conductance was monitored continuously in well PS33 for 7 months. Specific conductance remained constant during recharge events (fig. 5), providing additional evidence that the Nittany Dolomite is an interconnected fracture-conduit flow system and behaves as a granular porous medium at the well field.

Constant-Discharge Aquifer Tests

Three 72-hour constant-discharge aquifer tests were conducted at 7 to 11 observations wells: Well PS35 was pumped at 1,000 gal/min, well PS33 was pumped at 630 gal/min, and wells PS33 and PS34 were pumped simultaneously at 2,400 gal/min (Todd Giddings and Associates, Inc., 1989). Water was discharged into Spring Creek, a stream perched above the water table.

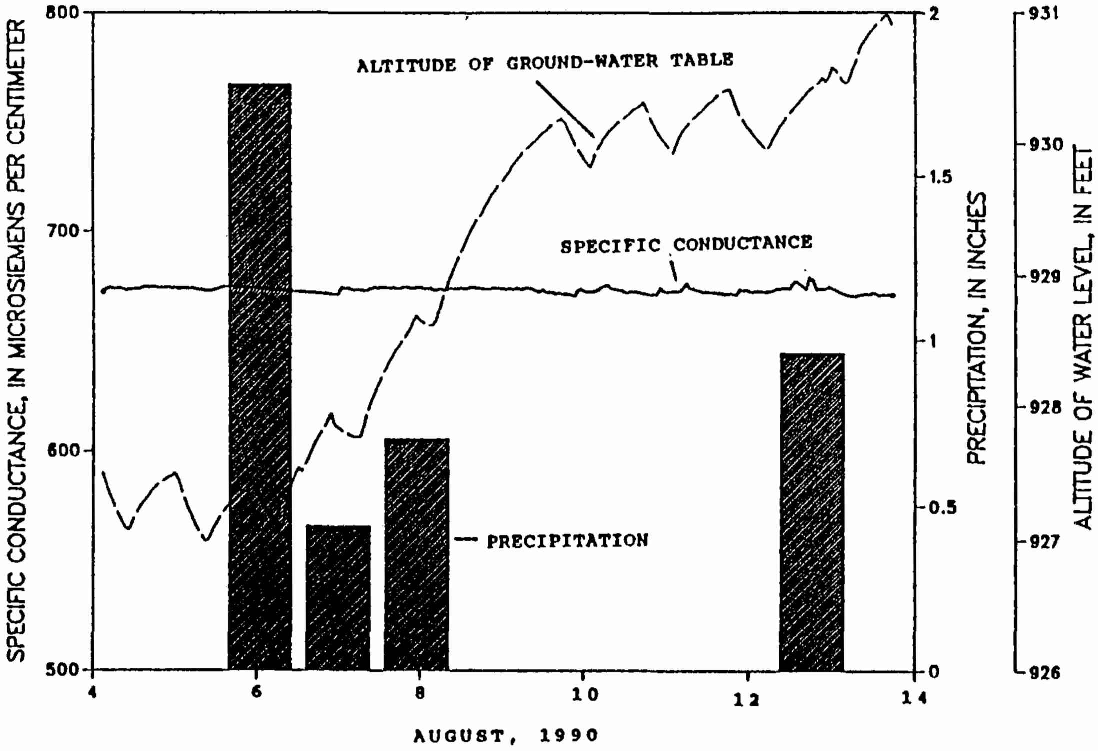


Figure 5.--Water-level altitude and specific conductance of water in observation well OB8, and amount of precipitation, August 4-14,1990.

Qualitative Analysis

The shape of the drawdown curves shows that declines in water levels did not stabilize during the aquifer tests (fig. 6a, 6b, and 6c). Therefore, the aquifer's water budget did not approach a new equilibrium during the pumping. A new equilibrium can only be established if the volume pumped is balanced by decreases in ground-water discharge to surface-water bodies, such as Spring Creek and Slab Cabin Run and springs CE-2 and CE-27 (fig. 1), or an increase in recharge from surface-water bodies such as Spring Creek and Slab Cabin Run or both. Apparently, ground water withdrawn during these tests was chiefly derived from storage.

The aquifer test at well PS33 involved monitoring water levels in 11 wells that surround the pumped well. Drawdown decreased uniformly in all directions with increasing distance between the pumped well and observation wells (fig. 2). The cross fault extending through the well field did not appear to be a barrier to, or a conduit for, ground-water flow. The lagtime between pumping and drawdown in well PS35, located on the opposite side of the cross fault as pumped well PS33, is similar to that in wells located on the same side of the cross fault as pumped well PS33.

Quantitative Analysis

The shapes of the log-log (fig. 6) and semilog drawdown curves from all three aquifer tests do not match any of the "classic" ideal drawdown curves such as those for (1) an unconfined, confined, or leaky, unconsolidated, homogeneous and isotropic aquifer; (2) a confined, fractured, double porosity aquifer; or (3) a single vertical fracture in a confined aquifer. The shape of drawdown curves commonly deviates from the shape of ideal curves because of recharge and impermeable boundaries within the pumped well's area of influence, well-bore storage, the partial penetration of wells (Kruseman and deRidder, 1990, p. 48-53), and interference from other pumped wells. The latter three conditions are not observed on the curves from the three tests; however, the shape of the drawdown curves are probably affected by boundary conditions. The nature of the boundary condition(s) could not be identified from the drawdown curves. Therefore, in order to simplify quantitative analysis of the aquifer-test data, a transient finite-difference flow model (McDonald and Harbaugh, 1988), which integrated the complex hydrogeology of the carbonate-valley aquifer into the solution, was used. The transient simulations were used to (1) estimate an average transmissivity and storage coefficient for the Nittany Dolomite and Axemann Limestone because a wide range of values have been reported for both formations (table 1), and (2) to estimate the areal extent of the area of influence at the end of 72 hours of pumping, so that boundary conditions may be better identified.

For the sake of simplicity, the water-table aquifer was simulated as a confined aquifer because drawdown is less than 10 percent of the aquifer's saturated thickness (Franke and Reilly, 1987, p. 19). The model area is 72 mi^2 (10.6 mi by 6.8 mi). Grid-cell dimensions are either rectangular or square and range from 100 ft to 2,000 ft on a side. Grid cells along the perimeter of the model were specified as no-flow cells because southeastern and northwestern model boundaries extend to and parallel the base of Nittany

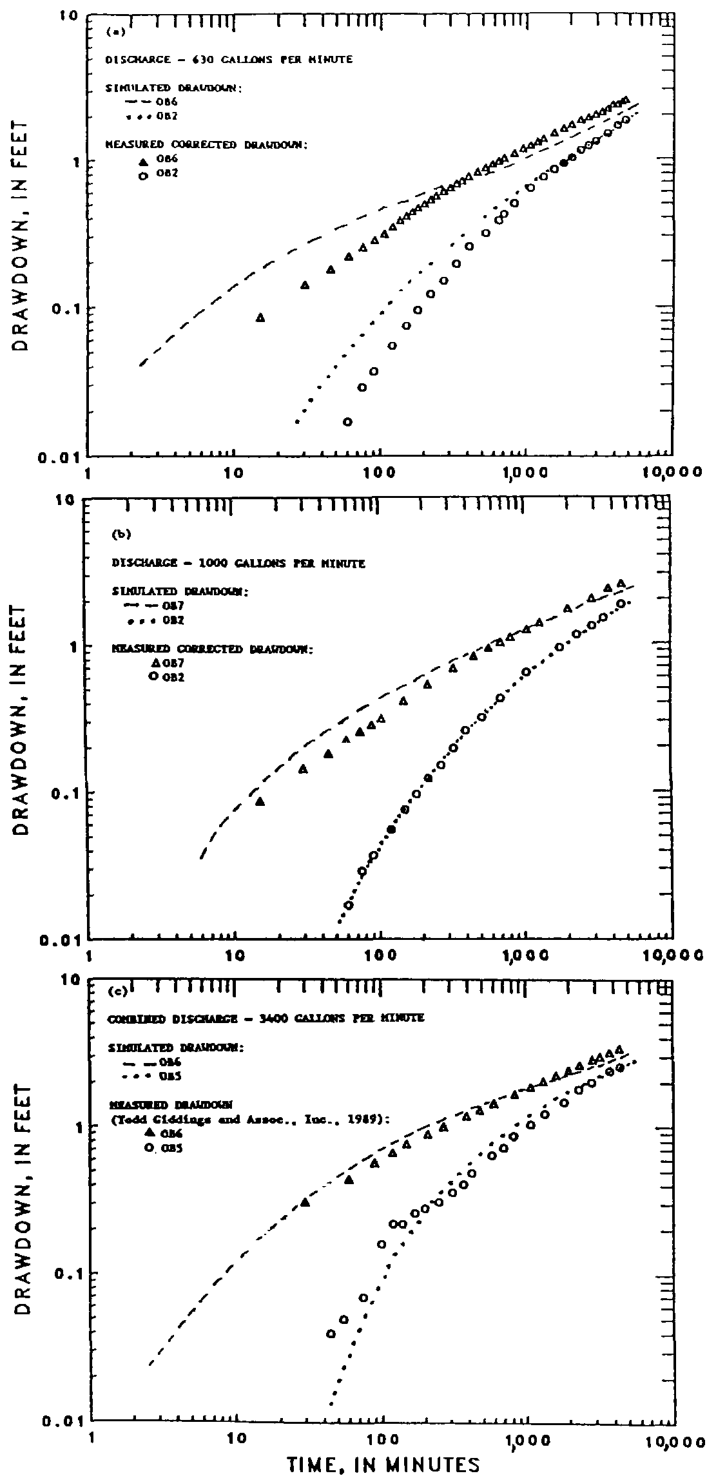


Figure 6. -- Measured and simulated drawdown for 72-hour constant-discharge aquifer tests pumping (a) well PS33, (b) well PS35, and (c) wells PS33 and PS34 simultaneously

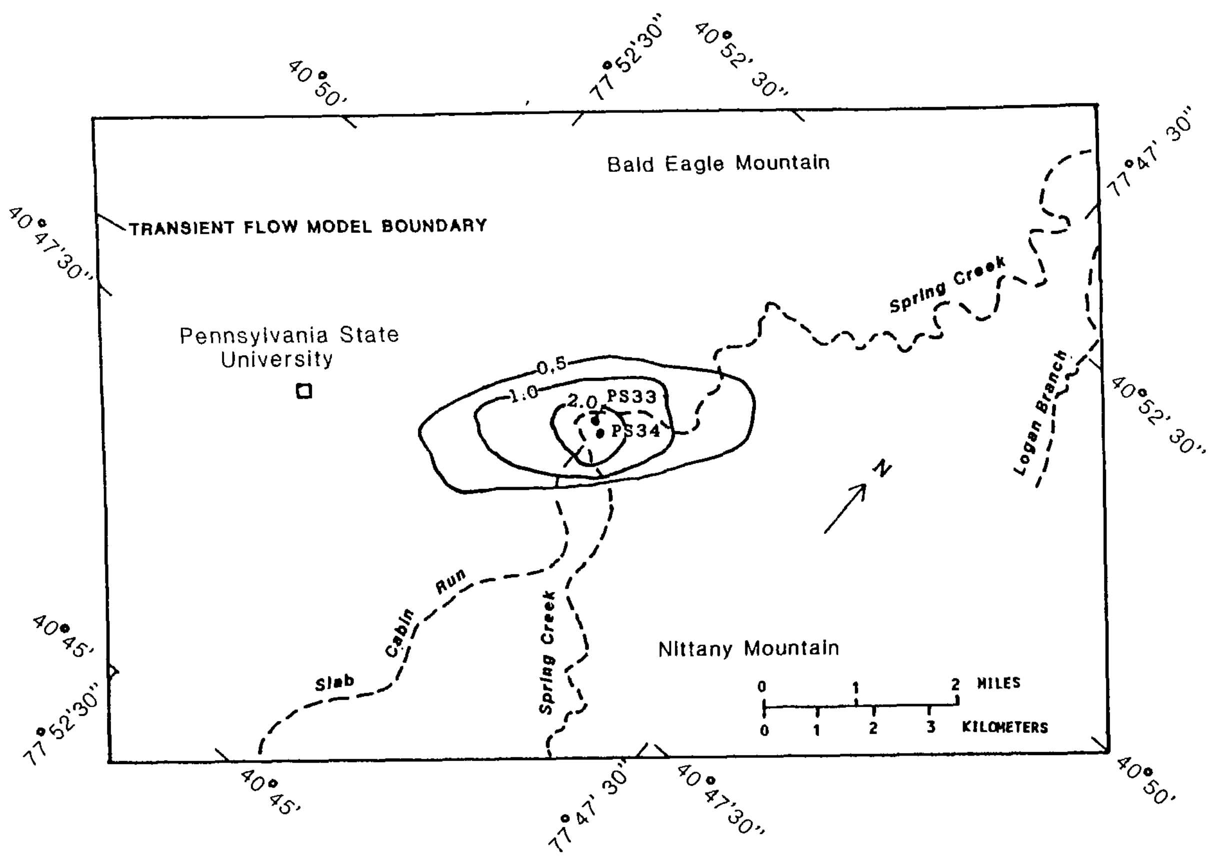
Mountain and Bald Eagle Mountain, and the southwestern and northeastern boundaries are in the valley bottom (fig. 1). Internal boundaries were not specified in the model because Spring Creek and Slab Cabin Run are perched near the well field. Sensitivity analysis aided in assessing the effects of no-flow boundaries in the valley bottom and also aided in estimating the hydraulic parameters used in the system analysis. No-flow boundaries in the valley bottom were evaluated by substituting constant-head boundaries. This substitution did not affect drawdown near the pumped well. Transmissivities used in the model are listed in table 1.

The three aquifer tests were simulated by use of transmissivities of 100,000 and 5,000 ft^2/d for the Nittany Dolomite and Axemann Limestone, respectively. The storage coefficient was assumed to be uniform for all formations. Pumping from wells PS35 and PS33 were simulated by use of a storage coefficient of 0.003 and the simultaneous pumping of wells PS33 and PS34 was simulated by use of a storage coefficient of 0.009. The need for additional storage to simulate the simultaneous aquifer test may be caused by heterogeneities in the aquifer. The storage coefficients used in all simulations are considerably smaller than the specific yield of 1.5 percent for the carbonate aquifer underlying Spring Creek basin (Giddings, 1974; Albert Becher, U.S. Geological Survey, written commun., 1991). Storage coefficients derived from short-term aquifer tests in unconfined aquifers are commonly less than the specific yield determined from long-term aquifer tests (Nwankwor and others, 1984).

The hydrographs simulated for each aquifer test (fig. 6a, 6b, and 6c) are generally in good agreement with measured aquifer-test hydrographs, especially for days 2 and 3 of each aquifer test. Simulated water levels show that pumping stress is propagated primarily within the Nittany Dolomite and Axemann Limestone for a few miles upgradient and downgradient from the well field (fig. 7). The area of influence encompasses approximately 3.5 mi^2, including springs CE-2 and CE-27, a 0.7-mi stretch of Slab Cabin Run, and 3 mi of Spring Creek.

APPROACH FOR DELINEATING AREAS OF CONTRIBUTION

Delineating contributing areas of the Houserville well field is difficult because of the absence of observation wells located upgradient (southwest) of the well field, the large heterogeneity of the Nittany Dolomite (table 1), poor definition of ground-water-flow directions, and poor definition of ground-water and surface-water interactions. A hydrogeologic model was developed and used to delineate the area of contribution to supply wells. Results of simulation of the 72-hour aquifer test pumping from wells PS33 and PS34 simultaneously at a combined rate of 2,400 gal/min, indicate that the area of influence (cone of depression) (1) develops primarily where the Nittany Dolomite and Axemann Limestone crop out because the adjacent Stonehenge Limestone and Bellefonte Dolomite act largely as barriers to flow (fig. 7), (2) is elliptical, and (3) extends into areas not monitored during the aquifer test. Borehole geophysical logging and mapping, monitoring of ground-water levels, slug test, constant-discharge aquifer test, dye tracing, monitoring ground-water-quality fluctuations, and simulation of aquifer tests show that the aquifers supplying water to the Houserville well field respond as granular porous

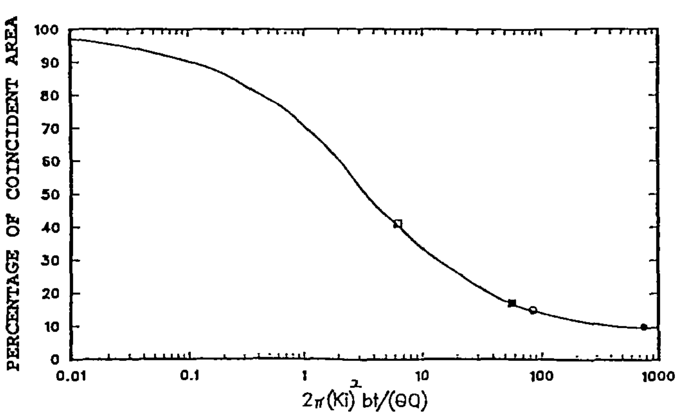


EXPLANATION

PS33 SUPPLY WELL AND LOCAL IDENTIFIER

0.5 LINE OF EQUAL DRAWDOWN OF WATER TABLE

Figure 7.--Simulated drawdown after 72-hours of simultaneously pumping wells PS33 and PS34 at 2,400 gallons per minute.



EXPLANATION

PERCENTAGE OF COINCIDENT AREA

K = HYDRAULIC CONDUCTIVITY, FEET PER DAY
i = HYDRAULIC GRADIENT
b = AQUIFER THICKNESS, FEET
t = TRAVEL TIME, DAYS
θ = AQUIFER POROSITY
Q = PUMPING RATE, GALLONS PER MINUTE
○ K=300, i=0.001, b=300, t=365, θ=0.015, Q=254
● K=300, i=0.003, b=300, t=365, θ=0.015, Q=254
□ K=300, i=0.001, b=300, t=365, θ=0.015, Q=3,400
■ K=300, i=0.003, b=300, t=365, θ=0.015, Q=3,400

ASSUMPTIONS:

(1) Flow is two dimensional and steady;
(2) Aquifer is homogeneous, isotropic, and extends infinitely in all directions;
(3) Hydraulic gradient is uniform

Figure 8.--Percentage of time-of-travel area coincident between that delinated using the fixed-radius method and a method that includes the water-table slope.

medium during nonpumping conditions and under short-term pumping stress. Thus, delineation of contributing areas of the wells by methods that assume a porous medium (Morrissey, 1987) can be made by estimating some or all of the following hydrologic characteristics: hydraulic gradient, direction of ground-water flow, aquifer thickness, porosity and horizontal hydraulic conductivity, ground-water and surface-water interactions, recharge areas, rate of recharge from precipitation, pumping rate, and pumping duration. If the hydrologic methods had indicated that flow in the fractured rock did not approximate flow in a granular porous medium, the same methods could aid in estimating a contributing area. Delineation of contributing areas by the fixed-radius method are summarized below.

Delineation of Contributing Area

The fixed-radius method is commonly used to estimate a time-of-travel (TOT) diversion zone of a well. The method is based on estimating the cylindrical aquifer volume through which ground water moves towards a well for a given magnitude and duration of pumping (U.S. Environmental Protection Agency, 1987, p. 4-6). The TOT diversion zone is the projection of this cylinder boundary at the land surface. Aquifer characteristics used in the calculations are listed in figure 8. In the case of supply wells PS33, PS34, and PS35 pumped simultaneously, discharge from each well is summed to represent an equivalent single well being pumped at a rate of 3,400 gal/min. The radius of the 365-day TOT diversion zone for pumping 254 gal/min from well PS35 is 1,125 ft and for pumping 3,400 gal/min is 3,400 ft. The percentage of a 365-day TOT diversion zone delineated by the fixed-radius method that coincides with methods that account for a sloping water table[1] is small: 10 percent when pumping PS35 at 254 gal/min with a hydraulic gradient of 0.003, and 18 percent when pumping 3,400 gal/min (fig. 8). The fixed-radius method is poorly suited for application to this well field because the method cannot account for the sloping water table. This fixed-radius method delineates an overly large zone of diversion down-gradient from the supply well(s) and too little upgradient.

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[1] Methods that delineate the contributing area(s) of a well and account for a sloping water table include use of the uniform-flow equation (Bear, 1979, p. 282), semianalytical equations such as used in the Wellhead Protection Area model (Blandford and Hyukorn, 1989), and ground-water-flow modeling. Ground-water-flow modeling is the most powerful method because it is capable of simulating complex hydrogeology; however, it also requires the greatest amount of data.

CONCLUSIONS

Borehole geophysical logging, water-level monitoring, slug tests, dye tracing, monitoring of variations in ground-water quality, and constant-discharge aquifer tests indicate that the aquifer supplying water to the well field in a fractured carbonate-valley aquifer in Nittany Valley, Pa., respond as granular porous media during nonpumping and pumping conditions. Fractures and conduits are more or less evenly distributed throughout the well field and are interconnected. Because the aquifer responds behaves as if it were a granular porous medium, areas of contribution can be delineated by techniques used for porous media. Uniform-flow and semianalytical equations and ground-water-flow modeling delineates contributing areas more realistically than does the fixed-radius method because the equations and flow modeling can account for hydraulic gradient and aquifer heterogeneity.

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APPROACH FOR DELINEATING THE CONTRIBUTING AREAS
OF A WELL FIELD IN A CARBONATE-VALLEY AQUIFER

Gary Barton and Dennis Risser

1. What you find with respect to aquifer properties depends upon how you look. Without tracer tests from sinkholes you will not understand the conduit flow component. Your information is useful, but only partly describes the aquifer. How can you justify using your data for wellhead protection when your study is so deficient in analysis aided by tracers?

   This project, as indicated in the background section of this paper, is charged with evaluating methods for delineating contributing areas to supply wells and not with wellhead protection. The data collected at the Houserville well field can be used to delineate a generalized contributing area as long as the accuracy of the delineation is indicated. Tracers injected in sinkholes (sinkholes have not been identified at the well field) or observation wells upgradient of the well field and recovered in the active supply well PS35 would increase the overall understanding of the ground water flow system and the contributing area of the well field. However, capturing a tracer in supply well PS35 during a forced-gradient test may be difficult, considering that the contributing area near and at the well field is very narrow (aquifer is highly transmissive and has a large hydraulic gradient) and therefore the tracer could bypass the pumping well.

2. Borehole geophysical logging was used to evaluate for fracture concentration, fracture orientation, and fracture thickness. What type of borehole geophysical log or logs were run to give such information? Were the geophysical logs run in open holes, water-filled holes, air-filled holes, and/or mud-filled holes?

   Borehole geophysical logging was used to estimate fracture concentration and gross thickness (aperture) and was <u>not</u> used to measure fracture orientation. Characterization of fractures was primarily based on caliper logs. Also, fluid resistance, fluid temperature, single-point resistance, and gama-ray logs were recorded. The geophysical logs were collected in air-filled and water-filled boreholes.

3. During the month of August, the specific conductance measurements did not vary. You state that this indicates interconnecting fractures and conduits. Wouldn't the opposite conclusion be expected during times of recharge -- because of the influence of lower-conductivity meteoric water?

   Please note that during the presentation I indicated that this is <u>one of several possible interpretations</u>. I believe this question is ambigious and can not respond with a clear answer.

EFFECTS OF QUARRY DEWATERING ON A KARSTIFIED LIMESTONE AQUIFER; A CASE STUDY FROM THE MENDIP HILLS, ENGLAND

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Abstract

The effects on conduit flow of dewatering, associated with development of a large limestone quarry in a karstified aquifer, are evaluated using *Lycopodium* spores and fluorescent dye tracing studies. Point to point and quantitative tracer tests are used to define the conduit network, while the nature of the conduits is indicated by travel time/discharge relations. The latter is strongly dependent on structure, strike conduits being vadose, while flow up dip involves hydraulically inefficient looping phreatic routes. During quarry deepening intersection of conduits by the quarry void has not occurred (although this has occurred elsewhere). Dewatering is shown to induce leakage from the conduit into the diffuse flow zone. Where conduits are vadose and/or distant from the quarry, leakage is minor and the conduit function is unaffected. Where conduits are phreatic, leakage prevents downstream propagation of hydraulic head over high points in the looping conduit, and the conduit becomes non-functional. Storm-related increases in conduit flow both increase leakage into the diffuse flow zone, and can cause reactivation of the conduit, as shown by increased tracer recovery at springs

Introduction

The Carboniferous Limestone of the Mendip Hills is extensively exploited to supply the demand for aggregates in southern England. Annual production is currently 20 million tonnes, much of which is abstracted from a small number of relatively large quarries concentrated in the East Mendip area. Until recently extraction was wholly from above the water-table, however 4 quarries are now actively dewatering the aquifer. Water is abstracted by pumping from sump pools in the lowest working levels, it is then ejected into perched surface streams which flow out of the area. Working quarries extending below the water-table act like pumped large diameter boreholes. As the level of operation deepens, there is a progressive increase in the associated drawdown and a widening of the cone of depression. The resultant lowering of the water-table can have serious effects upon surface hydrology and public water supplies. For example, mining activity in Hungarian karstic limestone dried up boreholes, springs and streams over an area of 2500 km^2 (Ford & Williams, 1989). There is therefore a need to assess the response of the aquifer to this abstraction, and an extensive network of observation boreholes and stream gauges has been established by the quarry companies and the National Rivers Authority (which is responsible for protection of the groundwater resources) to monitor the effects of dewatering. The National River Authority needs to know the effects of proposed dewatering schemes so that they can vet suggested quarry extensions and propose remedial measures if needed. At present it is extremely difficult to predict the exact nature and effects of dewatering karstic aquifers due to the heterogeneity associated with flow. In homogeneous aquifers, the extent of the cone of depression and the required pumping rates can be calculated using modified Thiem-Dupuit equations (Driscoll, 1986). These equations however assume homogeneity in hydraulic conductivity and laminar flow throughout the whole aquifer. Within karstified Carboniferous Limestone, the position and function of a conduit can substantially alter the nature of the drawdown surrounding a quarry (Atkinson et al, 1973a) and seriously affects calculations which are based on homogeneous aquifers. In this paper we examine a conduit network which is adjacent to an operating sub-water-table quarry and elucidate the changes in the function of this network which have resulted from dewatering, using information obtained from tracer studies.

The Study Area

Torr Quarry, owned by Foster Yeoman Ltd, is a large sub-water-table quarry on the southern limb of the Beacon Hill pericline at the eastern end of the Mendip Hills, England (Figure 1). The quarry has an area of 1.1 km^2, a current working depth of 50-60 m, and an annual production of 6-8 million tonnes. The original rest-water level in the area was approximately 150 m AOD, but in the winter of 1987, the quarry sump was deepened to 130 m AOD, and subsequently pumping levels have been between 135 and 140 m AOD. Average daily abstraction is 5.5 ML/d, but this may vary between 0.7 and 31 ML/d depending upon the weather conditions.

The Carboniferous Limestone comprises massive, well-bedded, pure, fine to coarse grained limestones of low primary porosity. In the study area the limestones dip at 30° to 40° to the south, and are overlain unconformably to the east by the Jurassic Inferior Oolite, a massive, fissured, coarse bioclastic and ooidal limestone some 15 m thick. The two limestone units are in hydraulic continuity, and provide the major water supply for the town of Frome to the north-east. At the base of the Carboniferous Limestone a shaly unit (the Lower Limestone Shales) is transitional to clastic sediments of the Devonian Old Red Sandstone, which together with Silurian volcanic rocks forms the core of the pericline (Figure 1). The Old Red Sandstone forms the highest land, rising to 288 m AOD. Water emerging from the Old Red Sandstone forms small streams, which sink after flowing onto the Carboniferous Limestone across the Downhead Fault. The

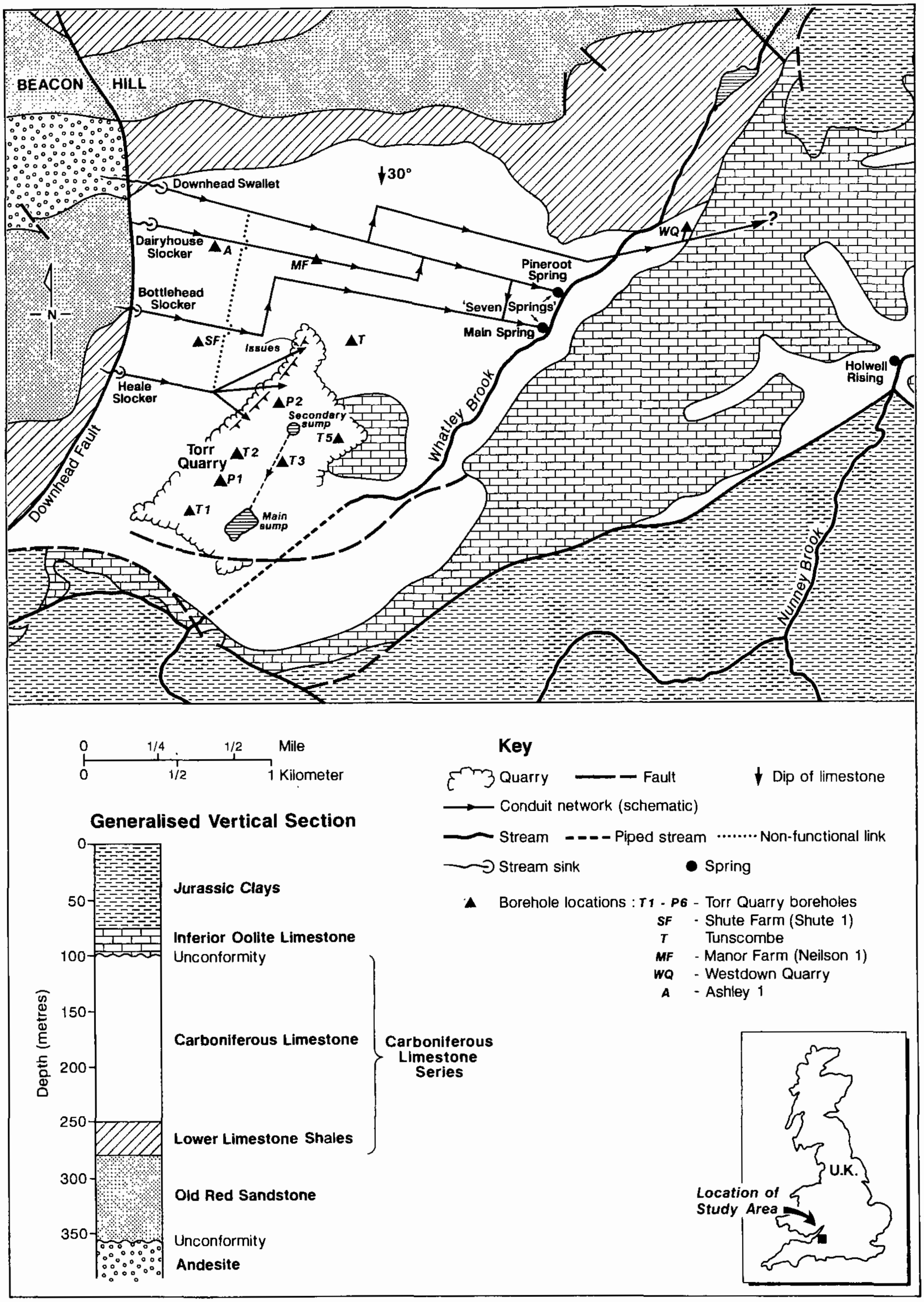


Figure 1. Geology, hydrology and hydrogeology of the study area.

main stream sinks are at elevations between 181 and 223 m AOD and comprise Downhead Swallet, Dairyhouse Slocker, Bottlehead Slocker and Heale Slocker, the latter being the nearest to Torr Quarry. Atkinson et al (1973b) have previously traced these stream sinks to Seven Springs in the Whatley Brook (128 m AOD). Although the swallets remain active throughout the year, Seven Springs ceases to flow in the summer months, and the Whatley Brook becomes influent at this point.

The Swallet/Rising Conduit Network

Tracing studies using *Lycopodium* spores (Atkinson et al, 1973b) and more recently fluorescent dyes (Smart et al, 1991), demonstrate that flow from the swallets to Seven Springs is via conduits. Velocities are high, and the dye breakthrough curves show a characteristic form with a rapid increase in concentration to the peak and a slower exponential decline on the falling limb (Figure 2). The minor peaks are related to backflooding of the spring due to the intermittant release of abstracted water from Torr Quarry into the Whatley Brook.

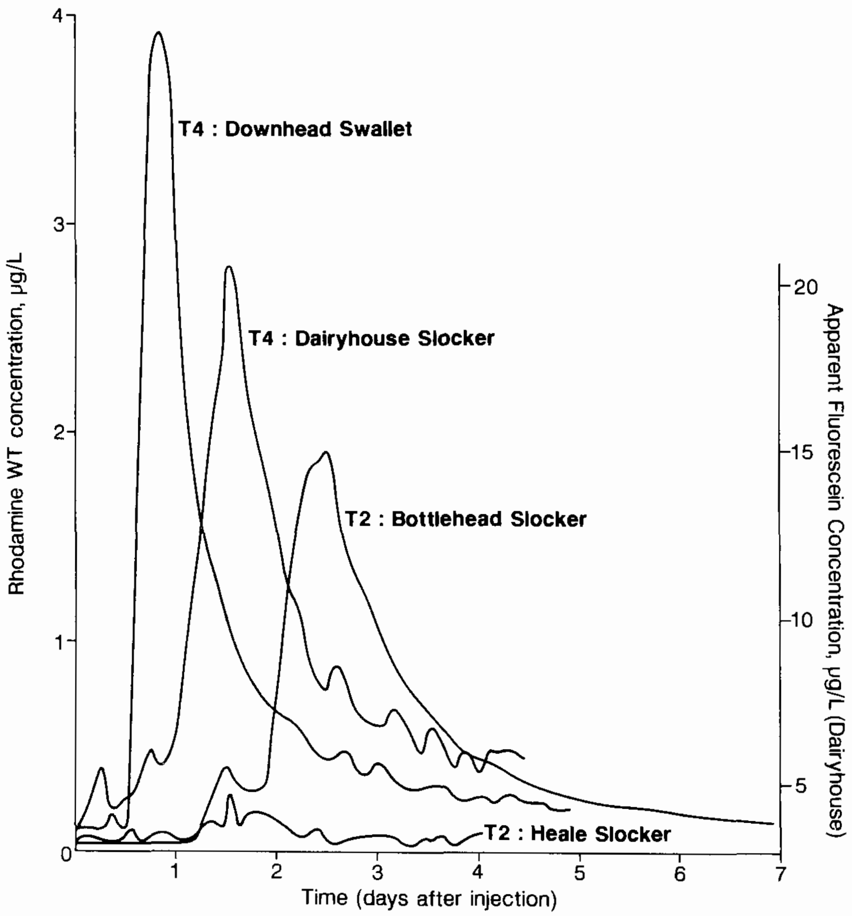


Figure 2. Tracer breakthrough curves at Seven Springs. (Main Spring) for Rhodamine WT injected at Downhead Swallet and Bottlehead Slocker, and Fluorescein injected at Dairyhouse and Heale Slockers (Tests 2 and 4).

The pattern of tracer recovery (Table 1) can be used to suggest the conduit network shown in Figure 1 (detailed arguments are presented in Smart et al, 1991). Downhead Swallet, Dairyhouse Slocker and Heale Slocker have been traced to both the Main and Pineroot Spring at Seven Springs, while Bottlehead Slocker has been proven only to the main spring. The latter is thus inferred to be a completely separate link, while the other three swallets are tributary to a single conduit. However, while Dairyhouse Slocker shows complete recovery of the tracer at Seven Springs (Table 1), Downhead Swallet has much lower recoveries (30 to 50%), and a distributory upstream of the junction with the Dairyhouse Slocker conduit must therefore occur. This distributory is shared with the Heale conduit, which like Downhead Swallet, has been traced to Westdown Quarry borehole using *Lycopodium* spores (Atkinson et al, 1973b), indicating underflow below the Whatley Brook. The final outlet for this conduit is not proven, but the consistantly 100% recovery of tracer from Dairyhouse Slocker casts considerable doubt on the positive result at Holwell Rising previously reported by Atkinson et al (1973). It appears more likely that this spring drains the area of limestone to the south east of the Whatley Brook.

Preliminary results of time of travel/spring discharge relationships for the individual swallets have been used to infer if the conduits are water-filled (phreatic) or have an open water surface (vadose) (Smart, 1981; Smart et al, 1991).

The Downhead Swallet conduit is essentially vadose (gradient log time of travel/log spring discharge 0.5), as is the Dairyhouse Slocker conduit (gradient 0.7). The higher figure for Dairyhouse infers that part of the conduit prior to the Downhead conduit junction is phreatic. Evidence presented below will demonstrate that the Heale Slocker conduit is deep below the water-table and thus phreatic at least adjacent to Torr Quarry. This accords well with expectations based on the known structural control of cave passages (Ford & Ewers, 1978). Strike directed flow may utilise bedding planes which are laterally continuous in the direction of flow, and allow development of solution conduits with relatively minor loops (state 3 or water-table cave of Ford & Ewers, 1978). Conversely flow down-dip, or in the case of Heale and Bottlehead Slockers up-dip, must utilise infrequent and discontinuous joints to pass through the dipping limestones. This results in deep looping phreatic passages (state 1 or 2 of Ford & Ewers, 1978), which are hydraulically much less efficient than strike conduits because of their greater length and the high probability of sediment constrictions at the bottom of the loops. This may explain why the *Lycopodium* spore trace results show lower velocities for Bottlehead and Heale Slockers with a high up-dip conduit component, compared to Downhead and Dairyhouse (Table 2A). Farrant (1991) illustrates precisely this structural control in the newly discovered Cheddar Springs conduit.

Influence of Torr Quarry Dewatering on Conduit Function

In the preceeding section the results of our own tracing studies and those of Atkinson et al (1973b) have been used jointly to elucidate the nature of the conduit network. Our tests were carried out in 1987 (Tests 1-4) and 1990 (Test 5) after sub-water-table working had commenced in Torr Quarry.

Table 1A Details of Traces by Atkinson et al (1973b)

Injection Site	*Tracer & Quantity (g)*		*Seven Springs Main*	*Pineroot*	*Westdown Bh+*	*Holwell*
Downhead	Lycopodium	1000	D	D	D	ND
Dairyhouse	Pyranine	300	D	D	NS	D?
Bottlehead	Lycopodium	500	D	ND	ND	ND
Heale	Lycopodium	2000	D	D	D	ND

Note: +Not accessible during recent tests

Table 1B Details of Dye Traces (this study)

Test Number	*Injection Site*	*Dye + Quantity (g)*	*Seven Springs Main*	*Pineroot*	*Combined*	*Torr Quarry*	*Holwell*
1	Downhead	RWT-47.6	D	D	40	D	ND
	Dairyhouse	FL-200	D	D	109	ND	ND
2	Bottlehead	RWT-47.6	D	ND	37	ND	ND
	Heale	CBS-200	D?	ND	D	ND	ND
3	Downhead	RWT-47.6	D	D	F	D	ND
	Dairyhouse	FL-200	D	D	F	ND	ND
4	Downhead	RWT-47.6	D	D	50	NS	ND
	Dairyhouse	FL-200	D	D	106	NS	ND
5	Downhead	CBS-1000	D	D	30	2	ND
	Bottlehead	FL-600	D	ND	86	4	ND
	Heale	RWT-120	ND	ND	0	45	ND

Key

ND : not detected
D : detected
F : samplers frozen
NS : not sampled

RWT : Rhodamine WT
FL : Fluorescein
CBS : Tinopal CBS-X

Table 2 Flow velocities calculated using straight line distance with time of first arrival and time to peak dye concentration for A) swallet/Seven Springs conduits, B) boreholes and quarry sites.

A)

Stream Sink	*Groundwater Flow Velocity (km/d)*		
	First Arrival	*Peak*	*First Arrival*[+]
Downhead	>6.7 - 4.0	6.7 - 2.45	7.4< V <10.4
Dairy House	4.0 - 3.35	2.25 - 1.4	4.2
Bottlehead	>2.40 - 1.90	2.40 - 0.92	2.4< V <2.8
Heale	? - 1.65	? 1.60	3.1< V <3.7

[+]From Atkinson et al (1973b), Table 2.

B)

Test	*Injection Site*	*Detection Site*	*Groundwater Flow Velocity (km/d)* *First Arrival*	*Peak*
Test 1	Downhead	Neilson 1	10.2	2.04
		Tunscombe	0.96	0.63
		Shute 1	0.26	0.23
		Pineroot Sp	-	0.42
	Dairyhouse	Pineroot Sp	0.60	-
Test 5	Downhead	Pineroot Sp	0.70	
		Ashley 1	2.37	0.082 (0.022)
		Secondary Sump	0.24	0.096
		Issues	0.14	0.11
	Bottlehead	Secondary Sump	0.45	0.17
	Heale	Borehole P2	>0.85	0.85
		Sump	0.48	0.14
		Issues	0.11	0.050
		Tunscombe	0.13	0.045

These can therefore be compared with the earlier experiments of Atkinson et al (1973b) performed under natural (pre-dewatering) conditions.

1. The Heale Slocker/Seven Springs Conduit

A major difference between the present and pre-dewatering results is our failure to obtain clear positive results at Seven Springs from Heale Slocker, the closest sink to the quarry. As shown on Figure 2 only a very minor barely detectable dye pulse reached the springs. The tracer was also not detected at any of the monitored boreholes (Figure 1), nor at the sump outflow from Torr Quarry. Unfortunately, Tinopal CBS-X (a blue fluorescent tracer) has a low resistance to photo-degradation (Smart & Laidlaw, 1977), and may therefore have been lost when exposed in the quarry sump. Thus, for Test 5, Rhodamine WT which has a low and stable background fluorescence and is resistant to photodecomposition, was injected at Heale Slocker. It was not detected at Seven Springs but was first present in hand samples from borehole P2 in Torr Quarry (Figure 1) some 25 hours after injection (Figure 3a). The dye breakthrough curve rose rapidly to a peak then dropped exponentially, the same response as the other swallets at Seven Springs and strongly suggestive of conduit flow. Borehole P2 is cased to a depth of 50 m, but is then open to a depth of c. 200 m (c. -50 m AOD; all other boreholes sampled are uncased or have slotted casing). Therefore the Heale conduit apparently tapped by the unpumped borehole P2 is at some depth below the quarry floor, and has not yet been intersected by removal of limestone. Borehole P2 was not present while Tests 1-4 were carried out.

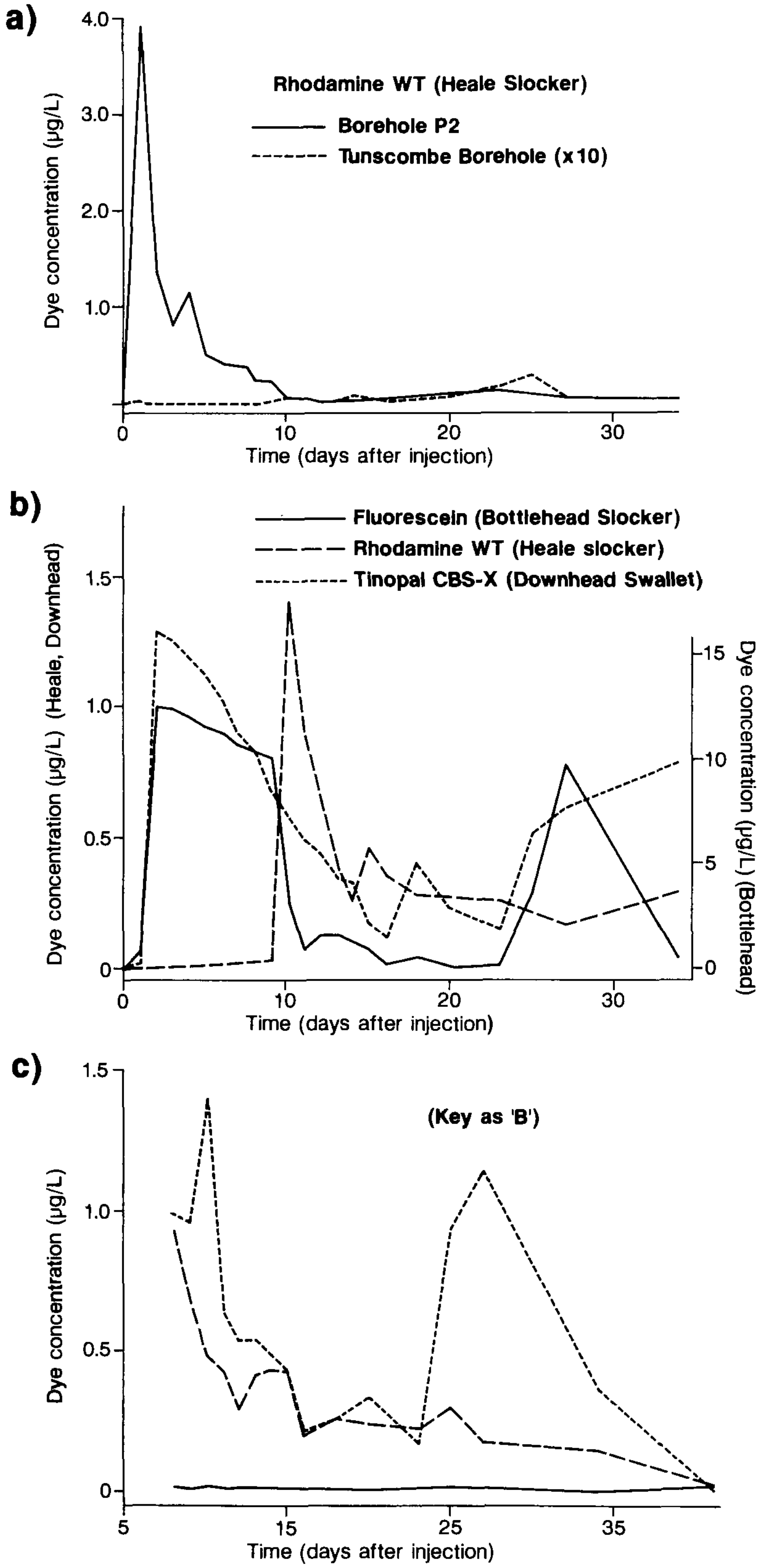
Since the completion of Tests 1-4, significant excavations of up to 30 m have occurred in the northern end of the quarry. At the time of Test 5 a secondary sump had been excavated on the quarry floor north west of the main sump, and when required to clear standing water on the quarry floor, water was pumped from this via a pipe and open channel to the main sump. Traces of Rhodamine WT from Heale Slocker were detectable in the secondary sump 4 days after injection at very low but rising concentrations which are indicative of highly dispersed diffuse flow (Figure 3b). Nine days after the injection an intense 25 mm rainstorm affected the East Mendip area, substantial areas of the north western part of Torr Quarry were flooded to depths up to 1 m. Water was observed to discharge at specific points from rubble masking the north western wall of the quarry (Figure 1). Tracer concentrations in one such issue ('Strong Spring') are shown in Figure 3c. Initially, much higher concentrations of Rhodamine WT were present than in the secondary sump, but these declined progressively with continued flow. Tracer was still present in the issues some 40 days after injection when flow ceased, and was also detected when flow recommenced after heavy rain 52 days after injection. The recovery of Rhodamine WT was budgeted using the rating for the pump discharging water from the secondary sump, which operated from days 10 to 19 after injection. The dye concentrations mirrored those from the issues, peaking initially as tracer was flushed out by the recharge event, then declining slowly, as would be expected for a diffuse flow system. Tracer recovery was estimated as 30% on cessation of pumping, but at this time hand samples from standing water on the quarry floor, combined with the observed extent and depth of flooding suggest an additional 2.5% to 3% of the tracer was present, which declined to about 0.5% on cessation of sampling. Significant tracer discharge also continued

Figure 3. Tracer breakthrough curves for Test 5:

a) Borehole P2 Torr Quarry and Tunscombe borehole for injection of Rhodamine WT at Heale Slocker.

b) Secondary sump outflow Torr Quarry for Fluorescein (Bottlehead Slocker), Rhodamine WT (Heale Slocker) and Tinopal CBS-X (Downhead Swallet).

c) Strong Spring issues Torr Quarry for Fluorescein (Bottlehead Slocker) and Rhodamine WT (Heale Slocker).



for the next 37 days (Figure 3c), but could not be formally budgeted as dilution in the main sump, which has a large volume (274×10^3 m^3) was too great. A maximum additional recovery of 12.5% is estimated from the observed tracer concentrations in the issues and their estimated combined discharge. Thus the total estimated recovery of the tracer injected at Heale Slocker is 45.5% all of which appears to have reached the quarry sumps via diffuse flow routes.

One other sample location Tunscombe borehole, showed the presence of the tracer from Heale Slocker (Figure 3a), although concentrations were very low. Tracer was first detectable following the heavy rainstorm on day 9. Then concentrations increased gradually until a further period of rain which commenced with 15 mm on day 23 and continued with smaller totals until 27 days after injection. This recharge event caused a second pulse of tracer to move into the borehole, but concentrations then fell to previous levels. Dye was no longer detectable 42 days after injection. Thus in addition to that discharged into Torr Quarry, some of the tracer from Heale Slocker underflowed the Quarry and was moving down the regional hydraulic gradient towards Seven Springs, apparently as a diffuse plume. No tracer was however detected at Seven Springs, probably because of high dilutions in the swallet conduits feeding the springs.

2. The Downhead Swallet and Bottlehead Slocker/Seven Springs Conduits

Fluorescein dye from Bottlehead Slocker was detected during Test 5 in the secondary sump in Torr Quarry within 48 hours of injection (Figure 3b), indicating a very rapid transmission. Fluorescein was not present in any of the quarry floor issues, once these entered the secondary sump and pumping commenced concentrations therefore decreased due to dilution. It was no longer detectable 16 days after injection. Recovery was estimated at 4.3%, compared to 86% at Seven Springs, giving a total tracer recovery of 90%. This is not significantly different from 100% given the probable gauging errors involved, but it is noteable that concentrations rose again in the secondary sump after pumping ceased, suggesting that unrecovered tracer remained in storage, either in the aquifer, or more probably the quarry sub-floor. No fluorescein was detected at any of the borehole sampling sites. The essentially complete recovery in Test 5 compared with only 37% recovery in Test 2 is significant and will be discussed further below.

In 1987, tracer injected in Test 1 at Downhead Swallet was not detected at the Torr Quarry main sump, but was detected in several boreholes (Figure 4). At Manor Farm, a peak occurred some 12 hours after injection, followed by a plateau-like low concentration tail. A similar tail was also observed at Tunscombe and Shute Farm boreholes, 1 and 3 days respectively after injection. The clearance of dye from the observation boreholes coincided with a decline of tracer concentrations at the Pineroot Spring (Figure 4), which has been shown above to be decoupled from the swallet conduit at low flow. This suggests that a pulse of tracer may have been moving in the diffuse flow part of the aquifer towards the spring, an observation similar to that made for the Heale Slocker (Test 5) test at Tunscombe borehole.

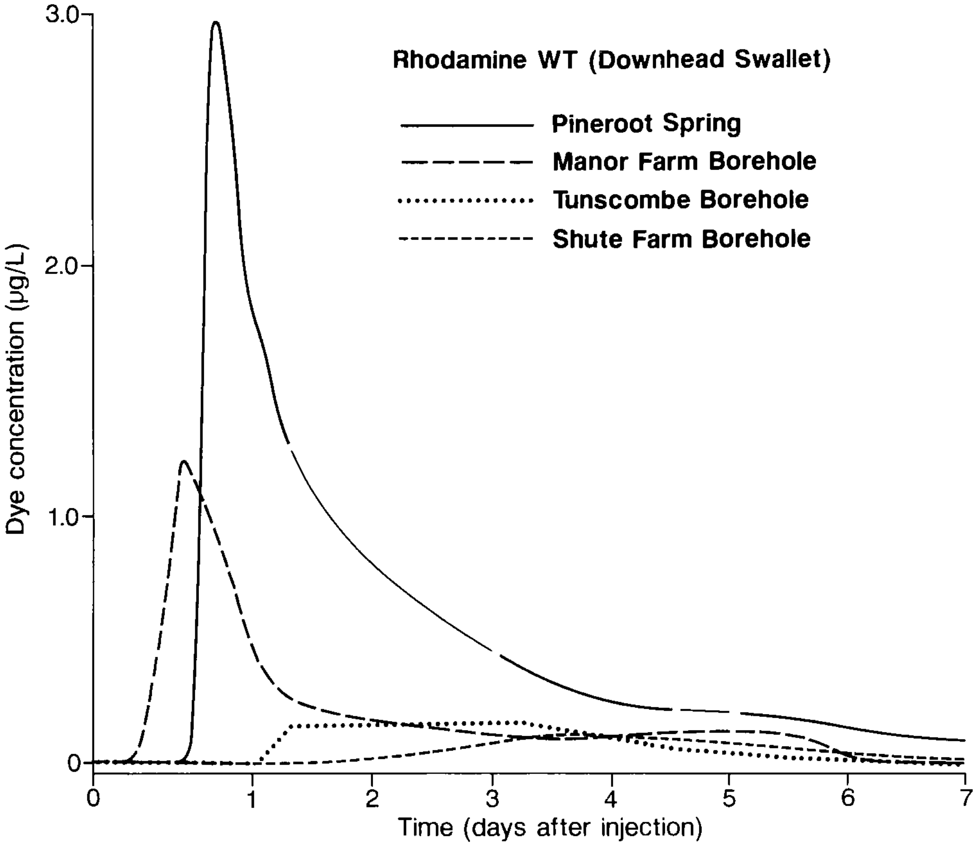


Figure 4. Tracer breakthrough curves for Rhodamine WT injected at Downhead Swallet (Test 1) at Pineroot Spring, Manor Farm, Tunscombe and Shute Farm 1 boreholes.

Test 5 was undertaken under similar groundwater level conditions at Test 1, but tracer from Downhead Swallet was not detected in either the Manor Farm or Tunscombe Boreholes (Shute Farm was not sampled). This may be partly because of the high and more variable background fluorescence in the Tinopal CBS-X (blue) waveband. In the recently drilled (1990) Ashley 1 borehole fluorescence readings in the blue waveband showed a double peak before declining to lower values 16 days after injection (Figure 5). These peaks coincide with and lag recharge events associated with rainstorms. There is however no comparable change in apparent concentration in the Rhodamine WT (orange) waveband, and the changes associated with the third rainy period (days 23 to 27 after injection) are much smaller (although sample frequency is much lower). This suggests that we are not simply monitoring changes in background fluorescence associated with organic-rich soil water arriving at the water-table, but that some of the labelled Downhead Swallet water moved from the conduit into the aquifer during recharge events. Confirmation of this association of tracer breakthrough and recharge is also apparent during Tests 1 and 3, which were undertaken during a period of generally falling groundwater-levels, onto which minor recharge events were superimposed. No tracer was detected at either the boreholes or the quarry sump for Test 3 (freezing of the samplers precluded determination of the tracers at the springs), which took place when swallet flows were in recession. In contrast Test 1, which proved positive, was associated with a minor recharge event and high swallet flows (50 L/s at Downhead compared to 24 L/s during Test 3).

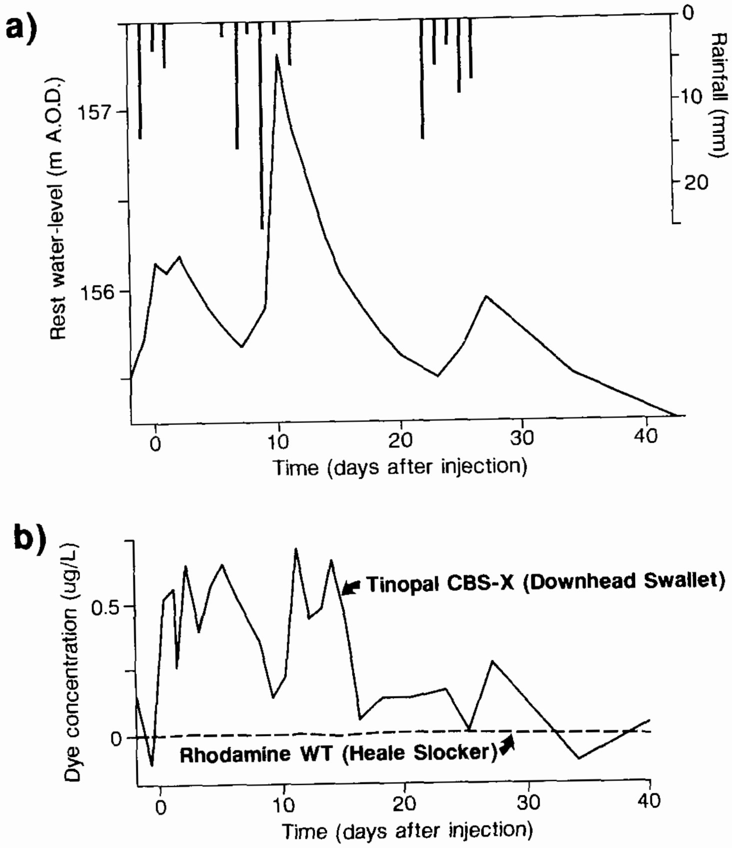


Figure 5. a) Rest water level in Ashley 1 borehole and daily rainfall at Torr Quarry.

b) Apparent concentrations of Tinopal CBS-X injected at Downhead Swallet in Test 5 and Rhodamine WT (Background fluorescence) in the Ashley 1 borehole.

In Test 5, the behaviour of the Downhead Swallet Tinopal CBS-X tracer in Torr Quarry was initially similar to the fluorescein from Bottlehead Slocker (Figure 3b), with rapid initial arrival. However, Tinopal CBS-X was also present in the new issues, (Figure 3c), thus concentrations in the sump fell less rapidly after the onset of pumping. Despite this, the tracer was rapidly exhausted, and recovery was only approximately 2% of the injected mass. Concentrations only rose again when pumping stopped. There is thus a clear contrast between the style of breakthrough from Bottlehead and Downhead swallets, where diversion from the conduits is minor and a single short pulse reaches the quarry, and Heale Slocker, where capture of the conduit flow is complete and a sustained discharge of tracer occurs via the quarry.

Discussion

The tracing results show an increasing degree of functional change in the conduit as a result of dewatering with distance from the quarry, and coincidentally between phreatic (near) and vadose (far) conduits. The Heale Slocker conduit no longer discharges to the Downhead Swallet conduit feeding Seven Springs, but there is no evidence that it has been directly intercepted by quarrying. Rather it appears that leakage by diffuse flow from the conduit under the steepened hydraulic gradient, associated with pumping from the quarry, prevents development of sufficient head within the conduit to enable pressure-flow over the downstream loop thresholds. This interpretation is supported by the widespread distribution of tracer from the Heale conduit within the Torr Quarry issues adjacent to borehole P2, which is indicative of diffuse flow, and by the movement of tracer down the natural hydraulic gradient north east of the quarry to the Tunscombe borehole. Note also that the quarry issues and Tunscombe borehole show 'flushing' of the tracer in response to rainfall events as swallet recharge pushes more dye tracer through the diffuse flow zone of the aquifer.

The Bottlehead Slocker conduit, which lies further to the north than Heale Slocker, is less directly influenced by dewatering. During Test 5 conditions were particularly wet and swallet flows high. A small pulse of tracer moved initially to the sump, suggesting leakage into the diffuse flow zone as for Heale Slocker, but the majority of the dye injected was discharged through the conduit to Seven Springs. During Test 2 swallet recharge was much less, and the hydraulic gradient towards Torr Quarry was higher because the sump was at a low level. As a result, tracer recoveries at Seven Springs were much lower (37%), the major part of the dye moving towards the quarry by diffuse flow according to the Heale Slocker model. This interpretation is supported by the very much slower travel time suggesting impairment of transmission along the conduit. In fact no tracer was recovered in the quarry, possibly because of high dilutions in the main sump, or because sampling was not continued for sufficient time. Calculated velocities for diffuse groundwater flow based on the tracer tests are typically an order of magnitude lower than for conduit flow using first arrival (Table 2A & B). However, given the much greater dispersion in diffuse flow, time to peak (or ideally time to centroid) velocities are more representative.

The Downhead Swallet conduit is least affected by quarry dewatering both because it is furthest away from the quarry, and because it is predominantly vadose. Thus although some leakage from the conduit into the diffuse flow zone occurs, as indicated by detection of tracer in the observation boreholes during Test 1, this water is not drawn towards Torr Quarry, but follows the natural hydraulic gradient to discharge from Seven Springs. In a vadose conduit leakage is independent of the head differential in the saturated zone and essentially constant. In fact the tracer evidence from Test 5 suggests that pulses of dye are expelled from the conduit during high flows, perhaps when pipe-full flow conditions occur. The movement of these pulses in the diffuse flow zone appears to be complex, and further tests are necessary to determine if the results reported are reproducible, and explainable in terms of differential head conditions imposed on the general regional hydraulic gradient towards both quarry and natural outlet.

To our knowledge this is the first well-documented example of change in the function of a conduit network as a result of quarry dewatering. Given the increasing use of this option in extraction of karstified limestones it is therefore of some interest. Changes in the conduit behaviour clearly occur even though direct intersection by the quarry void has not occured. This appears to be the result of diffuse leakage of water from the conduit which prevents pressure flow over phreatic loop-tops. Similar leakage from vadose conduits will have much less effect on conduit function because pressure flow is not involved. Despite tracer tests being the tool preferred by karst hydrologists for the study of karst conduits, hydrogeologists assessing the impact of quarry dewatering have generally adopted more conventional approaches. Perhaps the information obtained in this study may encourage a more wider utilisation of water tracing techniques for improving our understanding of both conduit and diffuse flow in karst terrains.

ACKNOWLEDGEMENTS

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BIOGRAPHICAL SKETCHES

Dr. P.L. SMART is a lecturer in geography at the University of Bristol, England, from where he also received his first degree and doctorate. He has extensive research experience in karst hydrology and geomorphology in many different areas of the world, including the Bahamas, China, Indonesia, Malaysia and several European countries. The application of natural and artificial tracer techniques to hydrological problems in karst areas has been a particular interest.

Dr. S.L. HOBBS. is a consulting hydrogeologist in a firm of Aspinwall and Co. and currently deals with a variety of projects on contaminated land, and site hydrological evaluation and instrumentation. His first degree is in Geography from Bristol University, where he also studied for his doctorate, which was on the hydrology of the saturated zone in a karstified Carboniferous Limestone aquifer.

Mr. A.J. EDWARDS is a research student in the Geography Department, University of Bristol, from where he also obtained his first degree. He is currently completing his thesis dealing with the hydrogeology of quarries and landfills in the karstified limestone aquifer of the Mendip Hills

Q. How reliable are the <u>lycopodium</u> traces and dye traces from Heale Slocker to Seven Springs that were done before the quarry was excavated to below the water table?

A. Exceedingly.

Q. Could you summarize the hydrology anticipated after cessation of quarring operations?

A. Simple computer simulations have shown that the water table, within an abandoned sub-water-table quarry in the East Mendips, may take up to 24 years before it will recover to pre-dewatering levels. This phenomenon is due to the immense volume the disused quarry void represents. If we assume excavations at Torr Quarry cease at 0 m AOD (Above Ordinance Datum, i.e., above sea level), approximately 165 million m^3 of water would be required to fill the remaining quarry void up to the pre-dewatering level. This immense lake (1.1 km^2) will dampen the seasonality of the surrounding aquifer water table, and may even have a significant effect upon the whole catchment's storage potential.

Session II:

Site Characterization

Approaches to Hydrogeologic Assessment and Remediation of Hydrocarbon Contamination in Clay-Covered Karsts with Shallow Water Tables

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Abstract

A conceptual model is presented of water and light phase hydrocarbon (LNAPL) movement through karsts mantled by clayey residual soils in which a water table is present above top of rock. This model is then incorporated into approaches to hydrogeological assessment in such karsts. These points are illustrated by examples from case histories in Alabama.

Water and hydrocarbon movement through the overlying soils and upper portion of a karst aquifer can be conceptualized in terms of the macropore systems of the soil and the subcutaneous zone of the rock. Cracks and other macropores would largely control flow of water and particularly of hydrocarbons through the soils. An underlying subcutaneous zone at and near top of rock would then converge the downward flow toward inlets into an underlying, more efficient karst underdrain system. Where a water table exists in the soils, cones of depression would be centered on these inlets while the greater water flow in the vicinity of these inlets would result in locally enhanced permeability and porosity, as well as a locally lower top of rock. A tributary system of solutionally enhanced routes along the soil/bedrock interface and in the upper few feet of rock feeds these inlets.

Assessment of contaminant movement and design of remediation in such a system would best focus on identification and characterization of the vicinity of such inlets. An assessment scheme could employ a combination of piezometers to define the effects of the underdrain system, soil and water sampling to use the existing hydrocarbon contamination as a tracer, rotary wash probe borings to locate top of rock by boring refusal and local

areas of greater permeability by water losses, and geologic characterization of the site vicinity.

Remediation could focus on source removal by treatment of contaminated soils and interception of contaminated ground water near the points where it is entering the underdrain system. Where fluid removal is appropriate, location of the pumping well in the vicinity of a an affected inlet with drawdown supplemented by vacuum is suggested to take advantage of the locally greater permeabilities presumed to be present there, improve connection to the system, and locally reverse natural gradients to stop further releases.

Introduction

This paper presents a conceptual model of the flow system overlying a conduit underdrained karst with a subcutaneous zone and significant clay cover, discusses hydrocarbon movement characteristics, and then presents approaches to characterization of such a flow system and remediation design. This paper considers the case where a water table is present in the overlying clay a significant portion of the time and results primarily from authogenic recharge.

The advantage to working in an area with a water table above the rock is that piezometers can be used as indicators of local water flow and sampling of water is facilitated before it enters the rock. However, interpretation of the resultant patterns must be based on some conceptual model.

Conceptual model of shallow water movement

This paper addresses the situation in which a karst underdrain system of solutionally enhanced conduits exists within an area. Uniformly distributed recharge from precipitation must be collected into the underlying system by some mechanism and this collection occurs because the underlying conduit system has a relatively low head due to its connection by an efficient flow path to a distant discharge area. Such a conduit system is considered to consist of a tributary system of conduits collecting water both by direct inflow from master drains connected to the top of rock in the vicinity of the conduit and by diffuse flow from the interconduit areas at a greater distance from a conduit. This paper assumes the master drains in direct connection to the conduit system are overlain by soil which moderates the inflow and assumes direct swallet access to the conduits is not present.

It is this collection system that is the subject of this paper. The basic conceptual model for shallow water flow into an underlying karst conduit was outlined in Gunn (1981) and Williams (1983). This collection system is assumed to involve discrete inlets into the conduit system and these inlets are considered to share the low head of the conduit system relative to the surrounding soil and rock. The flow toward these inlets may be either in soils (called throughflow) or in solutionally enhanced pathways at the soil/rock interface or in the upper portion of the rock (called subcutaneous flow).

Rain water percolating downward through the soils is assumed to be well distributed, but percolation may be concentrated along abundant macropores through the soil that have higher permeability to flow than the intact clay. The most important of these are expected to be the collapse cracks resulting from repeated collapse of soil cover as the top of rock is lowered over geologic time by solutional weathering processes. Such collapse cracks are subvertically oriented, continuous from the top of rock to the ground surface, dilated at the time of their formation, and very abundant. Non-plastic soils and organic matter may have been washed into these cracks as they heal.

Other sources of macropores considered significant would be cracks resulting from differential shrinking and swelling due to moisture change, including dessication cracks, and tree root holes. Observations of undisturbed soil samples and excavations in the soils of Birmingham, Alabama indicates the clays overlying limestone are typically fissured, often at very close intervals, and subvertical slickensided surfaces are common. A conceptual view of macropores in the overlying soils is presented as Figure 1.

The dense, underlying limestones are considered to be effectively impervious except at discrete discontinuities such as joints and bedding planes. The downward percolating water in the situation discussed here is assumed to accumulate at this top of rock interface as either a perched or continuous water table. From this point it must either move laterally toward a low head inlet into the conduit system, percolate into the diffuse flow system in the rock between conduits, or flow laterally through the soils toward some other discharge area.

Lateral flow through the soils is constrained by the pre-existing network of macropores, which do not readily change in response to flow. Flow within bedrock discontinuities and at

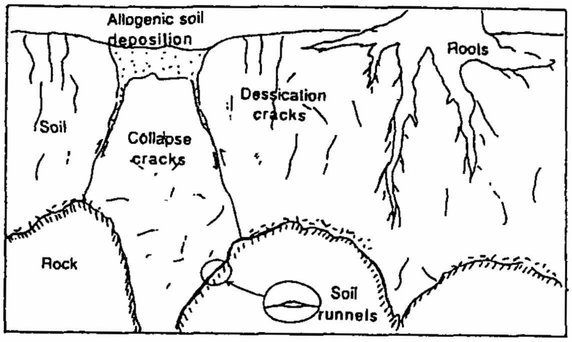


Figure 1 Common macropore types in soils

the top of rock/soil interface can change, however, due to solutional removal of rock bounding these flow routes. Such solutional removal is constrained by the kinetics of calcite dissolution as described by White (1977). For slow moving flow paths, most of the solutional enhancement occurs near the entry portions of such flow paths, but can be linked by mechanisms described in the network models of Ewers (1989).

Solutionally enhanced flow paths would be expected to form a network oriented to the low head drains, called a subcutaneous zone. A model of such a network is shown in profile in Figure 2a, from Smart and Friederich (1986) and the network connecting mechanism of Ewers (1989) would imply that in plan view such a subcutaneous system might look like Figure 2b. The enhanced flow paths would be expected to have some three-dimensional aspect. This is conceptualized to consist of discrete tributary channels, while the portions of the discontinuities between the enhanced tributary flow paths would continue to have low permeability and experience little flow.

The flow paths would be progressively interconnected toward the master drain, but, if developed like Ewers multitiered network linking models from the drain outward, would be progressively separated by divides of unenhanced permeability with distance from the drain. This suggests that relatively independent sub-basins would form within the area of influence of the master drain during development of the subcutaneous system.

The head distribution in such a flow system would produce a cone of depression centered over the master drain into the conduit as described by Williams (1983). Flow would be directed toward the low head of the subcutaneous network both laterally in the unenhanced portions of the rock and from the soil. Whether flow moves as throughflow in the soil directly toward the drain or enters into the subcutaneous system first, the cone of depression form would be expressed in the soil. In the later case, the head in the soil would reflect the entry heads into the subcutaneous system, but these would also decrease toward the main drain. As shown on Figure 2b, the cone of depression in

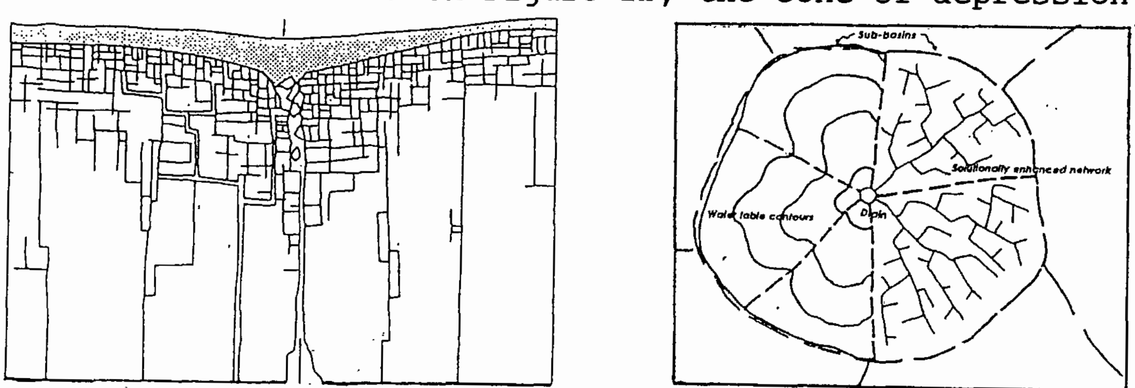


Figure 2a) Cross-section of subcutaneous zone
(Smart and Friederich, 1986)

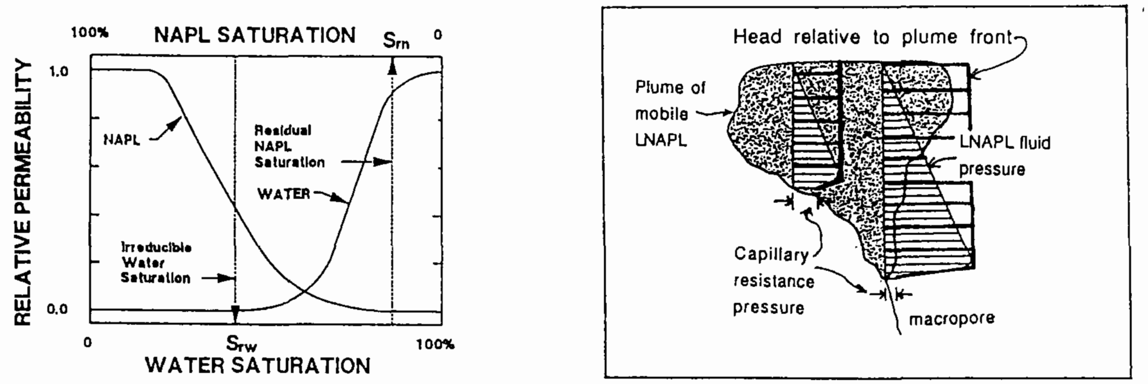
2b) Plan view of subcutaneous zone

detail should also reflect the low permeability divides between the proposed sub-basins if these are present.
In addition to solutionally enhanced flow routes developed in discontinuities within the rock, solutionally enhanced routes may develop at the top of rock interface. Patterns of solutional removal at the soil/rock interface, called runnels, could potentially develop enhanced permeability. In general, flow along this interface would be constricted as the weight of overlying soil presses the clay against the rock. As shown in Figure 1 however, a soil-roofed, rock-floored channel could develop at this interface due to solution along the floor and arching of soil above the runnel. The common observation during rotary wash drilling or coring of occasional water losses at top of rock in limestone areas might be explained by such enhanced runnels.

Aspects of hydrocarbon movement in soils

Hydrocarbon movement through clayey soils is determined both by the relative saturation of the soils with hydrocarbons and by the pore size of the soils. Figure 3a shows the two phase relative permeability relationship that governs movement of hydrocarbons in saturated soils. The three-phase relationship through partially air filled soils is similar in that permeability to a fluid is related to the existing saturation of the fluid in the medium. For movement into areas not currently containing hydrocarbons, the permeability to hydrocarbon flow at the front of the hydrocarbon plume is substantially less than that within the plume itself.

In addition, the hydrocarbon must overcome the greater attraction that water has for the soil particles. To displace water from a pore, the hydrocarbon must overcome the meniscus that expresses this greater attraction. The pressure required to overcome this meniscus will be called the capillary resistance. The strength of the meniscus varies as the perimeter of the pore or directly as the pore diameter. The expelling force varies as the cross-sectional area of the pore or the square of the pore



a) Effect of saturation on permeability
b) Pressure distribution within a plume

Figure 3 Source: EPA (1989)

diameter. Thus, a smaller capillary resistance must be overcome for hydrocarbons to expel water from a large pore than a small one. As a result, most hydrocarbon movement through soils will be through the macropores, which have a larger effective pore size, with little movement into the saturated clay itself.

A conceptualization of the movement of a hydrocarbon plume through clayey soils is shown in Figure 3b. A plume of mobile separate phase hydrocarbons is assumed to have formed due to a release from above. The shaded area in the figure contains hydrocarbons and would thus have a permeability to hydrocarbon flow many times greater than that at the edge of the plume where the hydrocarbons have not entered. The rate of flow within the plume would be constrained by the rate of advance of the front of the plume. In effect, the hydrocarbons within the plume are like in a bag created by the low permeability at the front.

As a result, nearly all of the available head would be lost at the front of the plume, producing a large head gradient there. Because of its higher permeability, only a small head loss would be required within the plume to maintain the flow rate permitted by the advance of the front of the plume. The hydrocarbon can be considered to be mounded inside the plume relative to its front.

This high head gradient at the plume front, produced by the confinement of hydrocarbons within the plume, is a significant factor in promoting advance of the plume and is most effective at the lowest part of the plume. Where the plume has advanced farthest in the vertical direction, as in the lower right portion of Figure 3b, the gradient at the plume front favoring further advance of the plume is also greatest and thus would tend to advance faster than the remainder of the plume.

To maintain itself in the pores, the hydrocarbon must have a higher pressure at this interface than the water. As a result, the capillary resistance pressure portion of the head at the plume front is unavailable to promote flow. As noted above, the capillary resistance is inversely related to pore size. Thus, where flow has followed a macropore or encounters a zone of larger pores, the available gradient for advancing the plume is further increased.

The above discussion is particularly important for a sub-vertically oriented macropore. Not only will the initial permeability be greater along such a feature, but the available gradient at the plume front in the macropore will be greater than elsewhere along the plume front due to smaller capillary resistance, and as the plume preferentially advances along the macropore the available head gradient will further increase relative to that in the remainder of the plume.

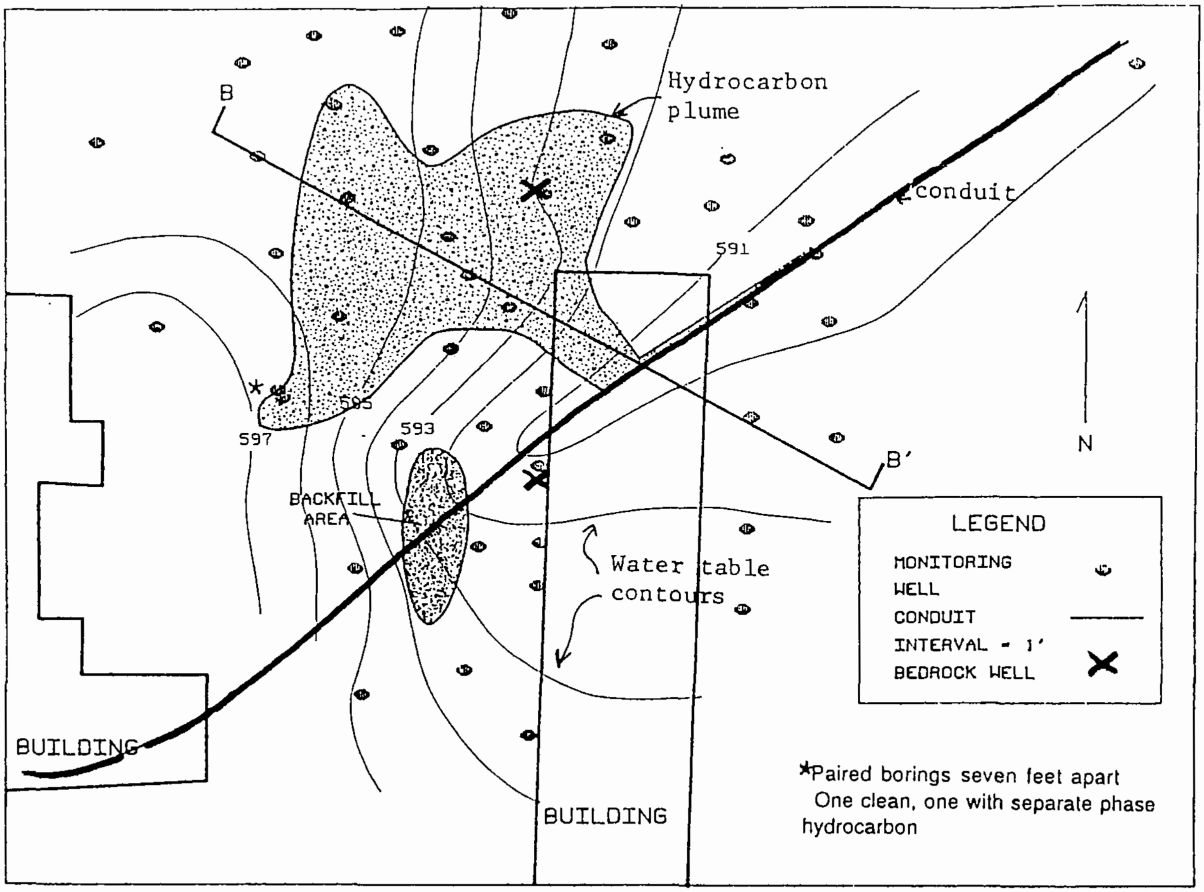
This favoring of macropores and the gradient magnifying effect of mounding within the mobile separate phase plume result in an irregular spread of the plume. Figure 4 illustrates an example from northern Alabama. In this case, a release of Number 5 fuel oil occurred in a clayey soil overlying a flat-lying limestone (Tuscumbia formation) with a water table about 3 to 5 feet above top of rock in the area of the release. A zone of greater chert content, and possibly of solutionally enhanced runnels, was present at or near the top of rock.

As seen in the cross-section of Figure 4b, the hydrocarbon plume apparently did not spread when it encountered the water table. At the point the plume just reached the water table, the difference in elevation between the base of the plume and that of the release resulted in a mounded hydrocarbon height of as much as 13 feet. This head would have been available for lateral spreading along the water table as well as for further vertical penetration. However, the capillary resistance of the small pores in the clay must be overcome for movement into clay and this may potentially be greater than that of many feet of water head.

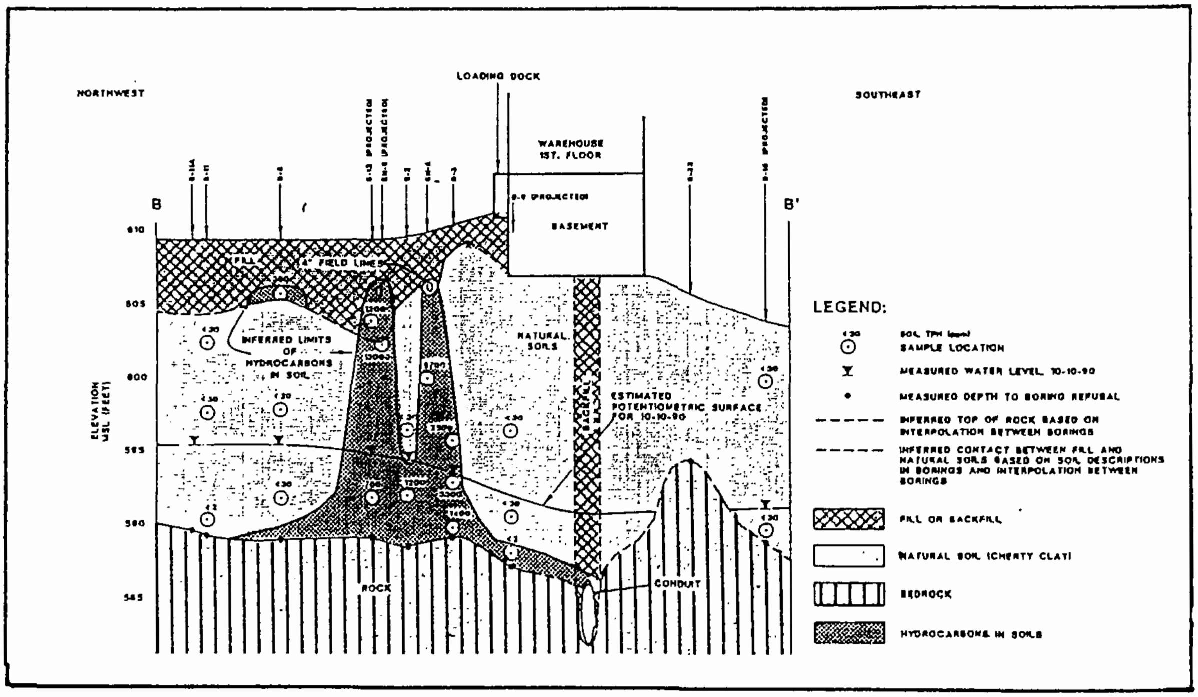
It is hypothesized that subvertically oriented macropores, such as collapse cracks or tree root holes, allowed the hydrocarbon access to the chertier or runnel-containing zone at top of rock. The large pore size of this top of rock zone then provided a route for spread of the plume at a smaller capillary resistance than that of the clay. In addition, the height of the hydrocarbon in the plume would be greater relative to the top of rock zone than at the water table, providing more available head and thus a steeper gradient to promote expansion of the plume at the deeper level.

The resultant plume, as defined by the borings shown on Figure 4a, has a very irregular form, apparently exploiting local differences in the pore size and permeability along this top of rock zone. Hydrocarbon analyses of the soils (CAL-DHS modified 8015, TPH) did not detect hydrocarbons in the soils outside the immediate source area except within a few feet of the top of rock. It appears that the greater permeability and larger pore size present near or at the top of rock controlled the spread of the hydrocarbons at this site.

As shown on the cross-section of Figure 4b, this mechanism may also allow mobile LNAPL's to penetrate into a solutionally enhanced subcutaneous zone or conduit system a moderate distance beneath the water table. In this case the surrogate for such a system is a man-made brick conduit 44 inches in diameter constructed in 1903. This conduit was at the depth shown on Figure 4b and likely had areas of deteriorated mortar that allowed water into the conduit. A finger of the plume apparently extended along the top of rock zone to the conduit as the plume advanced.



a) plume distribution and water table



b) cross-section B-B'

Figure 4 Hydrocarbon and water table distribution, Huntsville

It is concluded that once the hydrocarbon encountered the open conduit, it flowed into the conduit until it apparently drained the mounded mobile hydrocarbons that had been driving the expansion of the plume. The absence of a capillary barrier for flow into an open conduit and its extreme permeability would have favored such flow. Once the mound had dissipated into the conduit, the remaining heads within the LNAPL plume were relatively ineffective at further expanding the plume in other directions against the capillary resistance and low permeabilities of the plume front.

Approaches to assessment

The special goals of assessment considered here are: 1) determination whether the site is underdrained by a karst system, 2) location of the intakes to the conduits if present, and 3) determination of where and how to monitor. These goals supplement the goals of a conventional exploration of outlining the soil and water plumes and characterizing the hydrogeologic properties of the soils.

Approaches to achieving the above goals would include: 1) detailed definition of water table contours as a clue to flow patterns, 2) considering the site in the context of possible discharge areas and geologic information, 3) use of the observed hydrocarbon distributions in soils and water as an "opportunistic" tracer, 4) mapping the top of rock configuration, 5) pumping monitor wells before sampling to expand their radius of influence and draw water from the subcutaneous system, and 6) drilling some probe borings with rotary wash techniques and recording water losses as an indication of the presence of more permeable zones. The combination of these approaches at a particular site would depend on the judgement of the hydrogeologist and the particular conditions of the site.

The detailed delineation of the water table surface in site soils is to check for the cones of depression or other perterbations that should be associated with entries to the underground conduits. To delineate the detailed potentiometric surface, a network of piezometers and monitor wells should be installed. Most of these should be screened at the top of rock/soil interface and not extend into rock to avoid possibly interconnecting two areas of differing head. A limited number of piezometers into rock sealed a short distance into rock and paired with piezometers in the soil at the soil/rock interface are recommended to check for relative heads between rock and soil.

The definition of the water table in the soil is the target for this exploration because it is more economic, avoids the risk of spreading contamination to greater depths, and provides good information on the location of the entry points to the conduit system. If the rock communicates with the soil infrequently so that water movement is mostly by throughflow to the drains, a cone of depression defining this flow would be apparent. Alternately, if flow is mostly vertical into a subcutaneous collector system, the decreasing heads in such a system toward the drain should also be apparent in the soils near the entrance to the subcutaneous system and the cone of depression in the subcutaneous system should thus be reflected in the overlying soils. Of course, if a significant difference in head is noted between piezometers in soil and those in the underlying rock, this approach should be modified.

The greater economy of starting with a piezometer network at top of rock with the greater density of data allowed by this economy would recommend putting most of the initial phase of piezometers at this location. Most of these piezometers could be installed in unsampled probe borings once sufficient soil borings and monitor wells are installed to characterize the plume. The maximum spacing between piezometers will depend on the size of the expected cone of depression in the soils, but might be on the order of 40 to 70 feet at sites with 10 to 20 feet of soils if there are little topographic clues. The borings for the piezometers may also provide top of rock information.

Figure 5a is an example of such a detailed water table definition. This shows the elevation of the water level in soils at the site and indicates that water flow is to the southeast and southwest with a groundwater divide within the site. Gradients are fairly steep, being about .05 to .07, while the overall pattern suggests drain locations off-property to the southeast and southwest, though the divide may be simply a sub-basin divide within a larger general cone of depression.

The obtained water table and other potentiometric data are then considered in the context of topographic and geologic clues to potential discharge areas and flow patterns. As shown on Figure 5b, the site is located between three possible discharge areas to local creeks. The maximum available gradients to these areas

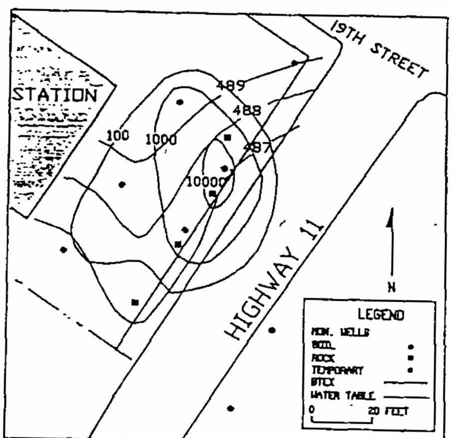
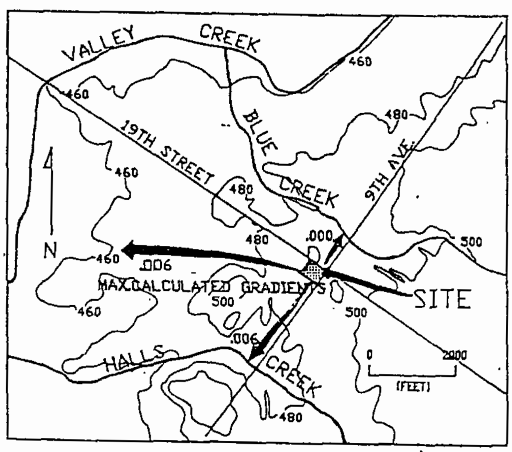


Figure 5a) Site water table and BTEX contours



5b) Site context, potential discharge areas from the lowest water levels recorded on site was calculated as the remaining drop in elevation divided by the straight line distance to the potential discharge area. The actual gradient of flow to these areas is likely less both because the straight line distance is probably not the flow path and the on-site gradients are probably not at the lowest possible point. The maximum possible gradients were .006 to two of the areas and zero to the third, ruling out the later. The factor of ten difference between on-site gradients and the maximum available gradients to move on-site water the rest of the way to the discharge area confirms underdrainage of the area by a more efficient system than the soils.

The directions to the potential discharge areas are discrepant to those in the soils, also indicating underdrainage. The geologic information that the site is underlain by dipping limestone striking northeast parallel to a major fold axis provides an additional clue that the deeper flow is toward the southwestern option, but this was not confirmed at this site.

The pattern of hydrocarbons in the soils and ground water can also be used to indicate flow direction, as well as travel times if the time of release is definitely known. A dissolved BTEX (benzene, toluene, ethyl benzene, and xylenes) plume on Figure 5a indicates that the water table defined by the monitor wells is compatible with the actual water flow.

Since the hydrocarbon plume represents true long-term flow patterns, this will generally be more reliable than the current water table pattern at some sites as site work or seasonal effects may change water table recharge patterns controlling the flow patterns shown by piezometric data.

The top of rock configuration can also provide clues for entries to the rock system. Where water flow into the rock is concentrated, it is common for the increased flow to have locally removed more rock, resulting in a depression in the top of rock. Top of rock data does not necessarily indicate current activity, however, and should be interpreted in terms of associated potentiometric data because a low top of rock may result from other patterns of ground water flow or deposition of sediment in conduit systems may have changed patterns of flow over geologic time.

Figure 6, from the northern Alabama site discussed earlier, shows the top of rock data gathered during piezometer installation together with one of the obtained potentiometric surfaces. One clearly defined low area extended to a depth of 9 feet deeper than nearby borings in the area just west of the eastern building on-site. Nevertheless, this is not reflected in the water table surface even though a piezometer was placed inside this low area. In addition, a Type III monitor well installed in the rock 50 feet to the north showed a head in the rock 1.8 feet higher than that in the piezometer in the depression. The

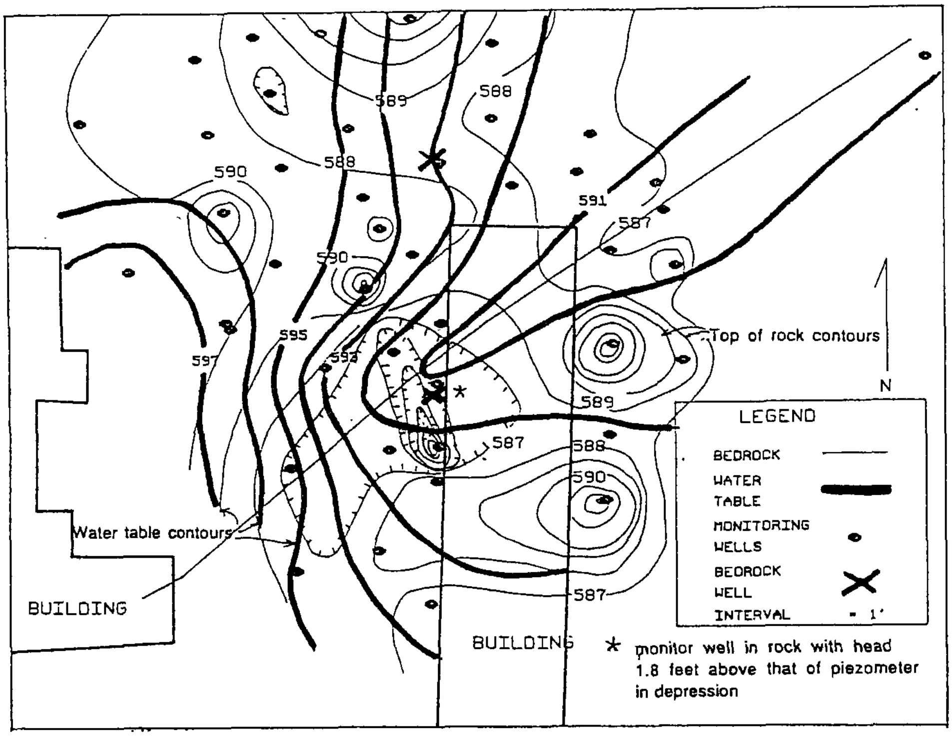


Figure 6 Rock and water table contours, Huntsville.

top of screen of this rock well was one foot above the bottom of the piezometer in the depression. Considering that the site is underlain by flat-lying rock, the bedding planes connecting to this depression were apparently not enhanced in permeability or were clogged with sediment as there was no mound of upwelling water on the water table as would be expected for such a gradient towards the depression.

Relative to monitoring a site, the above characterization of the flow regime is important for interpreting the data gathered and checking that monitoring wells are correctly located. If much of the movement is in the soils as throughflow, as indicated by the plume outline, then monitor wells screened at top of rock may be sufficient. However, if the water flow is into a subcutaneous zone in the rock, even though piezometers may correctly show a cone of depression and indicate the direction to the master drain, conventional monitor wells may miss downward moving dissolved contamination. Monitor wells sealed into the shallow upper rock would be an appropriate supplement to monitor wells just in the soil.

However, in both the case of monitor wells in the soil and those in the rock, as illustrated by Figure 2b, a passive well may miss the flow of contaminated water due to the convergent nature of flow toward the favored flow routes. For this reason, it is recommended that for slow flowing wells, which thus indicate poor communication to the system, the purging operation be continued until on the order of 10 to 20 gallons have been obtained. This may require use of a supplementary vacuum to assist removal of this much water and then a recovery period to eliminate the effect of the vacuum on the volatiles. The reason for this is to temporarily reverse the local gradients to cause flow of water toward the well from the solutionally enhanced flow systems.

Use of drill water losses would primarily be useful for the location of fluid recovery wells for remediation to indicate a tie into the actual permeable subcutaneous system. For probe borings where drilling with water would not interfere with other goals of the drilling, this is low cost data to obtain and could also be used in probe drilling for an actual recovery well site.

Comments on remediation

Remediation of sites such as described would consist of source removal from soils and recovery of water containing dissolved contaminants. Source removal in the soils would proceed conventionally and will not be further addressed. Suggestions are made relative to fluid recovery, however, with respect to siting recovery wells, improving their connections to the flow system, and using a vacuum assist to provide an increased effective drawdown to better connect to the subcutaneous system and "reach deeper into the system". The following ideas are conceptual and have been proposed but not yet tried in practice.

Siting of the recovery well should be as close to the main drain as possible consistent with property ownership and other site constraints. This will put the well in the best position to intercept flow from the site toward the main drain. It should also be placed where the top of rock is locally deep to allow for a greater cone of depression. If water loss data was obtained by rotary wash boring or coring, a location where higher permeability was observed should be strongly favored over other considerations. It may even be desirable to do special probing in an area of pinnacled rock where the recovery well is to be sited to select the best location.

Head loss in communicating between subcutaneous system and the recovery well could greatly limit the effectiveness of the well. If the well is in soil, this connection may be enhanced by jetting a cavity into the soil at top of rock using a high pressure jet of water. This would then be backfilled with sand to stabilize the hole. In rock, the bedding planes might be separated to communicate to the enhanced routes in the rock by hydrofrac at fairly low pressure.

The primary suggestion made for recovery is to used vacuum assist to increase the effective cone of depression of the recovery well. A major limitation on recovery is that drawdown of a well is limited by the depth of enhanced permeability. Regardless of the depth of the well, the normal cone of depression cannot be drawn down deeper than the level at which the water is entering the well from the enhanced joints or bedding planes. This means the cone of depression cannot reach "downhill" in the subcutaneous system toward the master drain and if head losses are experienced due to poor communication to the enhanced flow routes, even interception of local flow may not be possible.

Application of a vacuum to a well results in an increase in the head available to water to enter the well. This has been used for decades by the dewatering industry to dewater fine-grained soils. An effective drawdown of more then 20 feet below the bottom of the well is often practical if the well is constructed with a vacuum in mind. If a dual pump system is used where water is pumped with a downhole pump while the vacuum is maintained from the surface, there should also be no water lift limitation on the drawdown obtained.

Figure 7 shows the basic concept. A vacuum assisted recovery well is shown that is near but not in direct communication to a subcutaneous enhanced pathway. Part of the effective drawdown

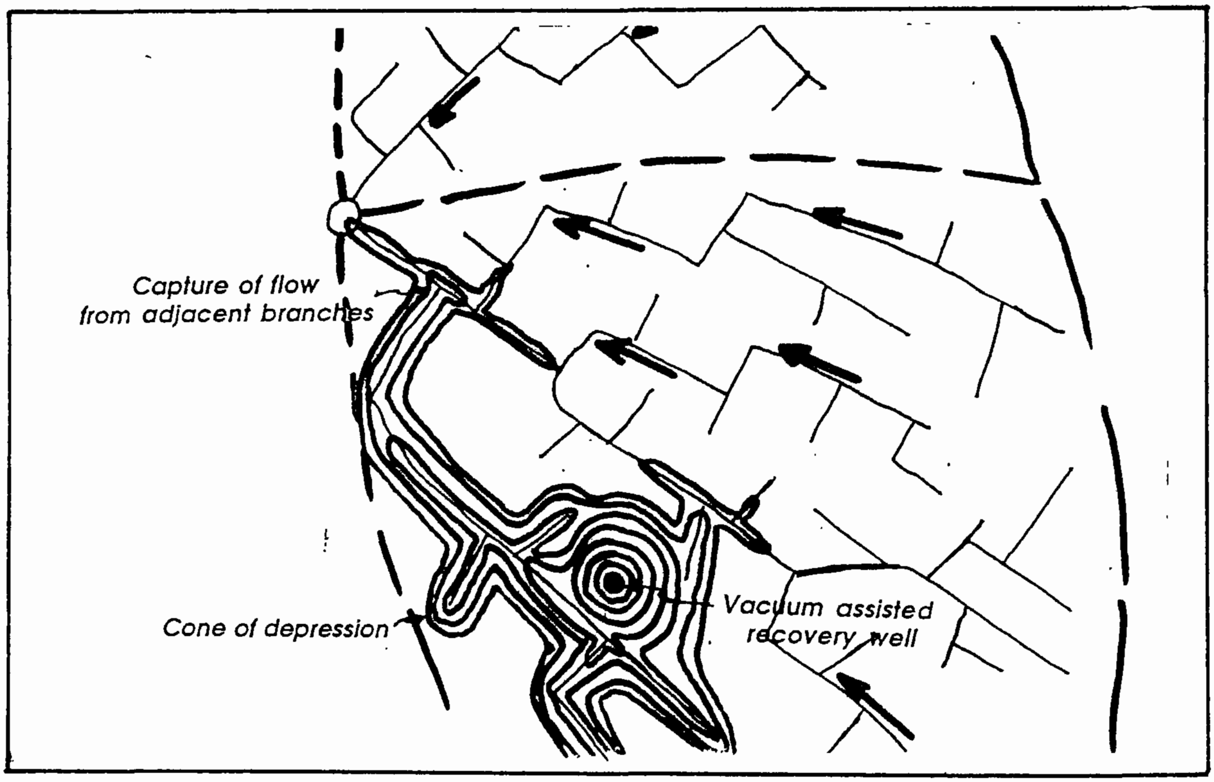


Figure 7 Vacuum assisted recovery well

is lost in the low permeability connection to this flow path. Once the enhanced flow path is reached, however, the cone of depression would expand preferentially along it. If the cone of depression is deep enough, this extension along the subcutaneous enhanced flow routes will reach toward the main drain and reverse the gradient of the pathway below the junction point of other branches. Thus, this vacuum assisted well can gather the flow from parts of the subcutaneous pathways outside its immediate vicinity and can stop further discharge of water from the site.

Conclusions

Movement of water and hydrocarbons through karsts with a subcutaneous zone will follow macropores in the soils and solutionally enhanced channels in the rock or top of rock toward master drains into an underlying conduit system. When a water table is present in the soils, this produces a cone of depression in the soils. The subcutaneous network is a tributary network that is integrated toward the master drain, but is probably not integrated laterally. Separate phase hydrocarbons follow the macropores preferentially and may be able to enter subcutaneous systems due to reduced capillary resistance and the mounding effect of the plume.

Assessment in karst areas should focus on determining whether the site is underdrained, where the intakes to the underdrain system are, if present, and how to monitor. Close piezometer networks and the geologic and topographic context of the site often provide the best clues, but use of an existing hydrocarbon plume as a tracer and top of rock configuration are also valuable. Pumping poorly producing monitor wells at least 10 to 20 gallons may improve detectability in the subcutaneous zone. Fluid recovery for remediation can be improved by location of the recovery well as close as practical to the drain intakes defined above, improving the connection to the subcutaneous system by hydrofracing or jetting, and using a vacuum assist to improve the effective drawdown of the well. The improved effective drawdown is intended to reach deeper into the system to intercept flow of adjacent branches of the subcutaneous system that connect to the drain down-flow of the recovery well.

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Biographical sketch

Tony L. Cooley has a B.S. in Geological Engineering from Washington University in St. Louis and has done two years each of graduate work in engineering geology/soil mechanics at Cornell University and rock mechanics at The Pennsylvania State University. He has worked as a consultant for 13 years as a geological/geotechnical engineer, the last four years of which were mostly assessments of hydrocarbon releases. He is a registered engineer in Alabama and a registered geologist in Georgia and Tennessee. At present he is a PhD graduate student in hydrogeology at the University of Kentucky in Lexington.

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Approaches to Hydrogeologic Assessment and Remediation of Hydrocarbon Contamination in Clay-Covered Karsts with Shallow Water Tables

Tony Cooley

1. Please comment on methods of detection of hydrocarbon contamination, especially with respect to economics.

The basic analyses used were BTEX (EPA Method 8020 or 602) and Total Petroleum Hydrocarbons (TPH) by either EPA Method 418.1 or California Department of Health Services (CAL-DHS) modification of EPA Method 8015. The CAL-DHS method is the more expensive of the two TPH methods but has fewer false positives. In clays, some of the non-polar naturally occurring hydrocarbons sometimes get polarized and get through the silica gel cleanup and show on the IR detector of the 418.1 method. For specific cases, a fuel identification using the GC portion of the CAL-DHS analysis and a sample of the free product may be done, but this is not common.

Economy is achieved by limiting the number of analyses run. Particularly for light fuels, all soil samples are screened by soil sample headspace analysis in the field with a PID or FID and profiles are plotted. This information, together with stratigraphic and water table information, are used in selecting the samples for analysis. Boring locations and number are also carefully selected and drilled in a certain order so that the results of each boring are used for field decisions to influence the implementation of the work plan. This results in the best economy. As noted in the paper, once the soils and ground water are sufficiently delineated, further borings for piezometers are logged by cuttings only but not sampled.

APPLICATION OF DYE-TRACING TECHNIQUES FOR CHARACTERIZING GROUNDWATER FLOW REGIMES AT THE FORT HARTFORD MINE SUPERFUND SITE OLATON, OHIO CO., KENTUCKY

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This abstract is being submitted for the Hydrogeology Session of the conference.

This paper discusses the application of dye-tracing techniques for determining the groundwater flow paths and characterizing flow regimes above a 120 acre underground limestone mine. The inactive mine is being used to store greater than 1.7 million tons of aluminum manufacturing by-product that, when wetted, emit toxic and noxious gases. Substantial quantities of water are entering the mine and levels of ammonia gas exceeding Immediate Danger to Life and Health (IDLH) concentrations, require that all personnel entering the mine use supplied air systems. This investigation is part of a Remedial Investigation/Feasibility Study (RI/FS) being conducted at the 700 acre site. One objective of this investigation is to identify the potential and/or actual groundwater flow paths into the mine. Greater than 25 sinkholes, and 31 springs were discovered onsite through aerial surveying and field reconnaissance. Dye receptor packets were located in each spring and at all water intrusion points within the mine. The results of 15 traces suggest that there are two aquifers located above the mine, which have the potential to "leak" into the mine: a turbulent flow limestone aquifer, which is perched above and serves as recharge for, a laminar flow shale/sandstone aquifer. This evaluation is substantiated by residual dye detected in the largest spring up to 109 days after dye injection. Sinkholes found to contribute water to the mine are located in areas where the faulting has displaced the low permeability shale/sandstone unit. The remaining sinkholes transmit water laterally along the limestone shale/sandstone contact located above the mine roof and discharge into surface streams. A large construction project was performed following the dye trace to plug sinkholes and to divert surface runoff away from the mine.

Application of Dye-Tracing Techniques
for Characterizing Groundwater Flow Regimes
at the Ft. Hartford Mine Superfund Site
Olaton, Ohio County, Kentucky

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INTRODUCTION

The Ft. Hartford Mine Site is an underground limestone mine located in rural western Kentucky. Since 1981 by-products of secondary aluminum recovery, called *salt cake fines* have been stored within the interior of the mine for potential aluminum and chlorides recovery by Barmet Aluminum Corporation of Akron, Ohio. Salt cake fines are a light gray powdery substance regulated as a solid waste. In a dry state the material is inert, however, when the fines come in contact with water several gases are generated: acetylene, methane, hydrogen sulfide, and ammonia.

Investigation began at the site due to environmental concerns that ammonia gas, chlorides, and possible metals were posing a threat to human health and the environment via numerous pathways from the site. In June 1988, the U.S. Environmental Protection Agency (USEPA) proposed that the Ft. Hartford Mine be added to the Proposed National Priorities List (NPL) or "Superfund" list for investigation and remediation under the Comprehensive Environmental Response, compensation, and Liability Act (CERCLA). Ft. Hartford was placed on the final NPL on August 30, 1990.

On September 20, 1989 Barmet Aluminum signed an Administrative Order by Consent (AOC) with USEPA Region IV to perform Expedited Response Actions (ERAs) and a Remedial Investigation/Feasibility Study (RI/FS) for the Ft. Hartford Mine Site. Two of the primary objectives of the ERAs were to: 1) identify the areas where water was entering the mine, and 2) to prevent the water intrusion and/or relocate salt cake fines away from the wet areas.

When ERAP work began in March 1990 there were five apparent locations where water intrusion was occurring into the mine. These five areas were collapse locations which occurred during limestone mining in areas of limited roof cover. These collapses allowed direct intrusion of surface water into the mine. The mine had accumulated approximately 37 million gallons of impounded water since mining began in the late 1950s. Additionally an aerial survey had located numerous "closed depressions", commonly referred to as sinkholes above the mine. All the sinkholes were scheduled for engineered closure during the ERAs.

Since the sinkholes would be closed prior to initiation of the RI/FS, the karst investigation was conducted concurrent with the ERAs to avoid losing data pertinent to the characterizations of groundwater flow regime(s) at the site. In addition, a determination of which sinkholes, if any, and the amount of water that they were contributing to the mine could be made.

PHYSICAL SETTING

The Ft. Hartford Mine Site is located in the east-central perimeter of the Western Kentucky Coal Field physiographic province. The Coal Field is located within the Interior Low Plateaus province of the Interior Plains Region as shown in Figure 1.

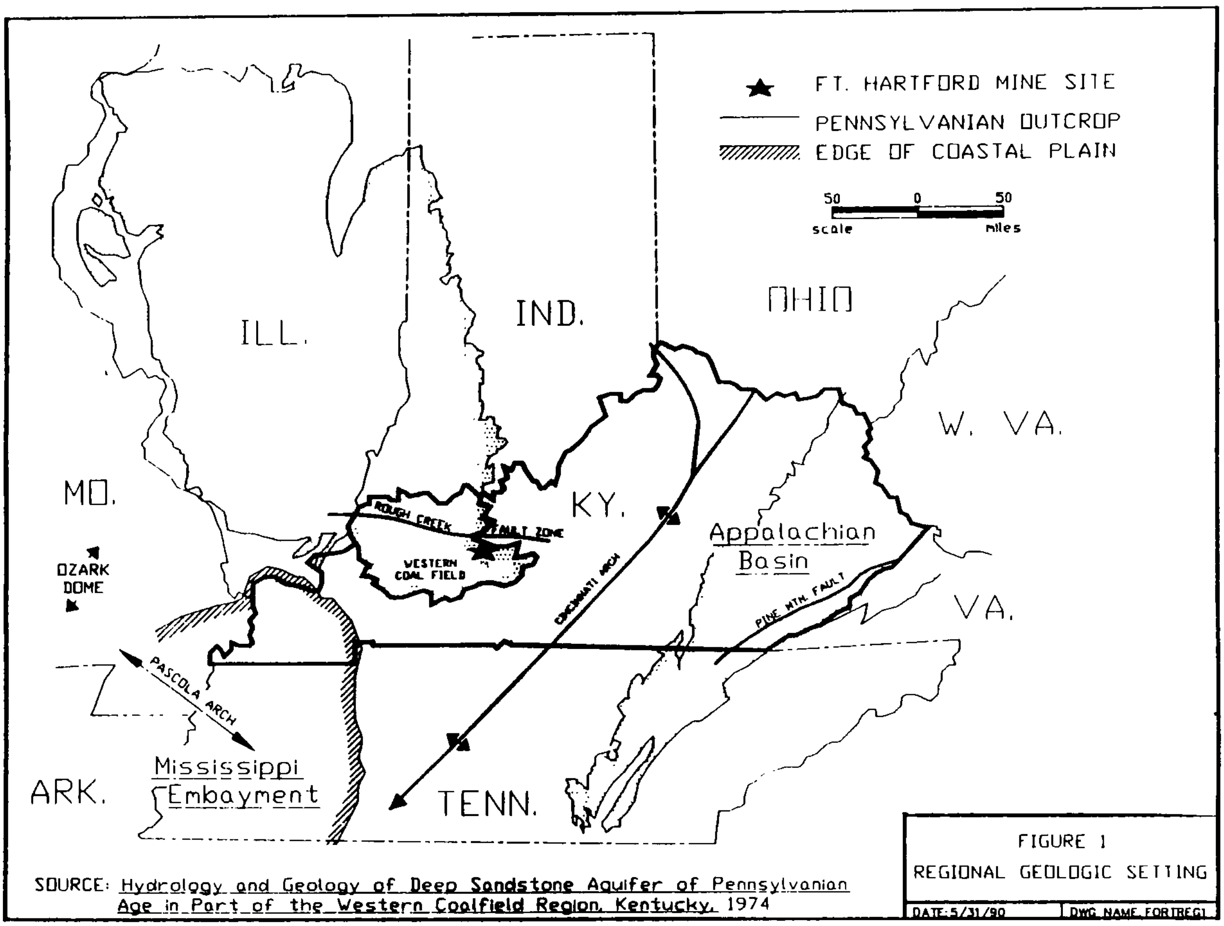

FIGURE 1
REGIONAL GEOLOGIC SETTING

Geologically, the area is in the southeastern section of the Eastern Interior (or Illinois) Basin. This region is characterized by low rolling hills formed of Pennsylvanian age shales, siltstones, and sandstones with occurrences of coal and limestones. However, the Ft. Hartford Mine Site is located within the Rough Creek Fault Zone and Mississippian age sandstones and limestones are exposed as a result of normal and high angles reverse faulting.

The alluviated valleys comprise a small portion of the area and have a general elevation of 380 to 420 feet (msl); the hills surrounding the site rise to a maximum elevation of about 625 feet. Surface topography in the Ft. Hartford Mine area is primarily rolling hills. Slope grades range from 5 to 20 percent. The highest and lowest points on the site form approximately 250 feet of relief.

The majority of the area above the mine is also heavily vegetated. Two large valleys traverse the site and trend northeast towards Caney Creek and Rough River. The most prominent surface features found on the property are the numerous karstic features present. Generally, these karstic features are identified by their surface expression as a closed topographic depression. The "sinkholes" as they are known, have the potential of transmitting large quantities of surface water into the underlying limestone aquifers. At the Ft. Hartford Site the sinkholes appeared to be transmitting surface water downward into the formations which overly the mine roof. Three surface water streams, Caney Creek to the east, Rough River to the north, and Cane Run to the west provide the primary drainage for the Ft. Hartford Site.

REGIONAL GEOLOGY

Mississippian age sedimentary rocks are exposed at the Ft. Hartford Site and throughout western Kentucky along the Rough Creek Fault Zone, a major faulted anticlinal feature cutting through the Western Kentucky Coal Field (Figure 1 and Plate 2). The Rough Creek Fault Zone represents one segment of a major lineament that stretches from the Appalachian region, across Kentucky, and into the Fluorspar Districts of southern Illinois, Kentucky, and Missouri.

The Rough Creek Fault System is a segment of the 38th parallel lineament, the designation given to a major basement fault system that stretches from the Appalachians to the Ozarks and possibly to the Rocky Mountains. The New Madrid Fault System extends from Mark Tree, Arkansas to Wabash, Indiana and crosses and connects with the 38th parallel fault system. At the confluence of these two fault systems is the Hicks Dome or anticline. The Hicks Dome has been described as a cryptovolcanic structure composed of mafic igneous intrusive rocks. These igneous intrusions are suggestive of incipient rifting along these major lineaments (Hook, 1970). The fluorspar districts of Illinois, Missouri, and Kentucky are developed within these intrusive rocks and their associated mineral assemblages. The other major faults that stretch to the east and west from the fluorspar district are the Rough Creek and Shawnee Town fault segments of the 38th parallel system, the St. Genevieve and the Cottage Grove Fault Systems.

The Rough Creek Fault System is a positive east-west trending overturned anticlinal structural complex where uplift has shifted Mississippian sedimentary rocks in the Ft. Hartford Mine area up into a primarily Pennsylvanian sedimentary rock terrain. This system is a compressional fold and is usually faulted as a single high-angle overthrust from the south that develops into a complex pattern of block faults that exhibit thrusting, dip-slip, and strike-slip movement (Rehn, 1968). The displacement along this fault system is divided along a series of step faults and drag faults. These step faults are predominantly gravity faults that progress in a step-like manner from horsts to grabens. One normal step fault segment crosses the Ft. Hartford Mine Site and the fault cut is plainly evident in an outcrop seen near the entrance to the Rough River side of the mine. These step faults usually dip between 60-90 degrees, but associated reversals of these faults typically produce dips within 5 degrees of vertical. Grabens caught between the major step fault systems can develop tension and shear fractures resulting in the development of additional step faults, antithetic faults and cross faults in the graben blocks. Antithetic faults are generally parallel to the strike of the fault system, have dips between 45-65 degrees and terminate at the juncture of a major step fault. Alteration along the actual fault zones usually consists of simple silicification in sandstones and only slight alteration in limestones and dolomites. Some strike-slip movement has been noted along the rough Creek Fault System due to torsional forces with the north wall of the fault moving to the west (Heyl and Brock, 1960). This movement is generally limited to the western portions of this fault systems and is in direct response to the wrenching movement related to the emplacement of the Hicks Dome mafic anticlinal feature in the Kentucky-Illinois Flurospar Region (Hook, 1972; Trace, 1970).

RESEARCH PROCEDURE

To trace the groundwater flow patterns in the vicinity of the mine and to determine which sinkholes, if any, were contributing to the impounded water in each lobe of the mine (Caney Creek and Rough River), the following research procedure was recommended by Dr. Nick C. Crawford of the Western Kentucky Center for Cave and Karst Studies and implemented at the site. This procedure is consistent with that in the USEPA report by Mull, Liebermann, Smoot, and Woosley (1988).

The first step was to conduct a comprehensive field reconnaissance to inventory the karstic features within the area of investigation. The field reconnaissance involved locating (field mapping) all sinkholes, springs, caves, karst windows, sinking streams, lineaments, and areas inside the mine where significant water intrusion was occurring. After all karst features and areas of water intrusion were located, a passive dye receptor packet was placed at each location. the receptor packets used during the study were small packets of activated coconut charcoal contained within aluminum or fiberglass screen mesh and small bundles of surgical cotton. A total of 36 dye receptor packets were placed at springs and in streams above ground, 5 were placed inside the mine at water intrusion points on the Caney Creek Side, and 8 were placed at water intrusion points on the Rough River side for a total of 49 dye receptor locations. Each receptor locations was given a letter and a number that corresponded to the group who placed it and the order in which it was placed.

After these receptors had been in place approximately one week, six locations were selected to be analyzed in the laboratory for background fluorescence to create a baseline for comparison. Water samples were also collected from major springs and streams and analyzed for background fluorescence. Results showed that there were no problems with background. Each receptor location selected for background was replaced with a new packet. Plates 1 and 3 show all receptor locations aboveground and inside the mine.

Once the inventory of the areas was completed, the dye receptors were in place, and background fluorescence was analyzed, dye tracing was initiated. The four dyes selected for the study are recommended by USEPA guidance documents (Mull, Liebermann, Smoot, and Woosley, 1988) and are as follows:

a) Fluorescein, Color Index: Acid Yellow 73
b) Optical Brightener, Tinopal 5BM GX, Fabric Brightening Agent: 22
c) Rhodamine WT, Color Index: Acid Red 388
d) Direct Yellow 96, Diphyenyl Brilliant Flavine 7GFF

These are the standard dyes most often used for dye tracing in karst aquifers. They are safe for this purpose in the concentrations used both for human consumption and aquatic life (Smart and Laidlaw, 1977 and Smart, 1986).

Since the dye traces were conducted during the summer months (dry, low flow conditions), all sinkholes were "flushed" with potable water in lieu of waiting for them to be flushed naturally by rainfall. A minimum of 500 gallons of potable water was injected into the openings to ensure that they drained sufficiently and to wet the soil to minimize dye sorption onto clays. The dye was then injected and flushed with a minimum of 1,000 gallons of additional water. Dye tracing was performed on a maximum of four sinkholes per test before the receptors were collected, analyzed, and replaced with new packets for the following traces.

Following dye injection and "flushing" with water, the receptors were left in place a minimum of one week. By allowing the dye to flush through the groundwater system prior to the next trace, the dye could be reused on subsequent sinks without confusing the results from previous traces. After the minimum waiting period, all the receptor packets aboveground and inside the mine were collected for analysis.

The charcoal packets were washed and elutriated with a solution of ninety-five percent isopropyl alcohol and a five percent potassium hydroxide to bring the Fluorescein dye to the surface of the charcoal. The packets were then visually observed for the presence of dyes; if there was a question concerning the interpretation of a charcoal dye receptor, the elutriant was compared with the elutriant of the background receptors on a Turner Fluorometer in the Hydrology Research Laboratory at Western Kentucky University. Rhodamine WT was usually visible on the charcoal because of the relatively large quantities used. However, the elutriant was always analyzed on the Turner Fluorometer for all Rhodamine traces and compared with background. Rhodamine WT, if present, was usually visible when treated with the above elutriant but the "Smart" elutriant consisting of a 5:2:3 mixture of 1-propanol, concentrated NH4OH and distilled water was used to analyze for Rhodamine WT as needed (Smart and Laidlaw, 1977).

The surgical cotton dye receptors were washed to remove as much dirt and debris as possible and then tested for Optical Brightener and Direct Yellow under a long-wave ultraviolet lamp. Positive traces for Optical Brightener appear as a blue-white glow under the lamp and Direct Yellow, appears as a pale yellow glow, when it is present.

DYE TRACING RESULTS

Dye tracing began with Sinkhole S10 on April 30, 1990 and continued until September 30, 1990 with the final dye traces at Sinkholes S9, S3, and S2. Extremely dry weather and residual dye problems account for the long duration of the karst hydrology investigation. Table 1 summarizes trace results with the sinkhole, type of dye used, date injected, and the dye receptor locations that were positive for each trace. Plates 1 and 3 show the dye traces (straight-line) from sinkhole injection point to spring or water intrusion into the mine.

The three most important springs in the vicinity of the Ft. Hartford Site are I1, E2, and E3. Spring E3, located west of Collapse locations 4 and 5, is flowing from a small cave at the based of the Glen Dean Limestone (Mgd) (Plates 1,2,3, and 4). It is perched directly upon the Hardinsburg Sandstone Formation (Mh) which is primarily shale in the vicinity of the Ft. Hartford Mine. The surface stream which flows from Spring E3 has a very steep gradient as it breaches the Hardinsburg perching layer which is about twenty feet thick at this location. As the stream flows off of the Hardinsburg (Mh) it sinks into the Haney Limestone (Mgh) and then flows to Spring E2 for a final resurgence before flowing into Cane Run Creek. (All dye traces which were positive for Spring E3 were also positive for Spring E2. However, a dye trace from E3 to E2 has not been made to confirm this assumption). During high discharge when the cave stream cannot accept the entire flow of the stream from E3, some of the water flows on the surface to Cane Run. Spring E3 is located along a pair of closely-spaced, east-west faults and the geologic structure is extremely complicated in this area (Plates 2 and 4).

TABLE 1
SUMMARY OF DYE TRACE RESULTS

Dye Trace Number	Sinkhole	Dye	Date Injected	Positive Dye Receptors
1	S10	1 lb. Fluorescein	4-30-90	-- (Not detected because Spring 11 was not found until 6-11-90)
2	S11B	2 liters Rhodamine	5-1-90	Spring K1 above Breakthrough 2
3	S1	5 lbs. O.B.	5-2-90	-- (Not detected because Spring I1 was not found until 6-11-90)
4	S11A	4 lbs. D.Y.	5-2-90	Spring G1 and into Cane Run
5	S1	4 lbs. O.B.	5-24-90	-- (Not detected because Spring I1 was not found until 6-11-90)
6	S8B	1 gal. Rhodamine	5-23-90	Springs E3, E2, and into Cane Run
7	S8A	2 lbs. Fluorescein	5-24-90	Springs E3, E2, and into Cane Run
8	S1	2 lbs. Fluorescein	6-11-90	Spring 11 and into Rough River
9	S6	5 lbs. O.B.	6-12-90	Inconclusive trace
10	S5B	1 gal. Rhodamine	6-13-90	Spring I1 and into Rough River
11	S12	4 lbs. D.Y.	6-13-90	Mine Intrusion RR2 and Springs E3, E2 and into Cane Run
12	S7	5 lbs. O. B.	8-10-90	Mine Intrusion RR1 and Spring E3, E2 and into Cane Run
13	S2	4 lbs. D. Y.	8-13-90	--Not detected (probably still in soil)
14	S4	Flush Pulse	8-20-90	Mine Intrusion CC-R16
15	S5	1/2 gal. Rhodamine	8-17-90	Mine Intrusion RR1 and RR2
16	S2B	4 lbs. D. Y.	9-18-90	Spring C1
17	S9	4 lbs. O. B.	9-20-90	Mine Intrusion CC-R22
18	S3	2 lbs. Fluorescein	9-20-90	--Not detected (may have flowed to an intrusion in the flooded sections of the mine).

Dye traces from Sinkholes S8B, S8A, S12, and S7 were all positive at Springs E3 and E2. All of these sinkholes are located on or near the east-west trending faults. The dye traces from Sinkholes S12 and S7 were also positive at water intrusions into the Rough Creek lobe RR2 and RR1 respectively, indicated that the Hardinsburg confining layer leaks in the vicinity of these faults allowing water to sink into the mine. Dye traces of Sinkholes S4 and S9, both located along or near these faults, flowed directly into the Caney Creek lobe at water intrusion locations CC-R16 and CC-R22 respectively and did not flow to Springs E3 and E2. The dye trace of Sinkhole S5 also located along or near these faults, flowed directly into the Rough Creek lobe at water intrusions RR1 and RR2 (Table 1, Plates 1 and 3).

Spring I1 is located along the Rough River north of the Rough River mine entrance. It flows out of a massive pile of rocks displaced by the mining operation and cascades about three feet into the Rough River. It appears that it probably resurges from a cave at the based of the Glen Dean Limestone (Mgd), perched upon the Hardinsburg Sandstone (Mh) confining layer, which has been completely buried by mine debris. The small valley north of the Rough River mine entrance has been filled with rock from the mine entrance to the river. The spring may have originally been located in this valley along the Glen Dean (Mgd) Hardinsburg (Mh) contact.

Spring I1 was not found during the initial field reconnaissance due to high flow conditions in the Rough River. Consequently, attempts to dye trace Sinkhole 1 on 5/2/90 and 5/24/90 and Sinkhole 10 on 4/30/90 were unsuccessful. As summer approached, the river level dropped, and the spring revealed on 6/11/90. Sinkholes S1, S10, and S5B were all traced to Spring I1. Sinkhole S6 dye traced on 6/12/90 may also flow to Spring I1. However, there is a chance that the positive cotton dye receptor at I1 may have been due to residual Optical Brightener still resurging at the spring from the Sinkhole 1 traces of 5/2 and 5/24/90. After performing traces of sinkholes near S6 which all went into the mine it seems reasonable that the drainage from S6 goes there also. However, Optical Brightener was not detected on any dye receptors in the mine or on the surface other than the one at Spring I1, but since there is a chance that it may have been residual dye form the Sinkhole S1 traces, the results of the Sinkhole S6 trace were considered inconclusive.

Plates 1 and 3 show the dye tracing results at the Ft. Hartford Site. Plate 1 shows the approximate groundwater basins identified by the dye traces. Usually, static water levels are obtained from water wells and monitoring wells which permit equipotential lines to be drawn for the uppermost aquifer. This information confirms dye tracing results and permits drainage basin divides to be delineated. Unfortunately there was only one water well in the area and monitoring wells have not yet been installed. The equipotential lines also provide important information, such as troughs in the water table, which often indicate the approximate routes taken by cave streams from dye injection points to springs. Without the water table data, only straight lines were used to indicate the flow from injection points to discharge locations. With water table data this lines could have been curved to show the approximate flow routes of cave streams. The highs in the water table, indicative of drainage basin divides, would have permitted more accurate delineations of the groundwater basins. Plate 1 shows the approximate drainage basins based on the dye traces, topographic divides, and geology.

CONCLUSIONS

The dye tracing investigation revealed that water sinking into some of the sinkholes in the Glen Dean Limestone (Mgd) above the mine flows along solutionally enlarged conduits that are perched upon shale layers in the underlying Hardinsburg Sandstone (Mh). In most areas the Hardinsburg (Mh) confining layer appears to prevent water in the Glen Dean (Mgd) from sinking into the mine located in the Haney Limestone (Mgh). However, sinkholes located on or near the east-west faults which cross the southern portion of the mine do permit water from nearby sinkholes to flow through the Hardinsburg (Mh) into the mine (Plates 3 and 4).

The following dye traces of sinkholes in the Glen Dean Limestone (Mgd) located above the mine revealed that water sinking into these sinkholes flows to springs and does not flow into the mine (Plates 3 and 4).

SINKHOLES	DISCHARGE SPRINGS
S1	Spring I1 and into Rough River
S10	Spring I1 and into Rough River
S5B	Spring I1 and into Rough River
S8A	Spring E3/E2 and into Cane Run
S8B	Spring E3/E2 and into Cane Run
S11	Spring K1 and into Collapse 2
S11A	Spring G1 and into Collapse 5
S2B	Spring C1 and into a tributary of Caney Creek

The following dye traces in the Glen Dean Limestone (Mgd) located above the mine revealed that water sinking into these sinkholes flows through the Hardinsburg (Mh) confining layer and into the mine. All of these sinkholes are on or near the two parallel, east-west trending faults which cross the southern portion of the mine and it appears that water draining from these sinkholes flows through the Hardinsburg (Mh) and into the Haney Limestone (Mgh) along these faults (Plates 3 and 4).

SINKHOLES	DISCHARGE SPRINGS
S4	CC-R16 (Caney Creek lobe)
S5	RR1 & RR2 (Rough River lobe)
S9	CC-R22 (Caney Creek lobe)
S12	RR2 (Rough River lobe) and also to E3/E2 and into Cane Run Creek

S7	RR1	(Rough River lobe) and also to E3/E2 and into Cane Run Creek

Primarily due to faulting, the hydrogeology in the vicinity of the Ft. Hartford Mine is extremely complicated. However, the results of this investigation permit the following tentative conclusions about the hydrogeology.

The Tar Springs Sandstone (Mts) does not restrict water from percolating through it into the Glen Dean Limestone (Mgd). Therefore, there does not appear to be a water table in the Tar Springs (Mts). Sinkholes in the Tar Springs Sandstone (Mts) support this conclusion. Water percolating through the Tar Springs (Mts) has dissolved voids in the underlying Glen Dean Limestone (Mgd). The Tar Springs Sandstone (Mts) has slumped into these voids and the erosion of sand through the solutionally enlarged conduits in the Glen Dean (Mgd) to springs is the method by which these somewhat unusual sandstone sinkholes have been created (Plates 3 and 4).

The Glen Dean Limestone (Mgd) has a karst-conduit, turbulent flow aquifer perched upon the Hardinsburg Sandstone (Mh) which in this area contains numerous shale layers. The aquifer is recharged primarily from surface runoff flowing into sinkholes. It also receives recharge from water percolating downward through the Tar Springs Sandstone (Mts). Recharge water flows rapidly through solutionally enlarged conduits at the base of the Glen Dean (Mgd) to springs usually located at the Glen Dean (Mgd) - Hardinsburg Sandstone (Mh) contact (Plates 3 and 4).

The Hardinsburg Sandstone (Mh), primarily a shale in this area, is responsible for perching the karst aquifer in the Glen Dean Limestone (Mgd). In most areas it prevents the vertical movement of water from the Glen Dean (Mgd) karst aquifer from sinking into the Haney Limestone. The dye trace investigation supports this conclusion as does the scarcity of solutional features intersected by the mine. However, in some areas faulting permits water from the Glen Dean (Mgd) perched aquifer to leak through the Hardinsburg (Mh) directly into the Haney Limestone (Mgh). Dye traces of sinkholes on or near the pair of east-west faults which cross the southern portion of the mine support this conclusion.

The dye study also supports the conclusion that the Hardinsburg Sandstone (Mh) contains a porous-media, laminar-flow aquifer. The top of this aquifer is the water table in the Glen Dean Limestone (Mgd) and the bottom is perched upon shale layers in the Hardinsburg (Mh). It appears that very little water is leaking through the Hardinsburg (Mh) except along faults and in places where it is thin and weathered. All five mine collapses occurred in places where surface streams were flowing over thin and weathered Hardinsburg (Mh). Most of the water intrusions into the mine (except those along faults) occur along the periphery of the mine where mining was stopped because it was approaching a valley where the overlying Hardinsburg (Mh) is thin and weathered thus permitting water intrusion into the mine.

The residual dye problems, at Springs I1, E3, and E2, also indicate two types of aquifers. The rapid flow through from sinkholes to springs revealed a karst-conduit turbulent-flow aquifer at the bottom of the Glen Dean Limestone (Mgd). However, residual dye was still flowing from Spring I1 109 days following dye injection. This appears to indicate that the porous-media, laminar-flow aquifer in the Hardinsburg (Mh) is recharged from the conduit-flow aquifer in the Glen Dean (Mgd). Therefore, the dye flows rapidly through the conduit from the sinkhole to the spring but the dye in the Hardinsburg (Mh) aquifer is released slowly over a period of several months.

The Haney Limestone (Mgh), protected by the Hardinsburg (Mh) confining layer, receivers very little recharge from above except where there are faults or the Hardinsburg (Mh) is thin and weathered. Very little water is entering into the mine except in these areas.

There is a shale confining layer at the top of the Big Clifty Sandstone (Mgbc). Johnson and Smith (1968) in their description of the Big Clifty Sandstone (Mgbc) on the Olaton Geologic Quadrangle refer to a "clay shale common in the uppermost part." Their stratigraphic column for the Big Clifty Sandstone (Mgbc) indicates that this shale is about 8 to 15 feet thick. In other areas where Crawford and Dotson (1990) have worked, this shale confining layer often perches springs which flow from a karst aquifer in the Haney Limestone (Mgh). Several springs at the Ft. Hartford Mine Site are located at the Haney Limestone (Mgh)-Big Clifty Sandstone (Mgbc) contact. They are springs E2, A6, C5, D1, and C4. These springs are small and cease to flow during dry periods. This may be indicative of the relatively small amount of water which gets through the overlying Hardinsburg (Mh) confining layer into the Haney Limestone (Mgh).

The shale confining layer at the top of the Big Clifty (mgbc) may also be responsible for the lakes inside the mine. The lake elevations appear to correspond to the dip of the Big Clifty Sandstone (Mgbc) although probably not the true dip. From north to south the lake surface elevations on the map of the mine decrease from 482 to 463 to 449 to 421 feet (msl). The lowest lake surface elevation may correspond to the surface elevation of Caney Creek and therefore may represent the water table. The other lakes are probably perched upon the shales at the top of the Big Clifty (Mgbc).

Since the base-flow, surface elevation of the Rough River is approximately 395 feet and the lake closest to the river is 482 feet, it certainly appears that this lake and the 463 feet and 449 feet lakes are perched upon the shale confining layer at the top of the Big Clifty (Mgbc). If the true water table corresponds to the base flow elevation of the Rough River (and it may not because of the faulting in the area) then the water table in the Big Clifty formation under the mine is probably located at an elevation of 420-450 feet. In the back part of the mine (south) the Big Clifty aquifer may be confined by the same shale confining layer which perches the lakes in the Haney Limestone (Mgh) in the northern areas of the mine.

Figure 2 is a geologic cross section showing the hydrogeologic profile across the Rough River side of the mine. The cross section illustrates the complexity of the hydrogeology in the vicinity of the Ft. Hartford Mine Site. The diagram shows the following hydrogeologic features.

There is a perched, karst-conduit, turbulent-flow aquifer at the base of the Glen Dean Limestone (Mgd). This aquifer is perched upon shale layers in the underlying Hardinsburg Sandstone (Mh).

There is a perched, porous-media, laminar-flow aquifer in the upper Hardinsburg Sandstone (Mh). This aquifer is also perched upon the same shale layers in the Hardinsburg Sandstone (Mh). Therefore this perched aquifer consists of two parts: the basal part in the upper Hardinsburg Sandstone (Mh) (porous-media, laminar-flow) and an upper part in the lower Glen Dean Limestone (Mgd) (karst-conduit, turbulent-flow), both perched upon the shale layers in the Hardinsburg (Mh).

There are cave streams at the base of the Glen Dean Limestone (Mgd) flowing to springs along the Glen Dean-Hardinsburg contact. Some of the cave streams flow to vertical openings in the Hardinsburg confining layer along or near faults. These openings permit water from the upper Hardinsburg - lower Glen Dean perched aquifer to enter into the Haney Limestone (Mgh). All of the sinkhole dye traces which were positive at water intrusion points in the mine were located along or near this east-west fault indicated on the cross section.

Figure 3 is a cross section drawn along the course of the stream which flows into Sinkhole 1 and then to Spring I1 on the Rough River. The headwaters is an ephemeral stream which flows north upon the Tar Springs Sandstone (Mts). As it flows off of the Tar Springs and onto the Glen Dean Limestone (Mgd) it flow directly into Sinkhole 12. Sinkhole 12 was dye traced to Spring E3 and also to mine intrusion location RR2. The stream channel continues beyond Sinkhole 12 and it is obvious that at times it cannot take all the discharge and some of the water continues north on dow the channel toward Sinkhole 1. It appears that the headwaters of this stream are being pirated by the cave stream which flows to Spring E3 and also to mine intrusion location RR2 probably due to the east-west faults in the area. Water flowing past Sinkhole 12 flows onto the Hardinsburg Sandstone (Mh), joined by small seeps which occur at the Glen Dean-Hardinsburg contact.

The Hardinsburg is thin and weathered and there is an east-west fault which crosses the stream at this location. Collapse 1 has the faulting in the area) then the water table in the Big Clifty formation under the mine is probably located at an elevation of 420-450 feet. In the back part of the mine (south) the Big Clifty aquifer may be confined by the same shale confining layer which perches the lakes in the Haney Limestone (Mgh) in the northern areas of the mine.

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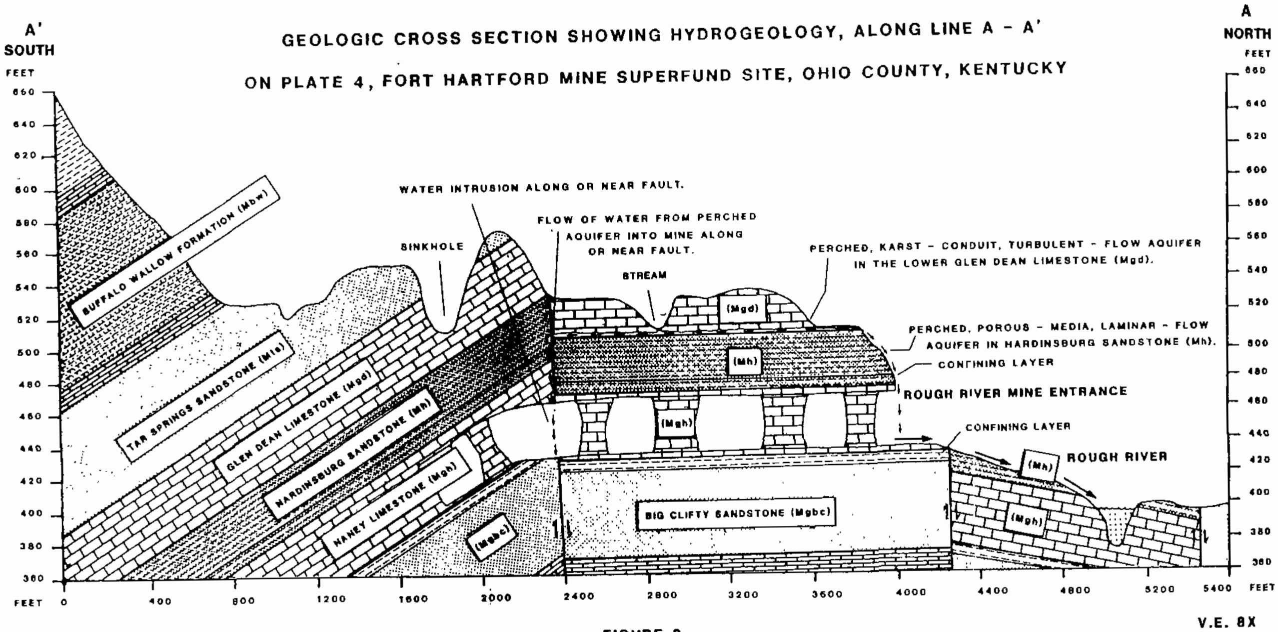

FIGURE 2

There is a perched, karst-conduit, turbulent-flow aquifer at the base of the Glen Dean Limestone (Mgd). This aquifer is perched upon shale layers in the underlying Hardinsburg Sandstone (Mh).

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The Hardinsburg is thin and weathered and there is an east-west fault which crosses the stream at this location. Collapse 1 has occurred directly under this stream. Collapse 2 has occurred directly under a tributary to this stream located about 200 feet to the west. Before these collapses occurred both streams flowed over the east-west fault onto the Glen Dean Limestone (Mgd) to a swallet at Sinkhole 1. The dye trace of Sinkhole 1 proved that this stream then flows through a cave, located at the base of the Glen Dean Limestone (Mgd), to Spring I1 on the Rough River.

Although the Ft. Hartford Mine Site is extremely complicated, primarily because of faulting, this investigation has revealed a basic understanding of the karst hydrogeology as it exists in the Tar Springs Sandstone (Mts), Glen Dean Limestone (Mgd), Hardinsburg Sandstone (Mh) and Haney Limestone (Mgh). Exploratory wells will be necessary to investigate the hydrogeology below the Haney Limestone (Mgh).

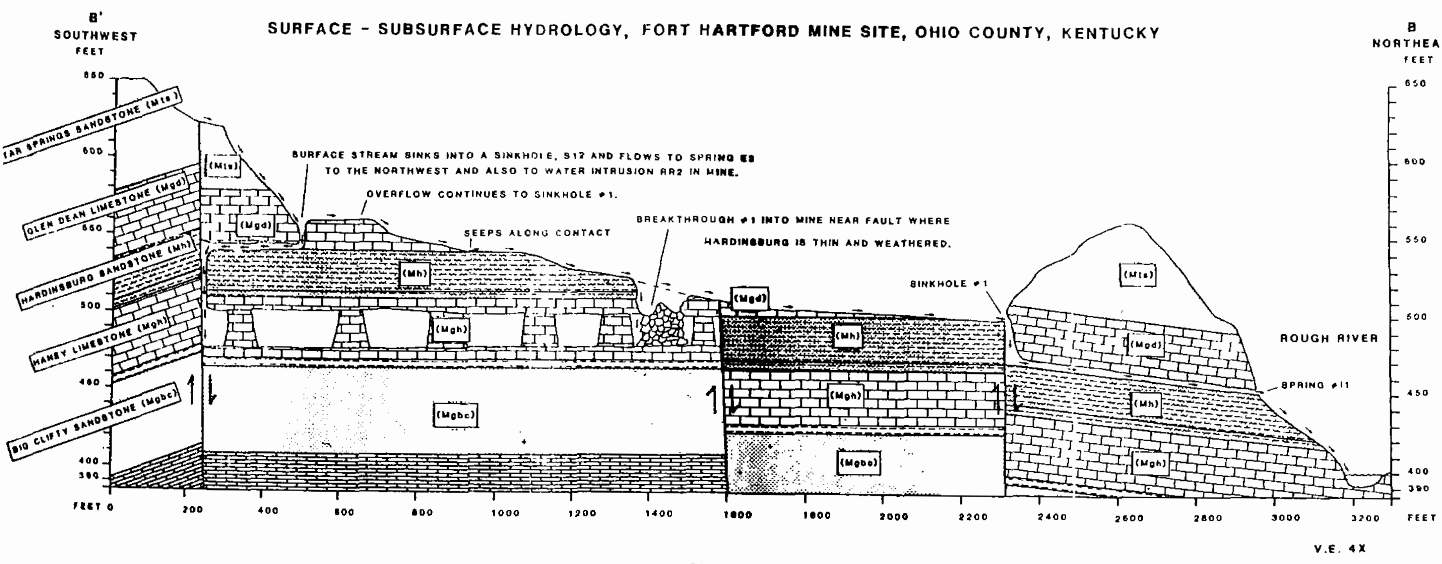

FIGURE 3

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NOTE: Copies of Plates 1-4 will be available upon request from the authors.

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Application of Dye-Tracing Techniques
for Characterizing Ground Water Flow Regimes
at the Ft. Hartford Mine Superfund Site
Olaton, Ohio County, Kentucky

Nicholas C. Crawford, Ph.D.
and
Ginny L. Gray

1. What reduction in water infiltration into the mine was accomplished by the $2 million sinkhole-filling and drainage diversion project you described?

As part of a National Pollution Discharge Elimination System (NPDES) permit with the state of Kentucky, the volume of water being pumped from the mine is currently being monitored for a four month period to determine the water infiltration reduction more quantitatively. Qualitatively, the reduction of water infiltration varies between the two lobes of the mine. Water infiltration into the Rough River lobe (which contained the massive collapses to the surface) has been dramatically reduced. When the KPDES permit was first obtained in November of 1990, there were approximately 15,000,000 gallons of impounded water in the Rough River side. This water has since been evacuated with only small amounts of recharge. The Caney Creek lobe did not contain collapses to the surface; however, several of the sinkholes aboveground were contributing water to the impoundments. This lobe initially had approximately 22,000,000 gallons of impounded water which has also been evacuated. Recharge in this lobe has been more substantial due to several partial roof collapses that continue to transport groundwater into the mine.

2. You stated the that the dye (Rhodamine WT?) was platykurtic (Broad) rather than the "normal curve" you expected for your main spring. You explained this as possibly due to diffuse recharge through the overlying sandstone aquifers. Might the platykurtic shape of the breakthrough curve be better explained by mixing and storage of the dye in the water mass impounded by the mine?

There were receptor packets placed within the impounded water and all other water intrusion points inside each lobe of the mine that were retrieved and analyzed to account for this possibility. Secondly, the traces which showed residual dye problems were in units above the mine roof which did not penetrate the Hardinsburg Sandstone (primarily shale above the mine) and enter the mine, but travelled laterally above the mine roof and discharged into the Rough River just north of the mine.

GROUND WATER MONITORING IN UNSATURATED AND SATURATED ZONES AT A SITE WITH PALEOCOLLAPSE STRUCTURE

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ABSTRACT

Paleocollapse structures can represent major obstacles to in site investigations for large engineered structures and underground facilities. Recognition of paleocollapse structures become a vital part of such site characterization.

Extensive underground mining of limestone has occurred in the greater Kansas City area. A landfill expansion was proposed over an abandoned limestone mine located about 200 feet below the surface. This case history deals with the development of a landfill over an abandoned underground mine, the impact of paleocollapse structures and a subsequent ground water monitoring strategy.

The site characterization work identified a paleocollapse structure, probably due to a deep seated cavity network probably within the Mississippian rock about 600 feet below the mine floor. This collapse had induced fractures in the overlying rock and soil. When the mine subsequently cut through the zone of paleocollapse structure, a major 14-acre area of mine-roof collapse occurred. The paleocollapse structure lead to the formation of a local synclinal basin with major fractures. These features have a major impact upon the mine stability and the development of a reliable ground water system for the site.

Ground water above the mine was found to be perched and isolated with flow through isolated joints, fractures and bedding planes. Ground water characterization must consider the perched and fracture flow ground water conditions, recharge of water into the mine, and the superimposed paleocollapse structural aspects in order to develop a ground water monitoring strategy for this unique site.

INTRODUCTION

There has been extensive mining of the Bethany Falls Limestone (Pennsylvanian Age) within the greater Kansas City area. Kansas City area mines are almost always dry and stable giving the city the distinction of being Number One in the world in terms of human use and occupancy of underground space. More than 200 businesses are located underground occupying more than 1,200 acres as of 1983 (Hasan, et al., 1988).

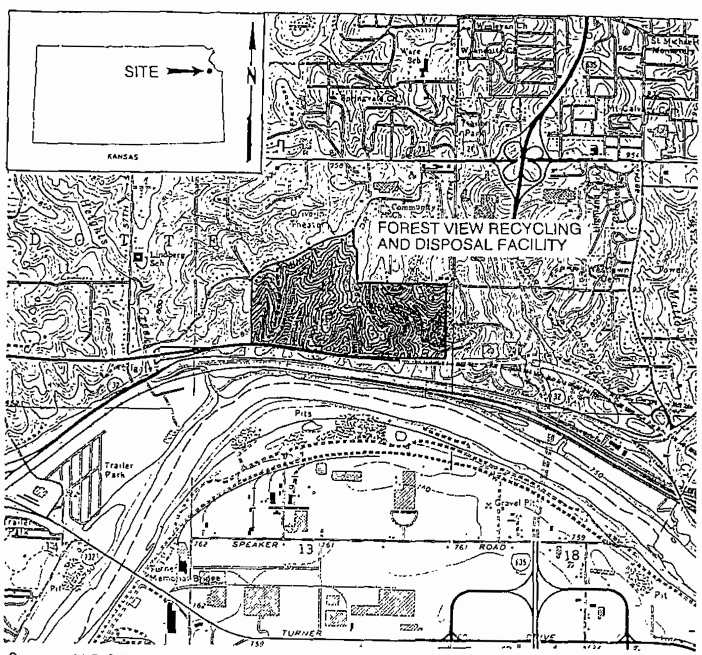
Due to other favorable (non-geologic) conditions, Waste Management of North America, Inc. chose in 1986 to expand its Forest View Landfill over an abandoned limestone mine underlying much of the site (Figure 1).

The loess soils over the mine contained major fissures which coincided with the area of mine-roof collapse. Preliminary site characterization by one consultant identified vee-shaped features in the loess soil blanket over the collapsed portion of the mine and suggested that the surface fissures were due to the mine-roof collapse (Figure 2). This was a reasonable conclusion, based upon limited information available at the time. Subsequently another consultant suggested that surface subsidence, and hence fissures, were not likely to occur based upon the general conditions within the mine but did not address the origin of the fissures.

There were two key site characterization issues to be resolved for this project:

- o First, characterization of the relation (if any) between mine-roof collapse and the surface fissures along with an assessment of mine stability;
- o Second, assessment of the hydrogeologic conditions so that a rational ground water monitoring strategy could be developed.

This paper provides a brief summary of the site characterization (from both the engineering geologic and hydrologic perspectives) found necessary to proceed with the landfill development over an abandoned mine which had undergone major roof collapse. The sequence of presentation within this paper follows that of the actual site characterization process.



Source: U.S.G.S. Shawnee, Kansas
Quadrangle, Photorevised, 1975.

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Figure 1. Site Location Map

GEOLOGIC SETTING

A thick deposit of Pleistocene loess overlays the bedrock. Bedrock is made up of an alternating sequence of cyclothems consisting of relatively thin beds of limestone and shale, which were deposited in migrating shallow seas (Figure 3). The Pennsylvanian System is underlain disconformably by Mississippian limestone and dolomite beds.

Regional bedrock dips gently (10 to 20 feet/mile) to the northwest. However, since the site is located on the northeastern limb of the Shawnee Syncline extension, the local dip is to the southwest. A local synclinal basin occurs over the collapse zone. In the early stage of investigation, it was not clear whether this local synclinal basin was due to paleocollapse or simply a local structural feature, which are commonly observed in the area with low amplitudes of about 3 to 10 feet.

Much of the Pennsylvanian strata observed in outcrops around the site, contain near-vertical joints. These joints are particularly evident in the limestones, where fractures of two dominant directions are seen, generally striking to the northeast and northwest directions.

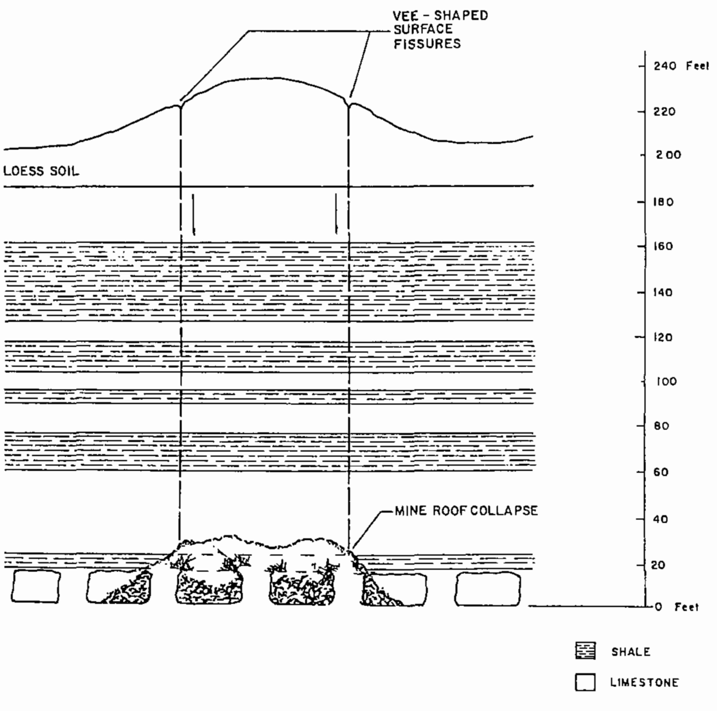


Figure 2. Previous (1986) Perception of Site Conditions: Mine Collapse has Lead to Surface Subsidence

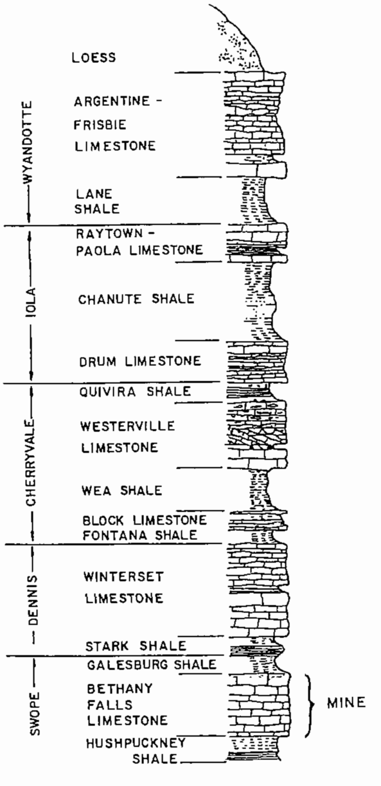


Figure 3. Geologic Section

CHARACTERIZATION OF MINE-ROOF COLLAPSE AND SURFACE FISSURES

Much of the data acquired to resolve the mine-roof collapse assessment and its relationship to the surface fissures was based upon geologic observations, from within the mine, from seven exploratory trenches cut into the loess soil, and at numerous rock surfaces and cuts exposed by trenching or by excavation during construction. An estimated 1,000 man-hours have been spent inside the mine, developing an initial map of the mine-roof collapse area, making engineering geologic observations, monitoring mine-roof collapse, and carrying out a QA audit of the mine backfill program. All observations within the mine and the surface were extensively documented by engineering geologic maps, cross-sections and color photographs.

Initial Findings Regarding Characterization of Mine-Roof Collapse and Surface Fissures

The following facts evolved as part of the initial mine-roof collapse characterization and its relationship to the surface fissures (Figure 4):

- o About 170 feet of limestone and shale lie over the mine;
- o The rock is covered by a thick loess soil up to 60 feet thick;
- o The fissures appear on the surface as large arc's making up circular patterns centered directly over the area of major mine-roof-collapse (Figure 5);
- o Observations in 7 trenches revealed that the fissures in the loess soil were much older (at least many hundreds to thousands of years) than the mine-roof collapse which occurred about 14 years before this investigation;

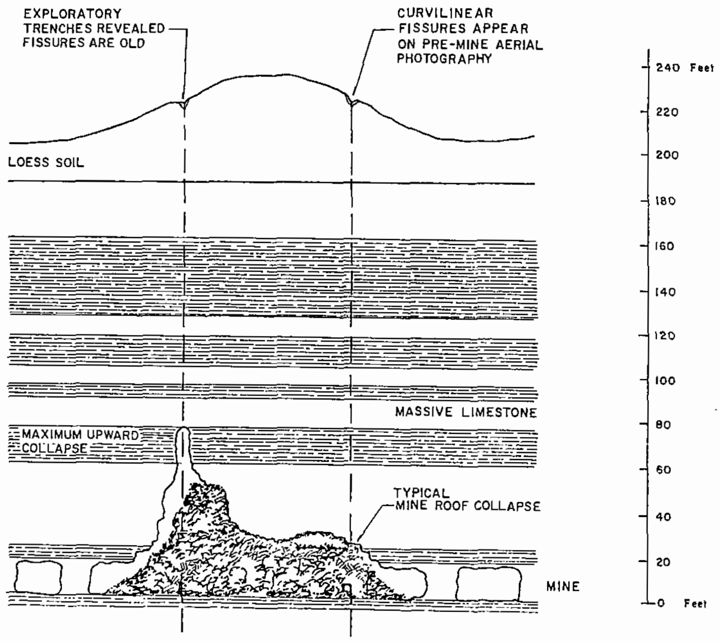


Figure 4. Initial Findings Regarding Mine-Roof Collapse

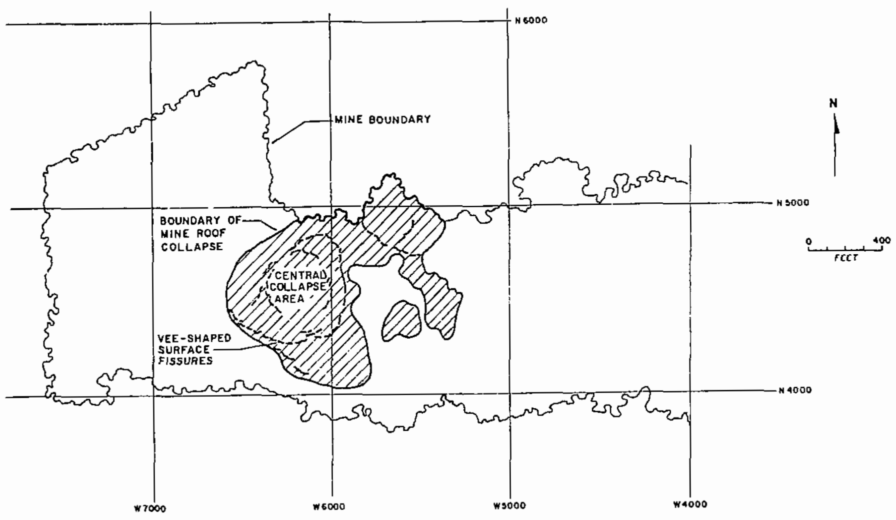


Figure 5. Spatial Relation of Surface Fissure and Mine-Roof Collapse

- Some of the surface fissures were observable on pre-mine aerial photography, indicating that the fissure existed before mining began;
- The mine is a room-and-pillar mine, fourteen feet high;
- A random pillar layout was used before the major roof-fall collapse occurred;
- A regular room-and-pillar layout was used after the major roof-fall collapse occurred;
- The regular pillars were approximately 25 feet x 25 feet and on 60-foot centers;
- The major collapse area (in which major upward collapse has taken place) covered an area of about 16 acres;
- The major mine-roof collapse occurred within the random pillar portion of the mine;
- Larger (80 acres) and much older portions of the mine with random pillars had very little mine-roof collapse and no major mine-roof collapse;
- Typical mine-roof collapse extends upward 10 to 15 feet through the Rubble Rock Zone, the Galesburg Shale, and the Stark Shale to the base of the Winterset Limestone;
- A limited number of local mine-roof collapse extends upward as much as 60 feet above the mine-roof to the base of the Westerville Limestone;
- Upward collapse was conical in shape becoming smaller upward (note that the angle of draw is inward contrary to the outward angle of draw commonly used in coal mine subsidence);
- The Westerville Limestone immediately above 60 feet was quite massive and thick (15 feet) and would tend to resist further upward collapse;
- Roof rock debris bulking factors of 20 to 75% were observed in the mine and averaged 42%;
- A bulking factor of 30% would choke off upward collapse within 80 feet from the mine-roof, about half-way through the 170 feet of rock overlying the mine;
- Permit conditions required backfilling of the mine. A backfill mixture of fly-ash, bottom ash and kiln dust was initially used to backfill the mine.

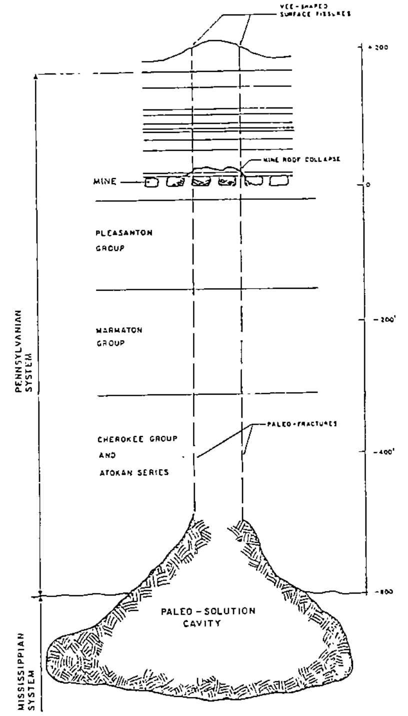
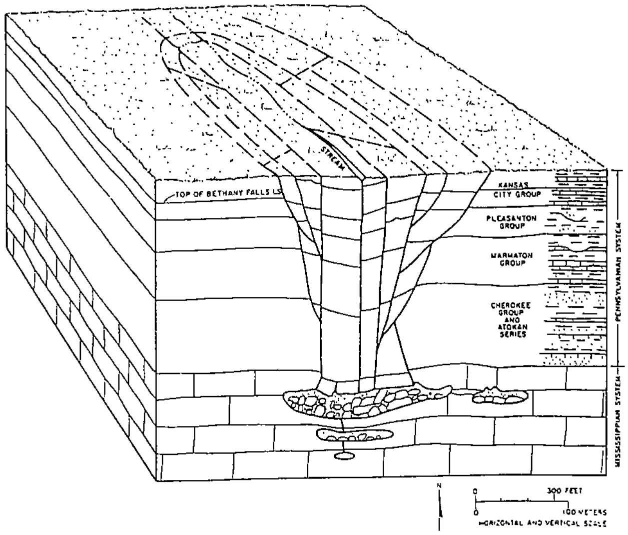


Figure 6. Conceptual Geologic Model to Account for Site Conditions



Source: Gentile, 1984.

Figure 7. Conceptual Model of Typical Paleocollapse Structure in Kansas City Area

Conceptual Model of Mine Collapse and its Relationship to Surface Fissures

The previous facts (the age of the fissures, and observations of fissures in pre-mining aerial photography) lead to the conclusion that the mine-roof collapse had not caused the surface fissures. A conceptual model which can account for these fissures incorporates the presence of a deep-seated paleocollapse structure deep below the mine (Figure 6). This model has been used to explain other paleocollapse structures (Figure 7) in the greater Kansas City area (Gentile, 1984). Subsequent investigations of rock exposures resulting from site development has strongly supported this conceptual model.

Additional Engineering Geologic Work Included:

- Developing a detail map of the mine collapse;
- Monitoring mine-roof collapse by quarterly inspections;
- Observations of surface rock outcrops exposed by construction;
- Stabilizing the mine by backfilling.

Developing A Detail Map Of The Mine Collapse

A detailed map of the mine-roof collapse area and the mine around it was developed so that the collapse areas could be monitored. This map has been revised periodically to reflect changes in mine collapse as well as to correct any errors in the map. The mine map also provides a means of safety in mine operations.

Monitoring Mine-roof Collapse By Quarterly Inspections

Monitoring of mine collapse consisted of numbering and photographing all of the collapse areas around the perimeter of the central collapse area. During each subsequent inspection, photos were taken for comparison. After five mine collapse inspections (1988-1991) mine-roof collapse slowly continues to expand laterally around the perimeters of the central collapse area (Figure 5).

Much of the mine-roof collapse is thought to be initiated by deterioration of a Rubble Rock which occurs at the top of the Bethany Falls Limestone. The Rubble Rock Zone is a slake-susceptible horizon found over the greater Kansas City area. The Rubble Rock Zone is known to deteriorate rapidly with moisture. Since 1987 mine flooding has increased substantially, leading to the degradation of the Rubble Rock Zone. Generally, mine-roof collapse extends upward (about 10 feet) through the Bethany Falls Limestone (which includes the Rubble Rock Zone), the Galesburg and Stark Shales to the base of the Winterset Limestone.

At a few locations, around the perimeter of the collapsed area, collapse continued to extend upward to as much as 60 feet above the mine-roof. At this point, the upward collapse encounters the massive Westerville Limestone which is 15 feet thick. Only three of these areas were inspected due to access and safety reasons. In each case, these upward extensions of mine-roof collapse were very localized (less than 15-30 feet in diameter) and became smaller as the collapse extended upward. The three upward extensions of mine-roof collapse that were observed were all located along a curvilinear line of the paleofracture and are thought to be linked to the paleo-structure.

Since the angle of draw is inward (contrary to the outward angle of draw commonly used in coal mine subsidence), and the bulking in each of the three areas inspected was close to choking off and stopping any further upward collapse. These observations support early assessments that the mine collapse would not lead to surface subsidence.

Observations At Rock Outcrops Exposed By Construction

Further evidence for paleocollapse activity is seen in rock surfaces and cuts exposed by construction at the site.

Observations included:

- o A cross-section through one end of the synclinal basin which lies over the central collapse area with an amplitude of about 15 feet;
- o Numerous fractures and shear zones along with sizeable open cavities associated with fractures; and
- o The fact that all of these features were much older than the relatively recent mine-roof collapse.

Based upon these observations, it is reasonable to assume that the local synclinal basin (which is spatially coincident with the Central Collapse Area and the area of paleocollapse) is in fact related to the paleocollapse feature.

Since the syncline has up to 15 and possibly 20 feet of relief, it is reasonable to assume that the miners had removed rock close to, or even above, the top of the Bethany Falls Limestone. This would have clearly weakened the mine-roof within this area.

Stabilizing The Mine By Backfilling

Surface subsidence is not anticipated based upon observations within the mine. However, management has made a decision to opt for a conservative approach and continue backfilling the mine. Backfilling of the mine is now being accomplished by a crushed-rock water slurry technique, which is a much more efficient method of backfilling the mine.

CHARACTERIZATION OF HYDROGEOLOGIC CONDITIONS

The primary source of ground water in the area is from wells in the Kansas River Alluvium, which has a thickness of 40 to 70 feet, yielding from 150 to 1,000 gpm (O'Connor, 1971). Only small quantities of ground water (1 to 10 gpm) are typically available from wells placed in Kansas River tributary valleys since they are predominantly silts and clays with low transmissivities (O'Connor, 1971).

Preliminary assessment of the presence of ground water, based upon drilling a number of mine fillholes, indicated that most, if not all, of the rock at the site is unsaturated. Two open boreholes were monitored with borehole television over a few weeks and did not make any water. Two existing monitoring wells had been dry for more than a year. Localized fracture flow and flow along bedding planes were occasionally found to occur in the uppermost highly weathered Argentine Limestone.

It had been proposed to use the Drum Limestone as a zone for ground water monitoring since it was the shallowest strata to extend under the entire site. Furthermore, the literature indicated small amounts of ground water were extracted from this zone. However, the two existing monitoring wells in this zone were found to be dry and have remained dry for a number of years.

The mine was slowly filling with water at a rate of about 1 million gallons per month. Since the mine was below the level of the nearby Kansas River (Figure 1) the river is a possible source of mine-water recharge.

Initial Findings Regarding Hydrogeologic Conditions

As a result of the initial information on ground water conditions, based upon literature, two existing monitoring wells and numerous borehole television logs, the following conclusions were made:

- o The 200 feet or more of rock between the mine and the surface was virtually unsaturated;
- o The one exception was the isolated encounters of perched water in the Argentine Limestone, the uppermost rock unit at the site;
- o The source(s) and rate(s) of mine-water recharge was unknown;
- o There was uncertainty as to the hydrologic connection of the paleocollapse fracture from the ground surface down to the mine.

Additional Hydrogeologic Work Included:

- o Assessment of the hydrologic connection of the paleocollapse fracture from the surface to the mine;
- o Further assessment of perched water zones;
- o Assessment of mine-water recharge;
- o An assessment of the nature of the mine-water quality; and
- o Reassessment of the ground water monitoring strategy.

Assessment Of The Existence Of The Paleocollapse Structure And Its Hydrologic Connection To The Mine

One of the more significant fissures in the loess soil was excavated to top of the Argentine Limestone. A major solution-enlarged joint was exposed showing possible displacement of 12 to 18 inches (Figure 8).

Two angle borings were made through the vertical extension of the exposed joint (paleocollapse fracture) and observations were made with downhole television. A number of fractures and voids were observed near the projected location of the paleofracture (Figure 8). In addition, it was found that these boreholes would not hold water.

The paleocollapse fracture was also observed within the mine at a number of locations. The perimeter of the mine-roof collapse was accessible in most locations for visual observations. However, the core of the collapse zone was not accessible due to choke off by fallen rock. Around the perimeter of the collapse zone, vertical cracks were observed at three locations. Each crack was relatively tight and would not accept a knife blade. Displacement was not observed at the crack but strata dipped downward at an angle of 10 to 20 degrees on the side of the collapse. These cracks are thought to be due to paleofracturing by the collapse of a deep cavity system.

One borehole encountered a solution-enlarged cavity in the Winterset Limestone along the surface projection of the paleofracture. This cavity was found to be about 2 to 3 feet in size and obviously enlarged by solutioning because of the smooth rounded rock observed on downhole television.

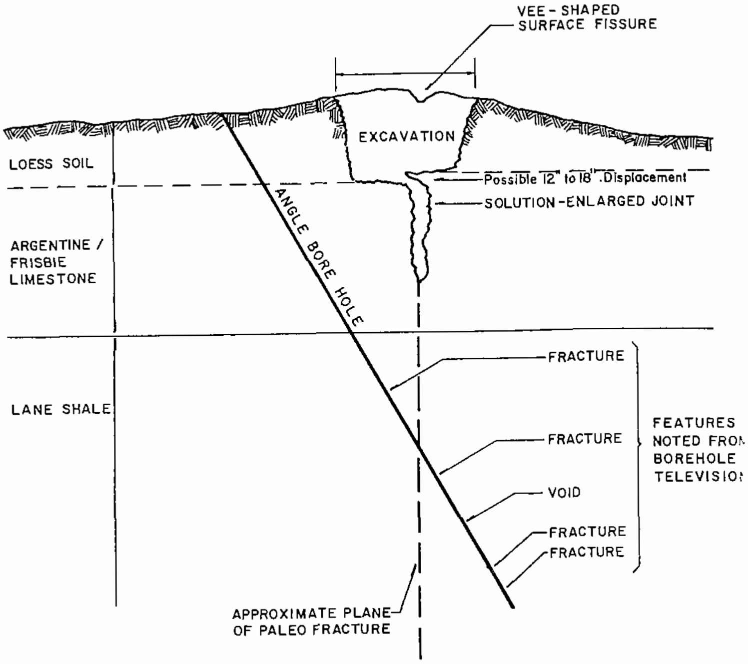


Figure 8. Cross-Section Through Surface Fissure and Paleocollapse Fracture

The hydraulic connection of the paleocollapse fracture extending from the surface downward to the mine (Figure 6) was obviously of concern since it would provide a pathway for leachate migration. This was assessed by a dye tracer program in which dye was injected along with the 30,000 gallons of water at rates of up to 200 gpm.

Seven points were selected around the perimeter of the collapse zone within the mine to sample mine-water for the presence of dye. The presence of background dyes in the mine-water was assessed by obtaining water samples and placing charcoal bugs at each of the seven sample locations before injecting the dye. After dye injection, these seven points were sampled at logarithmic intervals of approximately 1, 2, 5, 10, 20, 50 and 100 days.

Mine backfilling by the rock slurry method began at about the 133 day of the dye tracer test. As of this time, no dye was found within the mine. The process of mine backfilling involves withdrawal of mine-water which is then re-injected into the mine along with rock as a slurry at pumping rates of about 1,500 to 2,000 gpm. Mine backfilling has now occurred in 12 fillholes over more than 10 acres of the mine. More than 184 x 10^6 gallons of water have been pumped from the mine and re-injected. Since the mine contains about 60 to 70 million gallons of water, this amount of pumping and re-injection over an area of 10 acres has lead to significant mixing of mine-water. The last sampling for dye at nearly 500 days after injection was made after the significant mixing and therefore can be expected to yield any dye previously missed. Since no dye was found within the mine after a period of about 500 days, we may conclude that the paleofracture does not hydraulically interconnect the surface or rock strata hydraulics with the mine.

The lack of flow through the paleocollapse fracture can be accounted for by the presence of swelling shales (probably the Chanute Shale) which were seen to completely close off three open boreholes so that they would hold water within a 10-12 month period. The closure by shales was observed in three boreholes by downhole television and subsequent monitoring of water levels above the swelled shale.

Further Assessment Of Perched Water Zones

The injection of 30,000 gallons of dyed water provided a windfall in that it was found to have migrated laterally within the Raytown-Paola Limestones via fracture flow. Water was observed on borehole television to flow into one borehole via a fracture. This borehole was about 75 feet from the dye injection site. Samples of dye water were recovered in the Raytown-Paola Limestones at two boreholes, about 75 feet and 500 feet latterly from the injection site 1 day after dye injection. The identification of fracture flow and dye identified a possible alternate zone for ground water monitoring within the Raytown-Paola Limestones.

Assessment Of Mine-Water Recharge

Long-term monitoring of mine-water levels were made with an in-situ electronic water level instrument. Water levels were found to increase at about 15 inches/year (about 12 million gallons/year or about 23 gallons/minute).

The mine floor is underlain by a 22-foot thick sequence of shale and limestone strata. The upper 10 feet of this sequence has been determined to be quite permeable due to a very high secondary porosity related to dissolution-enlarged joints. This permeable zone consists of the Hushpuckney Shale, the Middle Creek Limestone, and the Ladore Shale. Recharge from the Kansas River (which is significantly higher than the mine) flows through this permeable zone into the mine accounting for a portion of the recharge of ground water into the mine. Other sources of mine-water recharge are also possible but were not assessed.

An Assessment Of The Nature Of The Mine-water Quality

A mine-water sampling program was initiated to characterize background quality of the mine-water and to provide geochemical characterization. Mine-water has been sampled at eight points within the mine and each of these points have been sampled three times to provide a statistically valid baseline sampling. Field parameters measured included pH, specific conductance and temperature. Laboratory analyses were carried out for the expected (not yet issued) parameters required for Subtitle D Phase I monitoring (US-EPA). In addition, the samples were analyzed for several pesticides and herbicides.

Geochemical analyses for major ions indicate that the mine-water is predominantly sodium sulfate in character. Changes in the geochemical data tend to correlate roughly with distribution of the original fly-ash backfill material within the mine. Mine-water composition sampled from areas more remote from fly-ash backfilling have higher calcium and magnesium ratios, but are still dominantly sodium sulfate in character.

There was some concern about the long-term dissolution of mine rock and crushed-rock backfill by the mine-water. However, at one location where surface water entered the mine under the mine portal road (through a rock cover of about 30 to 40 feet), extensive precipitation of calcite occurred. At this location, water dripping from the mine-roof would respond to rainfall events within 24 hours. The rapid response to rainfall events indicated that even with short travel time and distance through the limestone, the fresh rain-water had become saturated and would precipitate calcite. The observed deposition of calcite by water seeping into the mine, the high pH measured in the mine-water (7.6 to 12.4) and the positive saturation index (0.63) calculated from geochemical analyses, indicate that geochemical conditions are such that dissolution of limestone rock or crushed limestone used as the mine backfill should not occur.

Reassessment Of The Ground Water Monitoring Issue

Further drilling, hydraulic tests, and hydrofracturing were carried out in the Raytown-Paola Limestones and Drum Limestone to assess optimal locations and conditions for ground water monitoring wells. It was found that the Raytown-Paola Limestones contained water in 9 of 9 borings and piezometers and the Drum Limestone did not contain water in 7 of 8 borings, piezometers and wells.

Hydrofracturing improved hydraulic conductivity within the Raytown-Paola Limestones from 7.2×10^{-6} to 1.3×10^{-4} cm/sec. In the Drum Limestone the improvement in hydraulic conductivity was less. As a result the Raytown-Paola Limestones have been selected as the optimal location for the shallow monitoring wells.

The proposed ground water monitoring system is shown in Figure 8 with objectives to:

- o Monitor the zone of perched water within the Raytown-Paola Limestones;
- o Monitor mine-water quality; and
- o Monitor ground water within the permeable zone beneath the mine.

CONCLUSIONS

Careful site characterization consisted in large part of extensive observations and deductions to provide insight into mine collapse, mine stability, the paleocollapse structure, and aid in developing a ground water monitoring strategy at the site.

It is interesting to note that all four of the geologic constraints cited in "Geology of Greater Kansas City, Missouri and Kansas" (Hasan, et al., 1988) were found to exist at this site. They include: detrimental stability effects of older unplanned, random mine layouts; indications of the presence of paleocollapse structure such as described by Gentile (1984); infilled paleo-geomorphic channels; and thinning of roof strata (due to mining practices). The previous work by Gentile (1984) proved to be a key to the insight of the paleocollapse issue. Gentile has identified a number of these paleocollapse structures in the Kansas City area.

The mine stability issue is being resolved by backfilling the mine with a crushed-rock slurry (Figure 9). This technique has proven to be an extremely effective method and has been documented by a Quality Assurance Audit program including contour maps of the backfill (based on visual observations), along with photos and video documentation.

A ground water monitoring strategy has evolved, based upon identification of a perched water zone with fracture flow within the Raytown-Paola Limestones (Figure 9). Ground water monitoring has been enhanced by improving the hydraulic conductivity and the connection of wells to the fractures in this zone by at least two orders of magnitude through hydrofracturing the limestone rock within the screened interval. Monitoring of the mine-water quality will provide a back-up monitoring system and will also monitor any baseline changes due to the presence of the original fly-ash backfill used to fill the mine. Monitoring of the permeable zone below the mine floor (which is a major recharge zone for water flowing into or out of the mine) will provide a continued means of monitoring water quality coming into (or leaving) the mine.

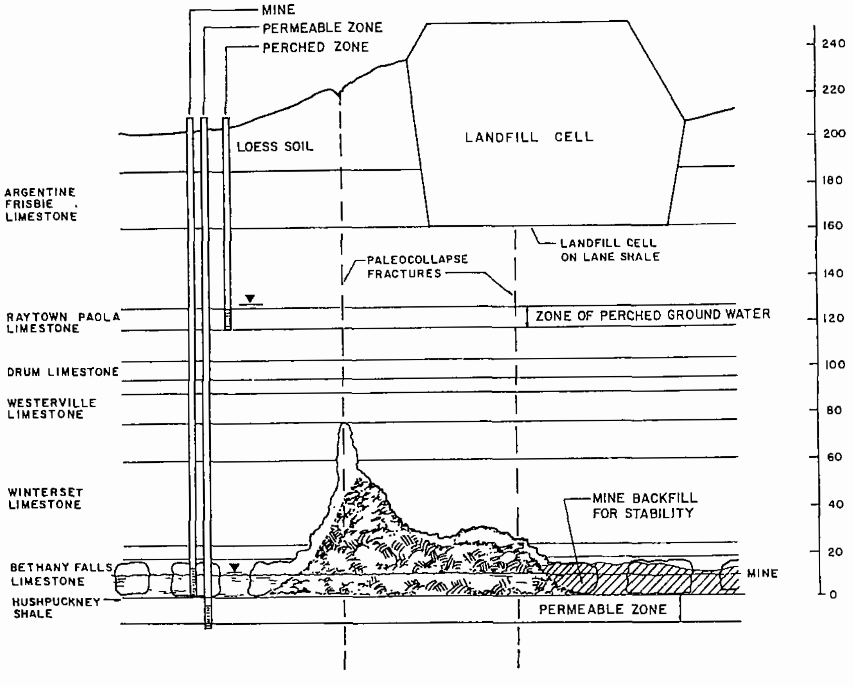


Figure 9. Conceptual Model of Site Conditions and Proposed Ground Water Monitoring System

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GROUND WATER MONITORING IN UNSATURATED AND SATURATED ZONES AT A SITE WITH PALEOCOLLAPSE STRUCTURES

Richard Benson, Lynn Yuhr and Allen W. Hatheway

1. Who was your client for this project?

   Waste Management of North America.

2. Is the increase in water level in the mine, as related to the Kansas River, part of a regional/long term trend or is it a local phenomenon related to the mining activity? How will the filling of the mine with what is being pumped into it affect water levels in the mine? What will be the effect of the rising water on the landfill?

   During active limestone mining, inflow of surface waters (portal run-in) and ground waters were pumped from the mine. Since abandonment in 1965, the mine has gone unwatered. The water level within the mine will now continue to rise until it completely fills the mine or at least to such a height above the mine-floor that is in equilibrium with the sources of recharge. One source of recharge is the Kansas River, some 1,200 ft distance from the mine.

   The crushed limestone backfill being placed within the mine is highly porous and will not affect the ultimate level of mine water at its final equilibrium. The rising water level should have no effect on the landfill, which is located upon more than 170 ft of overburden limestone and shale.

3. I do not question the reality of there being a paleocollapse (or paleocollapse subsidence) structure present. Prove to me, please, that the faulting and fissures at the surface are not a consequence of re-activation of that paleocollapse by the mining activity. If such reactivation has occurred (is occurring), how can you justify the integrity of the site until after such collapse is completed and is at some stage of equilibrium?

   A reactivation of the paleocollapse zone, from the mine workings, upward to the surface cannot occur due to mine collapse for the reasons cited within our paper (primarily 170 feet of rock over the mine, support of overburden rock by 25′ x 25′ pillars on 60′ centers, high bulking factors (an average of 42%), and bridging by the massive Winterset Limestone 60 feet above the mine-roof. Backfilling of the mine with crushed rock would also provide mitigation if significant subsidence were to be somehow occur.

   The rock, faults and soil fissures at the surface were initially caused by the paleocollapse. They have been perpetuated and possibly accentuated by the fact that they are highly permeable zones which readily accept surface water runoff. In contrast, the 20 to 60 feet of loess soils allows little percolation of surface water to the underlying rock.

   Re-activation of the paleocollapse zone above the mine-roof will not occur due to the on-going mine-roof collapse for the reasons cited. If paleocollapse reactivation would occur, such would be due to deep-seated factors, hundreds of feet below the mine floor. Although rare, it is not unknown for paleo-sinkholes to reactivate leading to further subsidence. We know of no such re-activation in the greater Kansas City area or within the tri-state (Kansas, Missouri and Oklahoma) lead mining area. Therefore, we feel that the re-initiation of paleocollapse by this cause is essentially an improbability.

Luncheon Lecture:

Chairman Mao: The Great Leap Forward, and the Deforestation Ecological Disaster in the South China Karst

Peter Huntoon,
University of Wyoming

Chairman Mao's Great Leap Forward and the Deforestation Ecological Disaster in the South China Karst Belt

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ABSTRACT

Stone forest aquifers comprise an important class of shallow, unconfined karstic aquifers in the south China karst belt. They occur under flat areas such as floors of karst depressions, stream valleys, and karst plains. The frameworks for the aquifers are the undissolved carbonate spires and ribs in epikarst zones developed on carbonate strata. The ground water occurs within clastic sediments which infill the dissolution voids. The aquifers are thin, generally less than 100 meters thick, and are characterized by large lateral permeabilities and small storage.

The magnitude and duration of the seasonal recharge pulse that replenishes the stone forest aquifers have been severely impacted by massive post-1958 deforestation in the south China karst region. The loss of seasonal upland storage in the "green reservoir" has resulted in both a reduction in the volume of recharge to the lowland stone forest aquifers and a shortening of the seasonal recharge event. This response is compounded by increased ground water withdrawals as the people attempt to offset the declining supply.

INTRODUCTION

The purposes of this article are to describe the hydrogeologic properties of stone forest aquifers which constitute an important class of thin, shallow, unconfined aquifers in the vast south China karst belt, Figure 1, and to qualitatively assess the impacts of deforestation on the stone forest aquifers. This article is an abridged version of Huntoon (1992a,b).

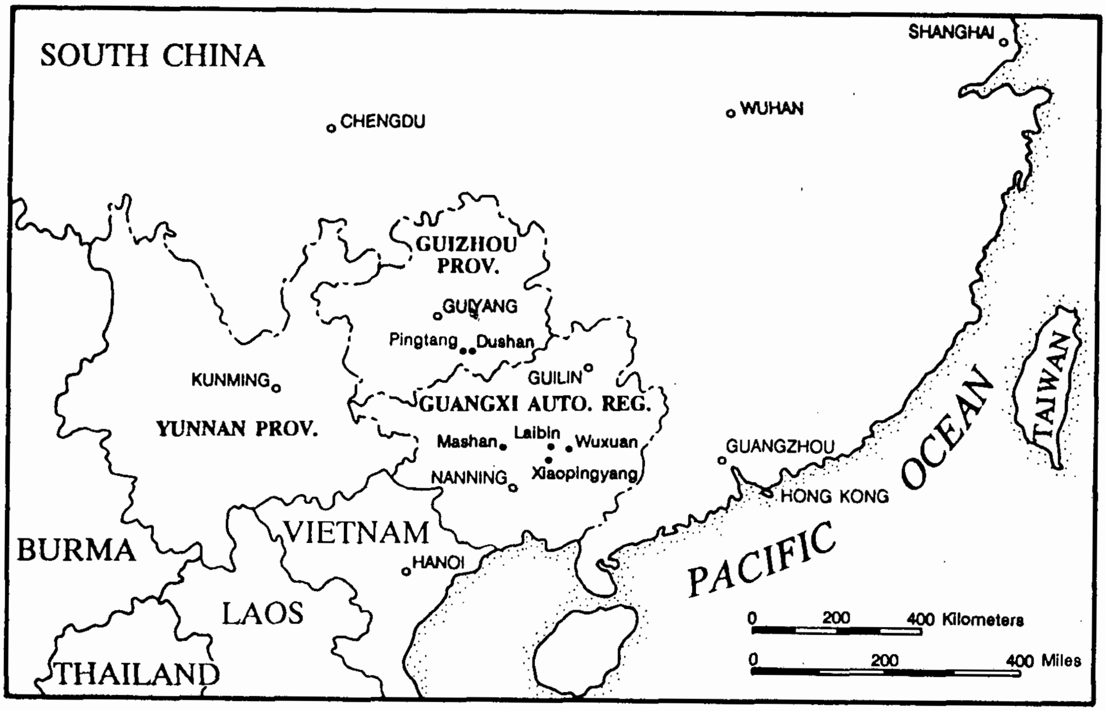

Figure 1. Location of Guangxi Autonomous Region and Guizhou and Yunnan provinces in south China.

SETTING OF THE STONE FOREST AQUIFERS

Centered around the southern Chinese provinces of Yunnan and Guizhou, and the Guangxi Autonomous Region are 500,000 km^2 of the most spectacular tower karst landscapes in the world (Zhao 1988). A class of shallow aquifers, herein called stone forest aquifers, occurs in this terrane. These aquifers owe their existence to intense but shallow dissolution of the carbonate bedrock.

The stone forest aquifers occur under flat areas within the karst belt. They are most extensive under the inland plains found within a few hundred meters or so of sea level (Figure 2). However, as shown on Figure 3, they are also common at much higher elevations including occurrences along river valleys in the mountainous areas, under large and small karst depressions within the elevated areas, and even under flats along ground water divides.

Figure 2. Carbonate peak clusters rising from a karst plain near Xiaopingyang, Guangxi Autonomous Region, China. A stone forest aquifer underlies the plain.

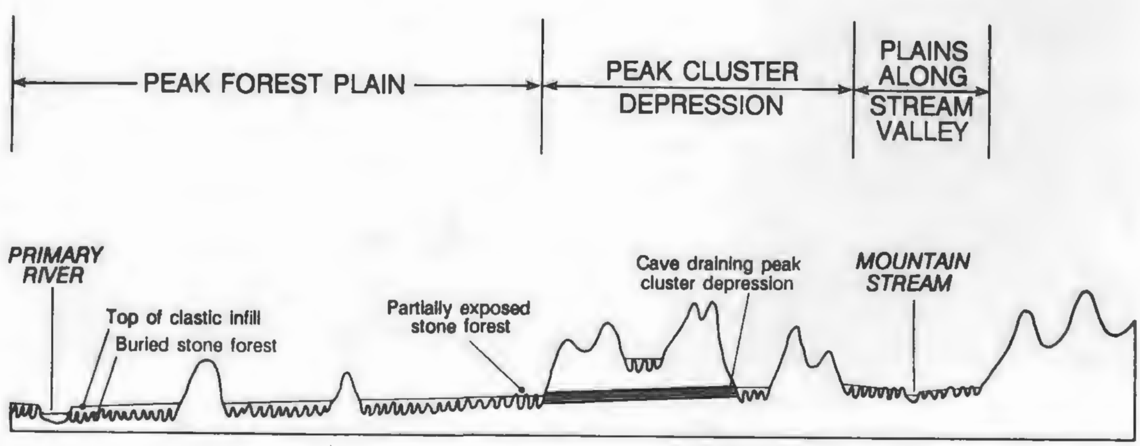

Figure 3. Schematic cross section, vertical scale greatly exaggerated, showing some typical settings for stone forest aquifers under flat areas within the south China karst. Epikarst occurs on all the carbonate surfaces; however, only the infilled epikarsts under the flat areas which host the stone forest aquifers are shown.

EPIKARST AND EPIKARST AQUIFERS

Superimposed on the south China carbonate terranes, both hills and plains, is an epikarst (Figure 4) which is an intensely dissolved veneer consisting of an intricate network of intersecting roofless dissolution-widened fissures, cavities and tubes dissolved in the carbonate bedrock. The depths to the base of the epikarst zone are variable, usually being less than 100 m. As revealed in quarries and hand dug wells, the crevassed, highly dissolved upper part ranges from 10 to 30 m deep. Widely spaced dissolution-widened fractures extend another 30 to 70 m below the crevassed zone. Externally derived sediments, soils, karst breccias and residual clays infill the solution openings within the epikarst zone.

Figure 4. The famous Kunming Stone Forest 110 km from Kunming, Yunnan Province, China. This type of bedrock surface when infilled with clastic sediments serves as the framework for stone forest aquifers.

The term **epikarst** aquifer was first defined by Mangin (1975) to describe saturated zones within the intensely dissolved veneers found on carbonate stratigraphic sections. Since the introduction of the concept of an epikarst aquifer, epikarst has been widely adopted as a noun to denote the morphology of the highly dissolved veneer itself. This usage is employed here. The epikarst zone owes its origin to dissolution of the carbonate substrate beneath a mantle of soils, residual clays, and clastic materials (Song, 1986).

The usage of epikarst aquifer by Mangin (1975), as well as by most subsequent workers, is restricted to the saturated parts of dissolution veneers occurring on elevated outcrops which are separated from underlying areally extensive aquifers by a vadose zone. Although there are important lateral circulation components within the elevated epikarst aquifers, they are conceptulized as compartmentalized collector systems which ultimately funnel water to infiltration conduits through the vadose zone (Williams, 1985). Vertical flow is thus emphasized over lateral circulation.

In China, most of the exploitable saturated epikarst zones occur on broad plains and along stream valleys. The water in them represents the top of a fully saturated substrate. There is no underlying vadose zone, and lateral circulation predominates.

STONE FOREST AQUIFERS

Storage within stone forest aquifers occurs in dissolution voids in the carbonates and in intergranular porosity within the infilling sediments. Permeability through the aquifers results from (1) interconnected dissolution cavities within the carbonate bedrock, (2) partings between the carbonates and infilling sediments, and (3) intergranular permeability within the infilling sediments. The permeabilities of the dissolution cavities and partings are often extremely large, whereas the intergranular permeabilities of the arkoses and clays are very small.

Figure 4 summarizes the porosity and permeability distributions in a typical stone forest aquifer. The fact that the stone pillars have virtually no porosity combined with the fact that the clastic infills have porosities of about 10 to 20 percent implies that the porosity of a stone forest aquifer is considerably less than that of a corresponding volume of clastic rocks. As Figure 5 shows, the reservoir capacities of the stone forest aquifers are severely limited by the thinness of the epikarst zone.

HYDRAULIC RESPONSE TO SEASONAL RAINFALL

Most of the south China karst belt lies in the humid subtropical monsoon climatic zone. In a typical year, 70 to 75 percent of the 1 to 2 m of precipitation falls during the monsoon season from April to August. This produces flooding on the karst plains and in karst depressions. The stone forest aquifers become fully recharged to the point that water levels lie at or above the ground surface. The water levels fall

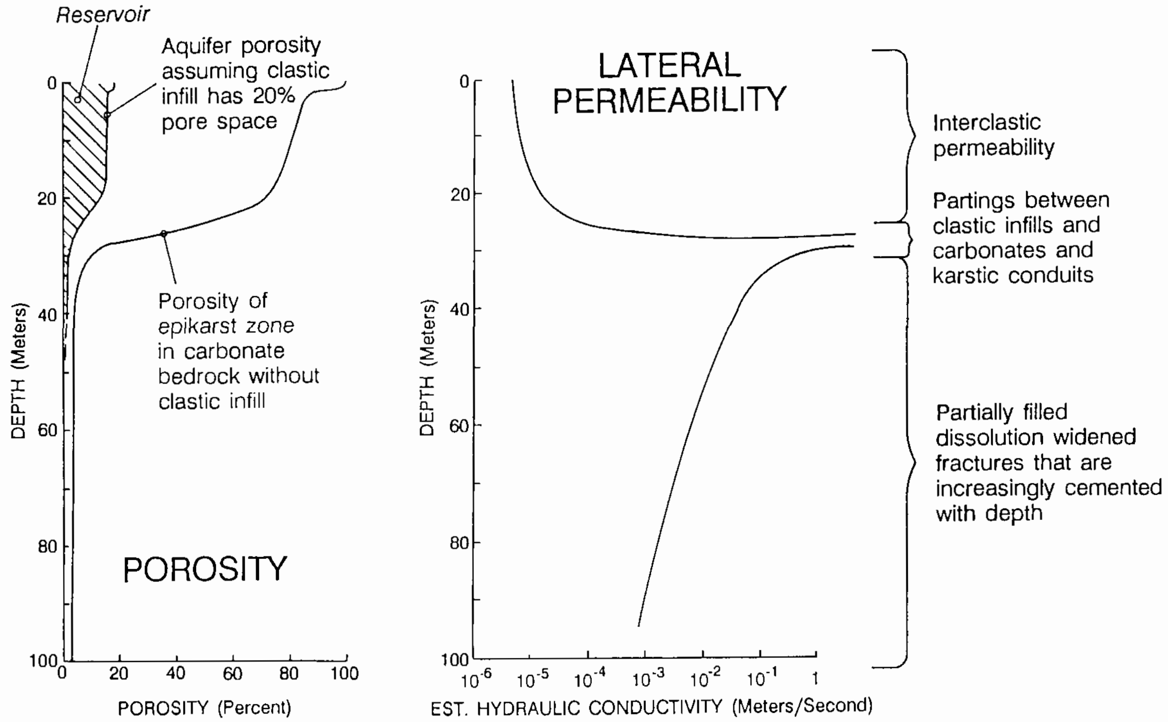

Figure 5. Idealized porosity and permeability distributions in a typical stone forest aquifer based on observations from hand dug wells and quarry walls in the vicinity of Xiaopingyang and Laibin, Guangxi Autonomous Region, China.

rapidly with cessation of the monsoon rains. Early on, the water exits the karst plains both as surface runoff and lateral discharge through the stone forest aquifers.

As the dry season progresses, increasingly greater percentages of the total discharge from the plains circulate out through the stone forest aquifers. By October or November, water levels have fallen below the land surface to depths that commonly exceed 1/2 meter or more, even in lowlands. By December, the bulk of the annual precipitation has flowed out of the region through the aquifer and the region is starting to show signs of drought.

QUALITATIVE IMPACTS OF DEFORESTATION

The deforestation of south China took place in three phases beginning with the Great Leap Forward campaign in 1958. Chairman Mao Zedong's objective in instituting the Great Leap Forward was to build a modern infrastructure in China, an effort that required steel and cement in huge quantities. The south China forests would fuel this effort. This first major assault on the forests began immediately and was particularly effective. The trees were cut and converted into charcoal. In many places in south China, as shown on Figure 6, thick subtropical forests were cut to the last tree on the tops of the most remote karst hills. Once this phase was completed, cutters returned to the forests and uprooted the stumps by hand which, in turn, were converted into charcoal. The vastness of this program is unknown to me. However, the forests were virtually eliminated, save only for a few special forest preserves, for as far as one could see from Guizhou and Yunnan provinces, and the Guangxi Autonomous Region, which I visited in south China in 1988 and 1990.

The magnitude of the deforestation is revealed not by grand statistics, but rather by the following account related to me by Chen Yao Yuan. Linxu Commune was the center of steel smelting near Mashan, Guangxi Autonomous Region. Tens of thousands of people were divided into many companies to mine, transport ore, fell trees, make charcoal, smelt steel and process timber. Many blast furnaces were built and their fires burned very brightly. In 1958, about 3,000 people were employed to cut trees, each responsible for supplying a kiln load of wood per day -- approximately 1 m^3. Every day 3,000 m^3 of wood were burned. The nearby karst hills were stripped bare.

Figure 6. Contrast between deforested karst hills and a surviving vegetated hillside behind a small village in the area south of Wuxuan, Guangxi Autonomous Region, China.

Deforestation followed in two more phases. Extensive cutting took place during the Cultural Revolution between 1966 and 1976 when anarchy prevailed and the peasants did as they pleased with nearby resources. The last serious deforestation phase occurred in the period following the decollectivization of the agricultural communes in 1979. As the peasants dispersed into the countryside, they reentered the forests to harvest the remaining trees in order to build new dwellings.

The ensuing impact on ground water supplies was the loss of what, in retrospect, the Chinese call their green reservoir. The amplitude of the flood-drought cycle has been exacerbated. This impact is compounded locally by climatic changes attending the loss of temperature and humidity moderation once provided by the forests giving rise to drier, hotter dry seasons. Desertification has begun in some of the drier areas.

The impact on the stone forest aquifers is directly attributable to loss of ground water storage in the formerly forested hills. Small springs on the flanks of the hills which were once perennial are now ephermeral. Surface flows, essential for late dry season irrigation and recharge, are now characterized by early and rapid recessions. These diminished releases of water from the green reservoir decrease dry season recharge to the stone forest aquifers which, in turn, is manifested as reduced discharges from springs and greater water level declines. Consequently, dry wells and dry karst windows have become more common as the dry season progresses. At the same time, the local population is attempting to develop more ground water to mitigate the losses.

Reforestation efforts have been undertaken. These included aerial reseeding in the early 1960s. Reforestation programs have met with some success, although two trends thwart regrowth. First, the area is experiencing a population explosion, and second, this population relies on plants for fuel. Everyday, without exaggeration, armies of peasants climb into the hills to cut brush and weeds to fuel domestic stoves and various cottage industries such as brick or lime kiln operations. The result is continuous cutting at a rate that appears to exceed steady-state regrowth in many areas. Where before a small forest area would produce relatively good quality fuel on a reasonably steady basis, now several hills must be scavanged to produce poor quality fuel having an equivalent BTU content.

The primary attribute of the ideal tree used in reforestation is that it grows rapidly. It is considered a great species if it can resprout if chopped off at ground level. Chen Yao Yuan provided the following list of species that are in common use: (1) Melia azedarach Linn, (2) Rader machera sinica (Hance) Hemsl, (3) Toona sinensis (A. Juss) Roem, and (4) Zenia insignis Chun. The Chinese foresters and hydrologists are continually seeking alternative species which they can test on karst soils.

SUMMARY

Stone forest aquifers have very large lateral permeabilities and very small storage capacities. Consequently they usually have poor development potential. Their extreme importance in the region results from the fact that they underlie most of the cultivated lowlands floored by carbonate rocks in south China.

The primary shortcoming of these aquifers is their poor ability to retain in storage the plentiful waters recharged during the April to August monsoon season. Within two to three months into the ensuing dry season, much of the water in the aquifers has drained out of the region owing to large lateral permeabilities.

The poor storage characteristics of the stone forest aquifers have been exacerbated since 1958 by massive deforestation in the upland areas. The "green reservoir" formerly retained large volumes of water in the uplands which was released to the down-gradient stone forest aquifers during the dry season, thus partially mitigating water level declines. Now this hydraulic moderation is greatly diminished. The result is a water supply crisis that has become more severe during the past few decades as a result of three mutually coupled trends. (1) The surface water flood-drought cycle has worsened through the cumulative hydraulic and climatic impacts attending deforestation. (2) Development pressure has increased on the stone forest aquifers as the surface water resources have become less reliable. (3) A population explosion in the karst belt has placed increasing demands on the total water supply and at the same time the people's requirements for wood fuels have contributed to a serious lack of progress on reforestation.

ACKNOWLEDGMENT

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Chairman Mao's Great Leap Forward and the Deforestation Ecological Disaster in the South China Karst Belt

Peter W. Huntoon

Question: Has a comparison been made between water balance calculations before and after the deforestation?

Answer: I was not privy to such information.

Session III:

Geophysical and Other Techniques for Studying Karst Aquifers

Electrochemistry of Natural Potential Processes in Karst

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Abstract

Natural potential (NP) anomalies are observed over caverns lacking streams as well as over those carrying karstwater. NP surveys have proved useful for tracing karstwater between endpoints of tracer tests and for facilitating the siting of monitor wells that tap directly into the stream course.

Electrokinetic processes are responsible for the principal observed anomalies over karst systems. A streaming phenomenon results in an electric potential gradient being established along a hydrologic flowpath that is proportional to the Darcian velocity. This gives rise to a conduction current in the reverse direction. This conduction current is measured as NP.

NP anomalies arise from four different mechanisms: refraction of regional current flow by the cave void, axial flow through the cave, infiltration into a cave from the ground surface, and flow immediately around a cave driven by evaporation and temperature.

Introduction

Irreversible processes occurring in the subsurface may lead to the generation of other, coupled processes. The primary process is sometimes referred to as a *primary flow* and may include heat-flow, fluid transport, or diffusion of chemical species. It is relatively independent of other flows that may occur and can be thought of as an independent variable. The generated process (*secondary flow*) depends entirely on the primary flow for its existence and constitutes a dependent variable. Of principal interest in exploration work is electrical current which is measured as natural electrical potential (NP). By making measurements of natural potential one may make inferences about the nature of the causative primary flow.

Diffusion or flow of groundwater through porous or fractured soils and rock is a pervasive primary flow. It is determined principally by the distribution of piezometric head and variation in hydraulic conductivity, and is quite independent of electrical potential in ordinary situations. Groundwater flow induces an electrical current and related electrical potential. The phenomenon in varying situations is called *electrofiltration, streaming potential*, or *motoelectricity*, but always results from a common underlying mechanism.

Physics of streaming potentials

A simple mathematical formalism is available to describe the interaction of arbitrary primary and secondary flows. However, a description of the physical mechanism underlying the streaming potential may provide more understanding of the phenomenon and insight into its variations.

A solid surface immersed in a liquid typically disturbs the fluid structure. A polar fluid like water will attain a preferred orientation with respect to the solid. In addition, material may dissolve from the solid and diffuse into the fluid; or material may become adsorbed onto the solid surface from the fluid.

This molecular and ionic structure produces an electrified interface displaying a potential difference between the interior of the solid and the bulk fluid. When equilibrium is attained much of the potential difference occurs across a narrow pair of charged planes at the interface. This parallel plate capacitor-like arrangement is termed the *Helmholtz pair*. An additional potential change occurs beyond the Helmholtz region through a diffused distribution of ions known as the *zeta potential*. Electric field in such an arrangement may be large; but, its direction and magnitude vary rapidly and when integrated in any particular direction it amounts to nothing.

If this electrified interface is now subjected to a flowing fluid, ions in the diffused charge region, which are of one sign, are carried preferentially by the flow. Near the interface, where fluid is stationary with respect to the solid, ions, which are of opposite sign, are stationary. The charge distribution is deformed and oriented in the direction of the fluid flow.

With an orienting fluid flow the integrated electric field now produces a nonzero electric potential along the flow known as the streaming potential. An important observation is that a distributed charge region (or zeta potential) is required to produce a streaming potential. Whatever eliminates the zeta potential, increased ionic concentration for instance, will eliminate the streaming potential. A reversed zeta potential reverses the sense of the streaming potential. Ordinarily positive ions are located in the diffused charge region. However, acidic groundwater against carbonate rock may result in a predominantly negative diffuse charge. Thus, a particular groundwater flow may produce a positive or negative NP anomaly.

Laboratory experiments have shown that NP acquired through flow of water in porous material is linearly proportional to Darcian velocity over a broad range of pressure gradient and fluid composition (Bogoslovsky and Ogilvy, 1972). Presumably this is also true of flow through fractures. Thus, one may justifiably treat streaming potential with a linear model.

Mathematical Model

In order to model NP one requires a mathematical theory based on the underlying physical process. In the context of NP arising from groundwater flow one need consider only the coupling between fluid flow in a porous solid and electrical current. The linearized, coupled equations describing this in 2x2 matrix form are:

$$\text{I.} \qquad \begin{bmatrix} j \\ v \end{bmatrix} = \begin{bmatrix} \sigma & C_{\sigma k} \\ C_{k\sigma} & K \end{bmatrix} \cdot \text{GRAD} \begin{bmatrix} \phi \\ h \end{bmatrix}$$

where; j = electric current density
σ = electrical conductivity
$C_{\sigma k} = C_{k\sigma}$ are cross coupling (Onsager) coefficients
GRAD = spatial gradient operator
ϕ = electric potential
v = Darcian velocity
K = hydraulic conductivity
h = piezometric head

If additional flows such as heat-flow or chemical diffusion must be included, the system of equations may expand to 3x3 or 4x4. In general the coefficients in these equations are functions of position; $\sigma = \sigma(X)$, $C_{xy} = C_{xy}(X)$, and $K = K(X)$. However, assume for the moment that material is homogeneous. Taking a divergence of both sides, and focusing attention on the first equation only results in

$$\text{II.} \qquad \text{DIV} \cdot j = -(\sigma \nabla^2 \phi + C_{\sigma k} \nabla^2 h).$$

Thus electric potential depends on piezometric head. The converse is also true but electrical potential has an insignificant effect on fluid flow in normal geological environments (electro-osmosis and electrophoresis are examples of the converse situation).

Setting $DIV \cdot j = 0$, which is reasonable in a conducting earth, causes equation II to assume the following form

$$\text{III.} \qquad \nabla^2 \phi = -C_{\sigma k}/\sigma \, \nabla^2 h$$

Since h is the primary potential we may assume that it and its Laplacian are known in advance. In this case Equation III is simply Poisson's equation; the behavior and solution of which is the subject of potential field theory. In simple situations an analytical solution is possible, but more often one must find the solution by approximate or numerical means. However, before any solution is possible one must examine complications which arise from time dependence, material inhomogeneity, and boundaries.

Time dependence

Although not shown explicitly in the mathematical model, time dependence may enter through two terms.

DIV $\cdot$ *j* is equal to the negative time rate of charge accumulation. It is not possible to accumulate charge in a conducting earth, so one may justify setting this term to zero.

Additional time dependence may enter through the primary flow. At steady state in homogeneous material $\nabla^2 h$ is zero. However, this term is not zero if the primary flow has a time dependence. Time dependence often enters NP field observations through propagation of the diurnal temperature wave into the soil. Since groundwater systems often have seasonal components of flow, one might also expect a direct time dependence of streaming potential.

Inhomogeneities

In inhomogeneous material the taking of a divergence in the linearized, coupled equations (Eqs. I) will result in terms of the form *GRAD X* $\cdot$ *GRAD H* where *X* is any material property and *H* is the corresponding potential. Thus, inhomogeneity, even in the material properties of the primary flow, will provide a source of natural potential.

Inhomogeneity creates distributed current sources that act proportional to the gradient of the primary or secondary potential. An extreme example is a sharp interface between dissimilar materials in which case there is a surface density of current sources on the material boundary that equals the product of potential gradient normal to the boundary and the difference in material property. Inhomogeneity may also occur as a gradual variation of material property over a finite distance. In this case there is a volume density of current source.

Boundaries

In homogeneous material at steady-state, where the Laplacian of the primary potential is zero, boundary conditions determine entirely the form of the natural potential. This appears to lead to a belief that sources of NP arise only on boundaries between dissimilar material. This is true only under the restrictive assumptions of steady state and homogeneous material. As we have shown, inhomogeneity and time dependence each add sources of NP.

Primary and secondary flows each must meet certain boundary conditions which need not be the same for each flow. For instance, electric current flow at the earth surface must be parallel to it; whereas, fluid flow may originate at the surface or pass through it. Fluid flow may not cross an aquiclude; whereas, electric current may pass through it unimpeded.

Boundaries may be replaced by a distribution of sources that force each potential to meet the required conditions. This is not a mathematical fiction. Because of changes that occur at boundaries in hydraulic conductivity, electrical resistivity, or other material properties, electric current sources actually accumulate there. If there exists a means of specifying these boundary sources the problem of determining the natural potential simply reduces to that of integrating the source density.

In the simplest case, where the earth is assumed to be homogeneous for both primary and secondary flows, the most important boundaries are the ground surface and cavern wall. Since electric current may not pass across either of these the ground surface and cave wall must be electric current streamlines.

Models of natural potential near caverns

There are several different effects that individually contribute to NP observed over and within caverns. For example, the void itself may refract regional currents to form a local anomaly; fluid flow within, around, and above the cave may contribute a streaming potential; and it is also possible that chemical diffusion or thermal convection play some role.

Influence of the cave void

Embedding a high resistivity cave (void) into a conductive material will refract any regional electric current. The regional current might be unrelated to the cave or even unrelated to anything geological. For example, there may be cathodic protection on a nearby pipeline.

Consider a model consisting of a horizontal cylinder (cave) of radius a at depth h as shown in Figure 1. A regional electrical current flows from right to left. An approximate electrical model of the cave is a dipole current source centered at $y = h$ with sufficient strength to make the walls of the cave a current streamline. Introducing this dipole disturbs the streamline on the ground surface. Addition of an image dipole at $y = -h$ will restore the surface streamline but disturb the cave streamline. Thus, one would have to have an infinite series of current dipoles and images to achieve an exact solution. However, a model consisting of a single dipole and its image is sufficiently correct to calculate the effect of the resistive cave.

As Figure 1 shows, the regional potential gradient, which is uniform in the absence of a cave, exhibits an anomalously steep decrease over a cave. The residual NP anomaly consists of a paired high and low with the crossover point directly above the cave. These total and residual anomalies are unlike those actually observed over most caves. In particular the theoretical effect of a void is to cause a deflection of the regional potential gradient without producing a local maximum or minimum. This is still true if one assumes that a region of altered resistivity exists above the cave.

One generally observes a local extremum of natural potential over a cave. From this observation alone it is obvious that natural potential observed over caves derives from mechanisms beyond simple current refraction.

Fluid flow above a cave

Presumably caves are loci of enhanced permeability. They occur along joints, fractures, or faults. There is reason to believe that there will be anomalous fluid flow above caves from this enhanced permeability. Also having a void in the subsurface may cause large and strangely directed piezometric gradients. It makes sense to assume that infiltration in the soil above a cave will be directed toward the cave in some way and enter it via the ceiling. This does not mean that water will drip from the cave ceiling -- it may simply evaporate. Here there will be a distributed positive source not much wider than the cave itself. Near the ground surface, at the point of infiltration, there will be a distributed negative source. This may be broad or narrow depending on the width of the infiltration zone.

Assume in the simplest case that the two distributions are of equal width. They also have equal charge density since by charge conservation their sum is zero. This distribution of vertical bipoles, shown in Figure 2, will exhibit a surface potential transverse to the cave that has a deep central negative anomaly superimposed on a broader, shallow negative anomaly. The broad shallow anomaly results from an inverse square decline of potential away from the sources; while, the additional deep negative anomaly results from modulation by a $\cos(\Theta)$ term. The important observation here is that the anomaly has one sign. Under the assumption of equal source density for positive and negative sources, and a vertical orientation of bipole axes, NP measured on a surface profile will always be of one sign.

Equal current source density for recharge and discharge flow regions seems an unlikely situation and restricts the variety of NP anomalies that one expects to observe. More likely water inflow to a cavern is collected over a wide region and funneled into a narrower region at the cave ceiling (Lange and Quinlan, 1988). Providing that the recharge region is quite broad, that the discharge is quite narrow, and that the vertical distance between the two is not large, the central portion of the NP anomaly reverses polarity. What results is a sombrero-type anomaly of one polarity or the other depending on groundwater chemistry. Figure 3 shows this situation.

Flow within a cave

Caves may contain flowing water or they may simply contain wet sediments on their floor.

A cave whose cross-section is completely filled with flowing water ought to have an NP field like that of a flowing stream. Fluid flow in this case is predominantly along the cave axis and constitutes a three-dimensional effect to be covered in the next section. An additional approximately 2-D flow pattern contributes NP from two sources; a diffusion front and a flow field.

Krajew (1957) observed that NP changes abruptly at the bank of a surface stream. To explain this he idealized a stream as providing a source of water to infiltrate the banks and bed producing a diffusion front between waters of differing chemistry. The diffusion front is an electrical double layer with the most mobile ions forming the layer farthest from the stream axis. When an image of the surface stream is added to make the ground surface an electrical streamline, the result is a closed electrical double layer in the shape of a cylinder coaxial with the stream. A theorem from potential theory states that the interior of this surface (the stream) will have a constant potential equal to ½ the potential difference across the double layer. The exterior of the surface should have zero potential. This model explains the abrupt change in potential at the stream bank; but, it does not square with observations taken across caves. According to this model, a cave completely filled with water should exhibit no NP anomaly since all ground surface profiles are exterior to the cave. Yet, NP anomalies are always observed on profiles transverse to caves.

Contrary to Krajew's idealization water does not necessarily flow radially outward from a stream or a cave. Caves, in particular those in dense carbonate rock, allow virtually no penetration of the surrounding rock through pores, but only along joints and fractures. This leads one to conclude that a perfectly cylindrical electrical double layer from diffusion is not likely to obtain, and that some NP signal from diffusion exists exterior to a cave. The resulting potential could take a variety of forms and it make little sense to enumerate all possibilities here.

Caves that contain free water may, like streams, be gaining or losing this water along their axes. Water is either entering the cave from the surrounding ground or leaving the cave to its surroundings. In the first case one expects cave walls to appear as a negative current source while the surrounding earth acts as a distributed positive source. An opposite situation holds in the second case. Our electrical model predicts a concentrated potential anomaly in the midst of a distributed anomaly of opposite polarity. The distributed source might be so diffused that it may not be readily apparent and not observed against background NP variations. In this case one observes what appears to be an isolated positive or negative anomaly.

Fluid flow immediately around a cave implies, again, generation of a sombrero-type NP anomaly. Figure 4 shows one possible model and its resulting NP profile. Note here that the anomaly has small amplitude which is a reflection of having sources of one sign imbedded within sources of opposite sign. The geometry is like a cylindrical double layer which has a zero external potential.

Three dimensional effects

Like surface streams, caves may be gaining flow in one segment and loosing in another. Two-dimensional models presented above are therefore abstractions. Only very unusual circumstances permit a purely two-dimensional fluid and current flow around a cavern.

One obvious three-dimensional effect is the obeservation that a stream in any segment of the cave will flow in at one end and discharge at another. This applies to the terminal openings of a cave at ground surface, and also to any place within a cave where axial flow speeds up (recharge) or slows down (discharge). Ions of one polarity are carried by a stream from recharge to discharge and a return current is set up in the surrounding rock or along the rock-water interface. A potential gradient along the cave axis is thus created.

A simple model for this situation is like one used to model NP over sulfide ore bodies (Kilty, 1984). The model, as shown in Figure 5, consists of a bipole current pair positive at the discharge end and negative at the other. A profile transverse to the bipole axis at the discharge end will exhibit a positive anomaly directly over the cave discharge. An opposite anomaly occurs over the recharge end. Somewhere between the two a transverse profile may have a zero anomaly, but this is not likely to be observed in any real situation.

Additional three-dimensional NP effects arise from influences unrelated to the cave itself. For example, there is often a correlation between ground surface elevation and NP. A streaming potential from draining vadose water is the most likely cause. This anomaly can be confused easily with that generated by sources directly related to the cave.

Cultural effects can be very important in populated areas. NP is generated by metals corroding in soil, cathodic protection systems, electrified railroads, and Faradayic rectification at ground points of the power distribution system. We have observed TV and radio towers to be the locus of NP anomalies. Grounded wire and chain-link fences have an influence, as observed by us in a number of surveys.

NP anomalies in dry environments

Certain caves over which Lange et al (1990) have observed NP anomalies are in very dry environments where very little moisture is expected in the surrounding ground. However, humidity within these caves can be very high (100%) and the floor can be muddy or wet. That an NP anomaly is observed at all suggests that there is at least enough soil moisture in the surrounding rock to provide electrical conduction; and this moisture may still participate in a fluid flow.

The floor of a cave that contains damp sediment acts as a source of water which may drain under the influence of gravity into the cave floor. In this case the cave floor will appear as a negative anomaly and the surrounding rock as a broadly distributed positive.

Even if water is not drain it may move under variations in capillary forces from evaporation or variations in temperature. It may be wicked up along or near the cave walls to re-enter the cave along its upper walls or ceiling. One would expect the cave floor to be a negative current source and the ceiling to be positive. Ceiling/floor polarization is reported in several caves by Lange and Quinlan (1988), Lange et al (1990), and Lange and Wiles (1990).

A ceiling/floor polarization constitutes a bipole current pair of essentially equal size and density. It should lead to a anomaly of a single polarity, the polarity being that of the current source nearest the ground surface.

Although there are no supporting NP observations for this, a cavern may act as an evaporative sink for capillary water in the surrounding ground. The entire perimeter of the cave is a discharge in this situation, and should exhibit a single electrical polarity. It resides within a broad and diffused current source of opposite polarity, the two sources constituting an electrical double layer. If the double layer were cylindrically symmetric it would result in there being no NP anomaly on a surface profile. More realistically, however, the double layer would have some cylindrical asymmetry leading to a wide variety of NP anomalies. In particular, if the line connecting centers of negative and positive current sources is not vertical a positive/negative pair of NP anomalies is generated; and, since one source is more compact the resulting anomaly would appear as an asymmetric sombrero.

Conclusions

We have examined several possible models of generating NP anomalies near caves. Variations in NP occur both transverse to the cave axis and longitudinally along it. Despite the apparent diversity of possible mechanisms they all share one characteristic. Each generates a paired positive and negative current source that are closely situated in space. If the two sources are oriented with one vertically above the other, and with the sources of equal current density, the resulting NP anomaly is of one sign. However, in reasonably realistic situations one current source of the two is more compact. This produces sombrero-type anomalies consisting of a sharp NP anomaly of one sign placed within a broader, more subtle one of opposite sign. At times the broader anomaly may be so subtle that it goes unrecognized in normal background noise.

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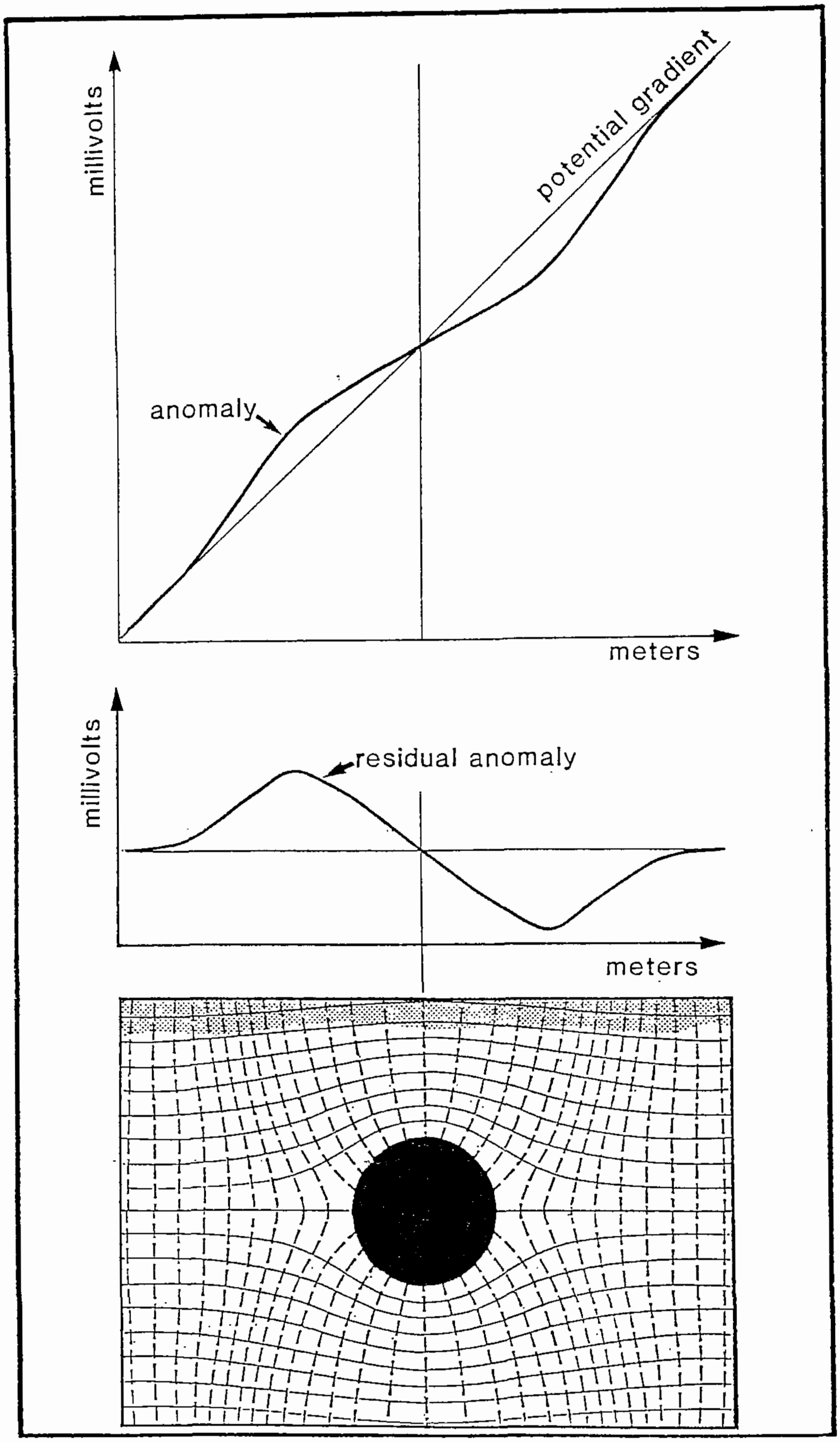

Figure 1. Regional electrical current flowing from left to right is refracted around the void representing a cave. This deflects the profile of potential away from a constant gradient as shown. A void may produce an anomalous potential gradient, but it cannot produce extrema of potential.

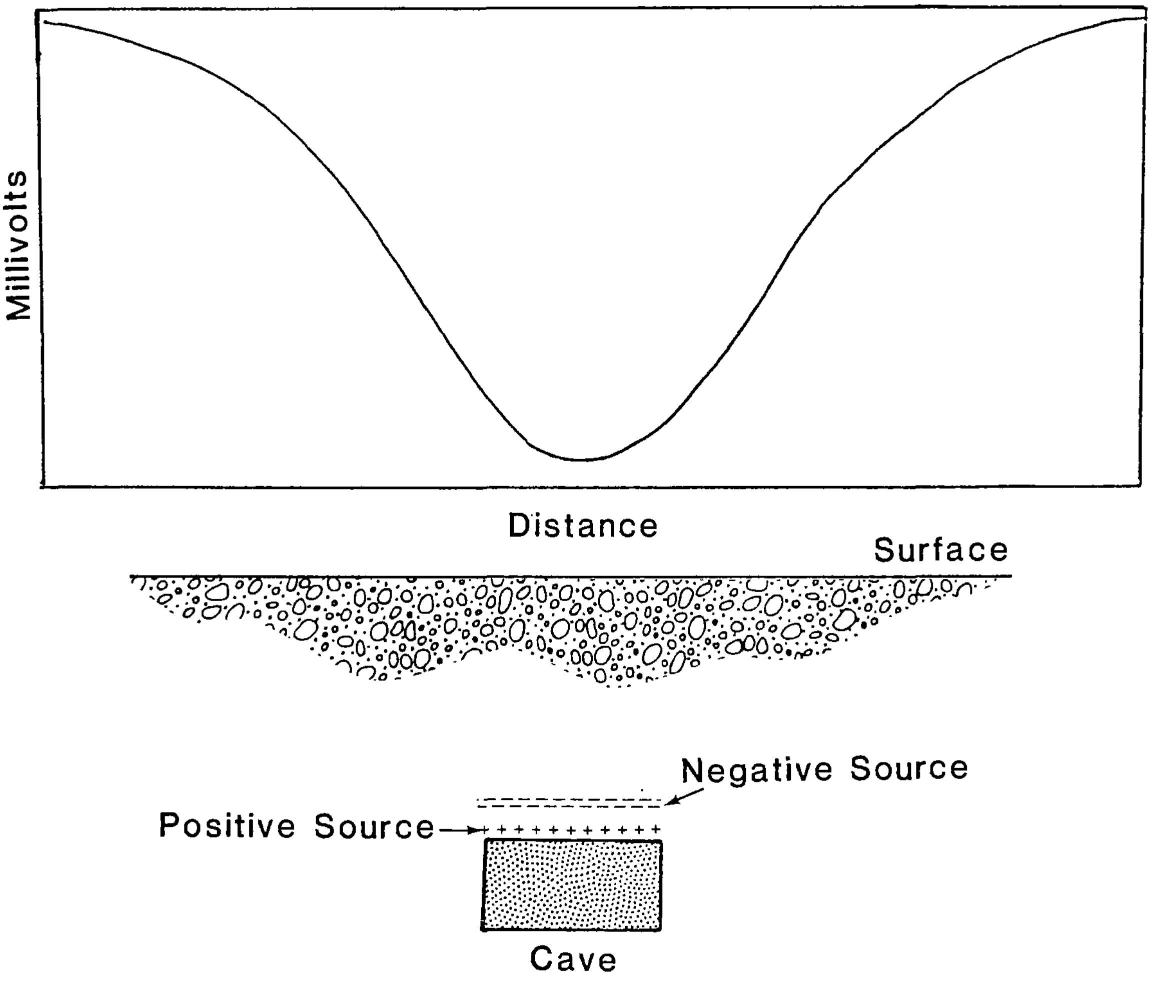

Figure 2. An idealized electrical model of infiltrative flow that originates in the soil above a cave. In this particular case the negative and positive current sources have equal density. Because the negative source is closest the ground surface, the resulting potential is entirely negative.

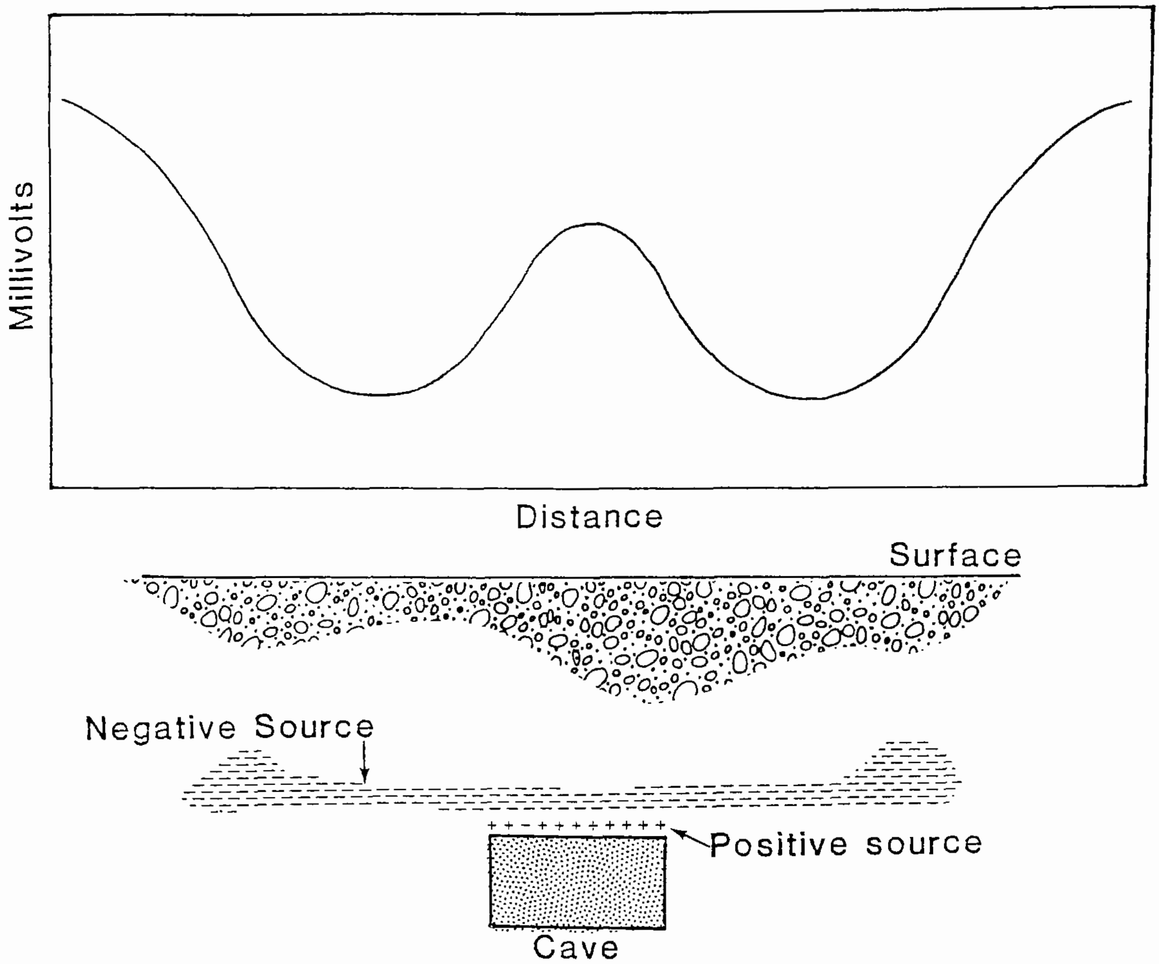

Figure 3. Another possible model of infiltrative flow. In this case the negative current source is very broadly distributed. Groundwater flow is funneled into a narrow region directly above the cave. This results in a more compact positive current source at the cave ceiling. An NP profile at the surface above this cave would have a sombrero shape.

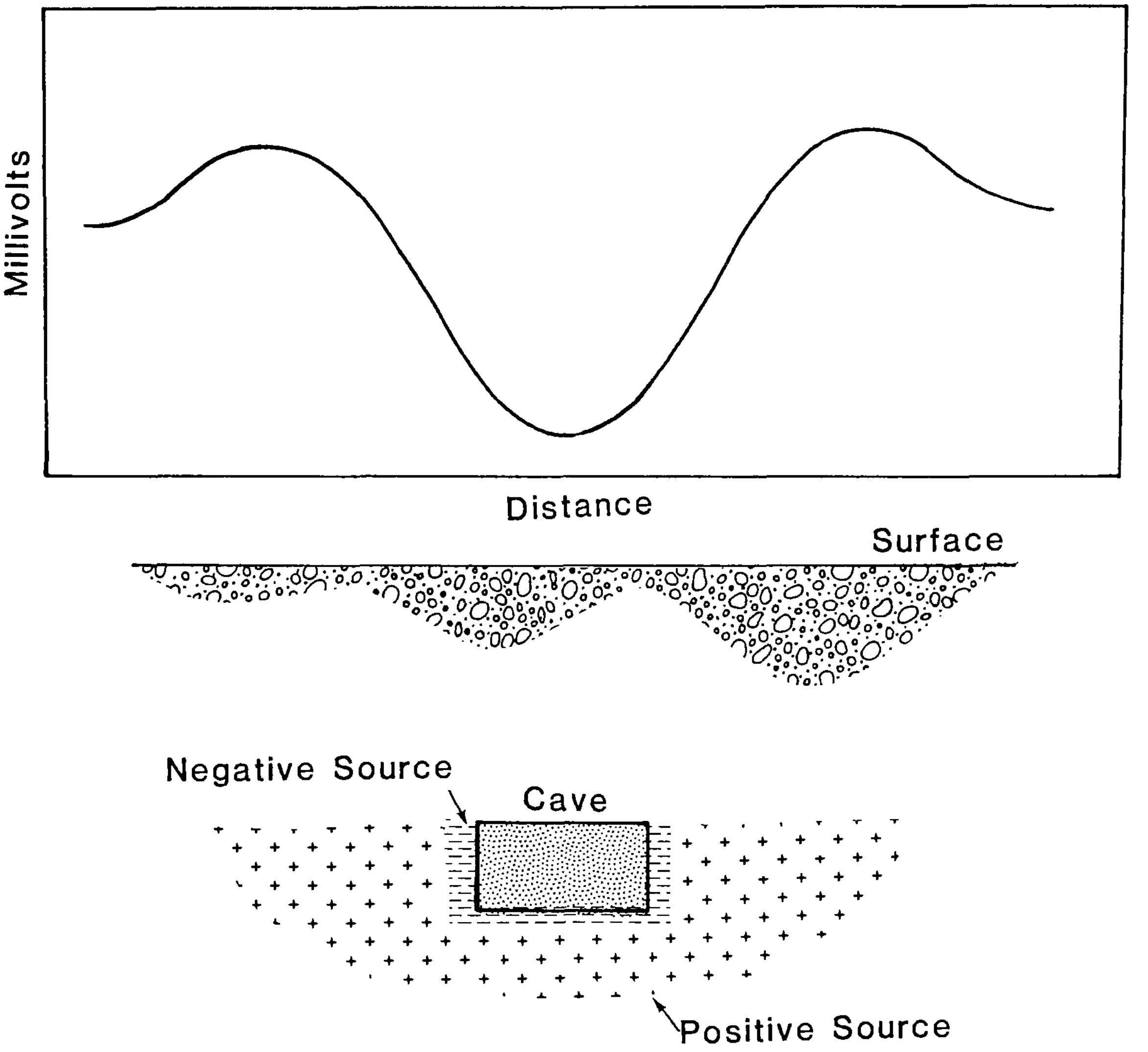

Figure 4. Water flowing or draining away from a cave results in a compact negative source at the cave walls and a distributed positive source around the cave within a large volume of rock. An NP profile at the surface over this cave would show an inverted sombrero shape. Since the negative current source is located within the positive source, the anomaly magnitude is greatly reduced over that of figures 2 and 3.

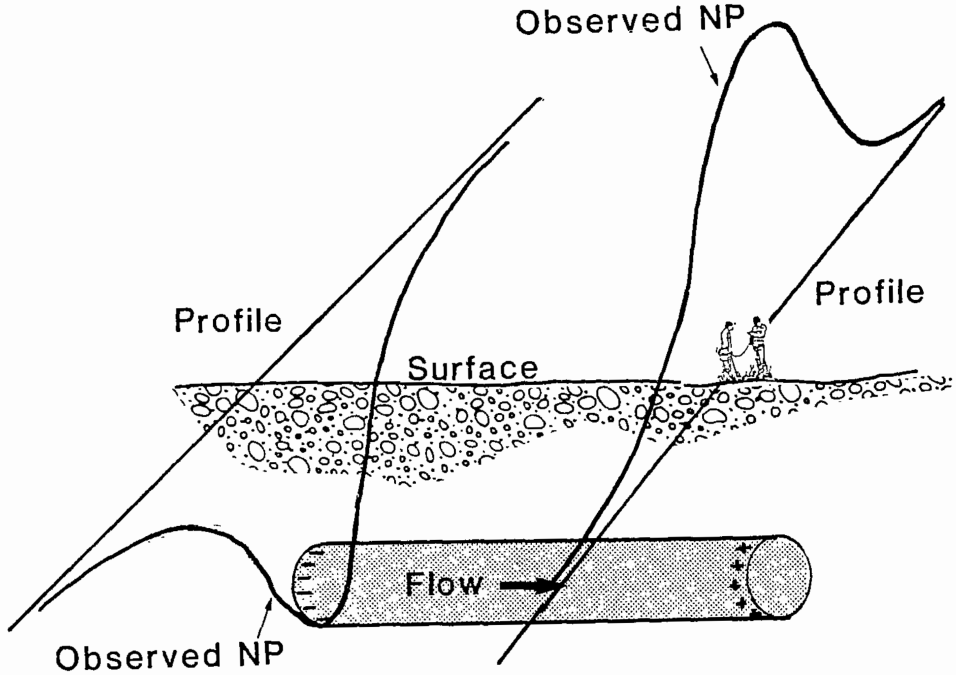

Figure 5. Water flowing axially through a cave creates a longitudinal NP gradient. At the recharge end of the cave NP is predominantly negative: at the discharge end it is predominantly positive. This 3-dimensional effect is in addition to 2-dimensional flow effects and results in additional variety of observed anomalies.

Kevin T. Kilty
Response to question "What would be the most appropriate way to confirm the cause of a compound (s-shaped) anomaly?"

Presumably by s-shaped the interrogator means sombrero-shaped anomaly, since this type of anomaly has a compound source of positive and negative currents. Two entirely different approaches might provide confirmation of the cause of such an anomaly.

In very specific cases, a second geophysical method could provide confirmation. For example, in the case of an anomaly resulting from an underground cavern, a gravity survey or resistivity survey could confirm the presence of a void space if the void were shallow enough for these methods to detect it. In many cases gravity or resistivity surveys cannot detect a void because it is too deep or too small to resolve. Natural potential might detect the same void because of altered ground water flow between the void and ground surface.

In many cases no means of verification exists other than drilling. The shape of the NP anomaly provides information about the maximum depth of the cause, and if drilling proceeds beyond this depth without finding a cause, then one must assume that the NP results from broadly distributed, shallow sources. In this case the cause is difficult to confirm no matter what approach one takes.

The most appropriate method will depend on which of the above methods is least expensive and provides sure confirmation. Most often drilling is the choice.

NATURAL-POTENTIAL RESPONSES OF KARST SYSTEMS AT THE GROUND SURFACE

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ABSTRACT

Caverns and underground karst streams have produced characteristic natural-potential (NP) signatures at the surface in a variety of karst environments. These effects have been mapped in regions as diverse as the Central Lowlands, Black Hills, Great Basin and the Edwards Plateau. Since the discovery of the NP phenomena over karst features in 1986, the technique has been tested for purposes of 1) Siting monitor wells; 2) Mapping extensions of known karst systems; 3) Avoiding contamination through placement of roads and structures over such systems; and 4) Minimizing the effects of petroleum drilling and deteriorating casings on caverns and groundwater.

Several different mechanisms contribute to the generation of the NP anomalies, involving, for the most part, the *streaming*, or *electrokinetic*, effect. In this process an electric potential gradient is established by water flowing in the ground, whether through grains of soil, rock fractures, or open conduits. Anomalies can result from water infiltrating the roof of a cavern; from capillary action of water moving upward from the moist cave environment; and from streams flowing bodily through open or tube-full conduits. In addition, effects of a void can be expected in the electric field produced by nearby, strong artificial sources.

Examples from caverns in the Edwards Plateau of Texas and Jewel Cave National Monument, South Dakota illustrate the surface effects over caves lacking known streamflow. Particularly sharp anomalies characterize flowing underground streams; such as, Honey Creek Cave, Texas; Parker Cave, Kentucky; Cave Valley Spring, Nevada and Lost River, Indiana.

Introduction

This paper provides field evidence from diverse regions of the United States demonstrating the existence and nature of natural-potential (NP) anomalies associated with cave systems and karst conduits. Hopefully, these examples will help to validate the electrochemical and physical mechanisms hypothesized in the previous paper by Kilty & Lange (1991). Actually, the mechanisms were arrived at inductively, based on field evidences accumulated over karst systems since 1986; hence, the models are the product of cogitating the field data, rather than the cornerstone on which the field tests were founded.

Acknowledgments

The data that we view here have been gathered in part through the research efforts of the authors; but also as a result of field projects funded by state and federal agencies. These include investigations for the development of Kartchner Caverns as a State Park for the Arizona State Parks Department; tests over Kentucky caves sponsored jointly by Mammoth Cave National Park and the Environmental Protection Agency; and studies over Jewel Cave National Monument, carried out under a grant from the National Park Service. These field projects were reinforced by a detailed investigation of the published literature on the natural-potential method funded also by the E.P.A., Las Vegas. In addition, many individuals and cave managers have played a role, and these have been cited in earlier reports and publications covering particular surveys.

Field procedures

Natural potentials, either on the ground surface or underground in a cave, are in effect *voltage distributions*, arising from natural or artificial d.c. currents in the ground. These currents occur everywhere upon the earth's surface--even in the oceans--and arise from redox processes around buried metals or mineral deposits; localized thermal sources and solar heating; biological activity and chemical gradients; artificial electrical sources such as cathodic protection devices on pipelines; and most importantly, from water moving into and through the ground. We can liken these sources to a collection of batteries of all different strengths and orientations, buried in the earth. What we measure between any two points on the ground surface is the difference of potential (voltage difference) resulting from the combined current of this entire assemblage of batteries.

Voltage differences are normally measured using probes; that is, electrodes, such as one would employ to troubleshoot a radio or TV set. In field applications where measurements in the millivolt range are required, the usual metal probes or stakes are unsuitable owing to their tendency to polarize or build up a charge; we employ instead non-polarizing sealed electrodes composed of a particular metal immersed in one of its own salts; most commonly a copper rod immersed in a bath of copper sulfate. The typical NP survey utilizes one stationary electrode partly buried at a central location--the reference or base electrode--and a second roving electrode that samples the ground at appropriate intervals along a grid of lines, oriented generally orthogonal to the axis of the target, such as an underground stream. The two electrodes are connected to a precision, high-impedance multimeter by means of a long, color-calibrated insulated wire wound on an aluminum reel. In karst work, digital meters that read to millivolts or tenths of millivolts are required; and under dry conditions, input impedances of 1000MΩ (megohms) or greater may be necessary. Under unusually noisy conditions it is necessary to employ signal processing equipment in order to obtain representative readings.

Readings are made by inserting the roving electrode in shallow (10cm) holes at intervals of 3 to 10m or less, depending on the target size and depth. Stations as close as one meter may be required to resolve small conduits. We normally take two or more readings at each station to insure that they are representative of the location. Furthermore, multiple readings are made *in situ* around the base electrode before and after reading every line, in order to measure temporal drift. Drift occurs in the electrodes due to temperature changes and, in the soil, to temperature, moisture and chemical fluctuations.

When electrode separations exceed one hundred meters or more, it may be necessary to install a stationary array of electrodes and a recorder to

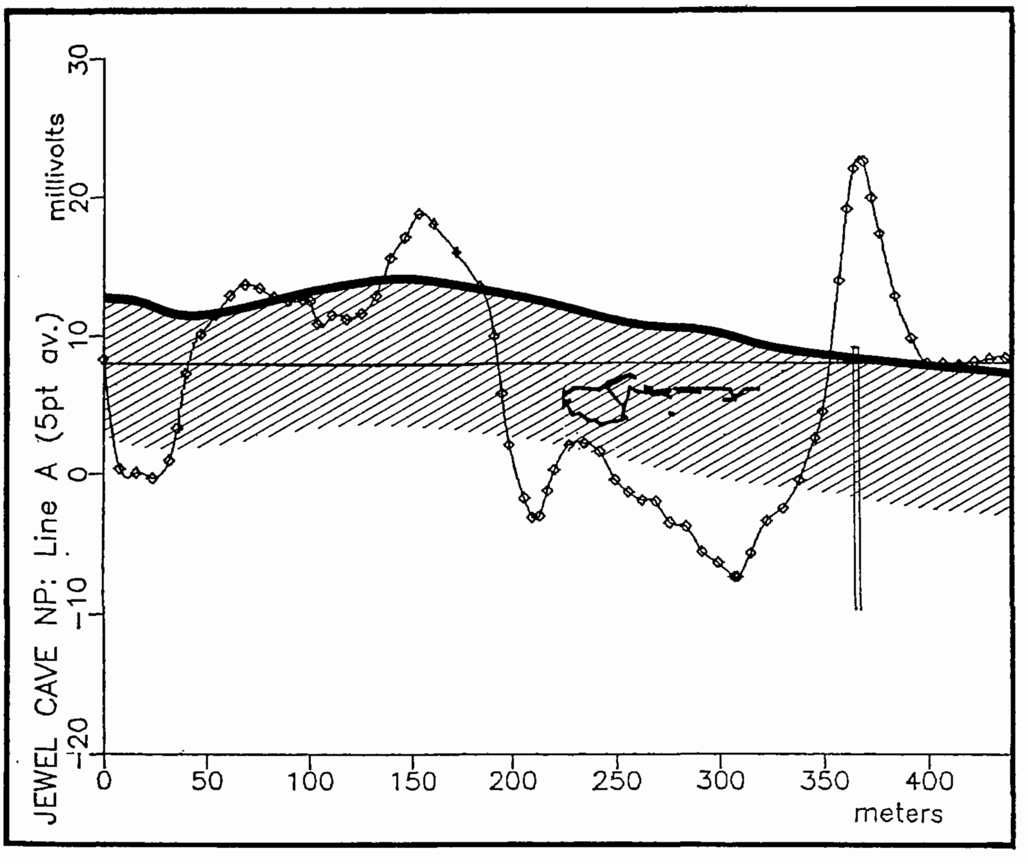


Figure 1. Natural-potential profile over a section of Jewel Cave, showing the typical infiltration negative associated with basic water. Positives on either side may constitute fringe effects of the anomaly; however, that on the right is in the vicinity of the water well to which it may be related. The leftmost low is a likely fault zone that may contain solution work.

monitor temporal variations resulting from natural telluric currents or urban electrical systems. This is particularly the case in areas of highly resistive ground during magnetic storms arising from solar flares.

At the end of each day field data accumulated in a log book or data logger are entered into a portable computer, so as to monitor the survey results. Specially designed programs correct for drift, and adjust for slope distances, topography and artificial potential gradients. In the case of a systematic, gridded survey, the final output is a display of stacked NP profiles, potential contours and an interpretive map, illuminating targets of interest, be they mineralized bodies, buried channels in alluvium, or active karst conduits.

Correlations of NP anomalies with subsurface voids are best viewed in cross-section; thus, the examples that follow will be in the form of profiles and their corresponding underlying structure as deduced by the geophysicist. In some cases these profiles will be drawn from a set of parallel profiles; in others, they will be individual lines measured for testing the response over a known or conjectured cavern. Figure 1 represents one of four trial lines over the labyrinth of Jewel Cave National Monument, wherein the underlying cave cross-section is an accurate portrayal of the mapped passages crossed by the surface profile. In the following observations, we present at least one such example for each of the generative mechanisms expounded in the previous paper (Kilty & Lange, 1991).

From this discussion, it should be evident that meticulous data-acquisition procedures and processing are required to resolve typical karst features from the surrounding electrical noise, even in remote locations. Understanding of the NP record and its correct interpretation is an art

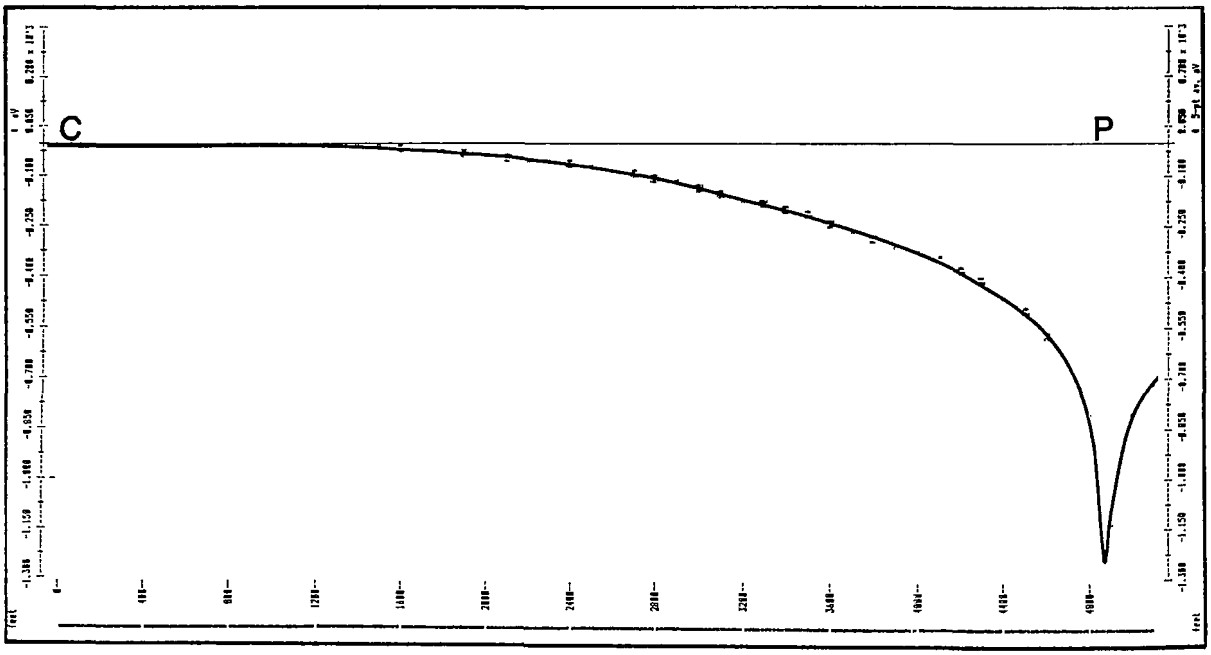


Figure 2. Potential profile between Parker Cave (C), Kentucky and a pipeline (P), having cathodic protection, 1.5km distant. Voltage at the pipeline measured 1.2V more negative than that at the cave.

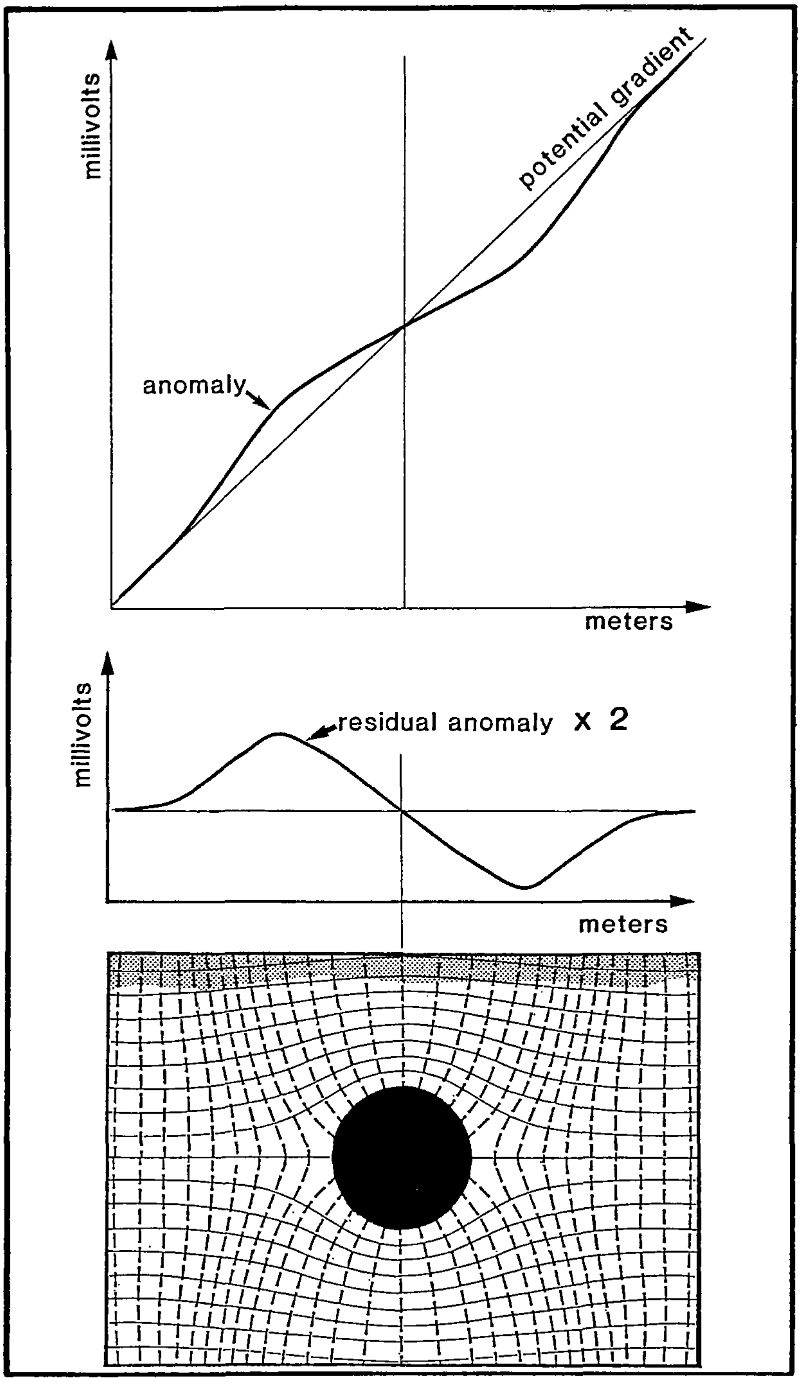


Figure 3. Effect of a cylindrical void in a horizontal electric field. Here, current lines are solid, while potentials are dashed. The effect on the surface potential is a sinuous anomaly superimposed on the electric field gradient.

that is not acquired from this paper or from a short course; it is the result of long, grueling days under adverse field conditions, numerous repeatings of data, and acute observations of the immediate environment, with attention to possible current sources. In the absence of such a commitment on the part of the operator, the instrumentation becomes little more than a dowsing rod employed for a bit of Sunday exercize.

Electrical refraction around a void

In the environment of an electric field gradient, as commonly occurs around a pipeline containing anti-corrosion devices (Figure 2), a component of the NP response of a void can be attributed to refraction of current lines around the opening (Figure 3). While profiles measured over Parker Cave, Kentucky clearly exhibited the effect of a pipeline more than a kilometer distant, their contribution to the *cave anomaly* was not detectable (Figure 4)(Lange & Quinlan, 1988).

During the course of that same field project, however, a profile was run over a railroad tunnel in the town of Park City, where a strong potential gradient can be attributed to nearby gas mains. The result is a sinuous anomaly over the tunnel, made more evident after removing the linear gradient (Figure 5). In this case, we evidently are dealing also with a filtration component that emphasizes the anomaly over the void and displaces the crossover to the right. The profuse dripping and ice formations in the tunnel walls testify to considerable filtration through the thin roof of the structure. In general, however, the refractive component of cavern anomalies is small, and is confined to the immediate vicinity of pipelines or other urban disturbances.

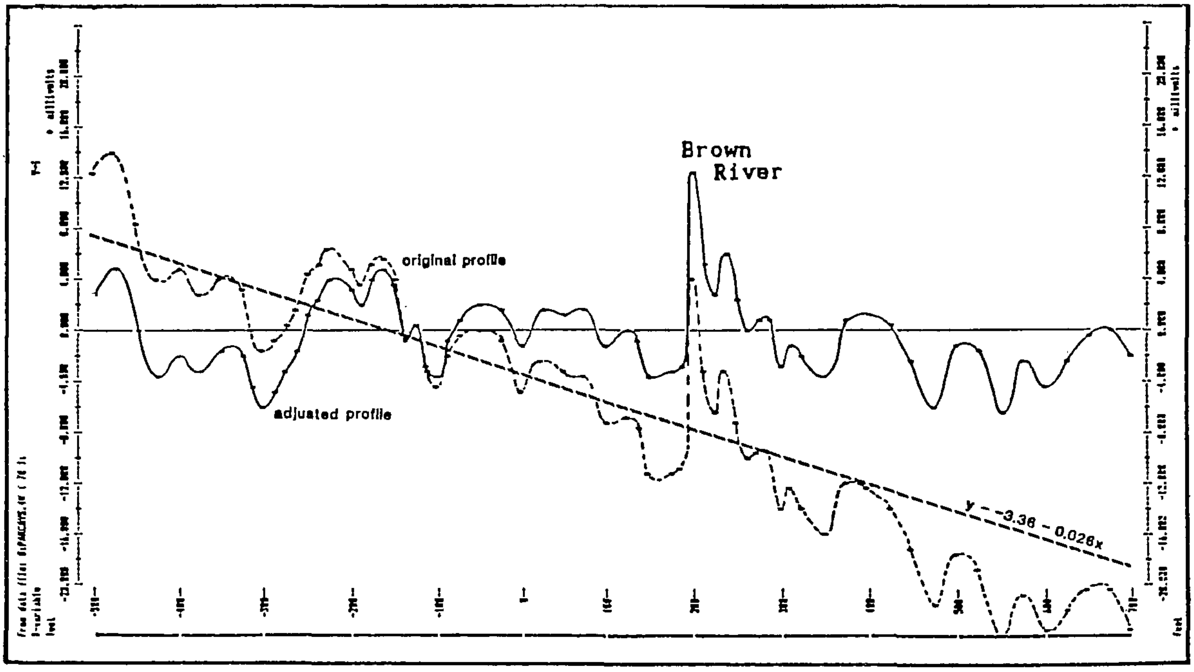


Figure 4. An NP profile (dashed) over Brown River as affected by the potential gradient of the nearby pipeline (cf., Figure 2). The solid curve is the residual anomaly after removing the pipeline linear trend. Though noisy, the anomaly is typical of that observed over a stream cave, wherein a peak (here positive) is flanked by lesser excursions of opposite polarity (negative). These results were obtained during January 1988 (Lange & Quinlan, 1988). See also Figure 9.

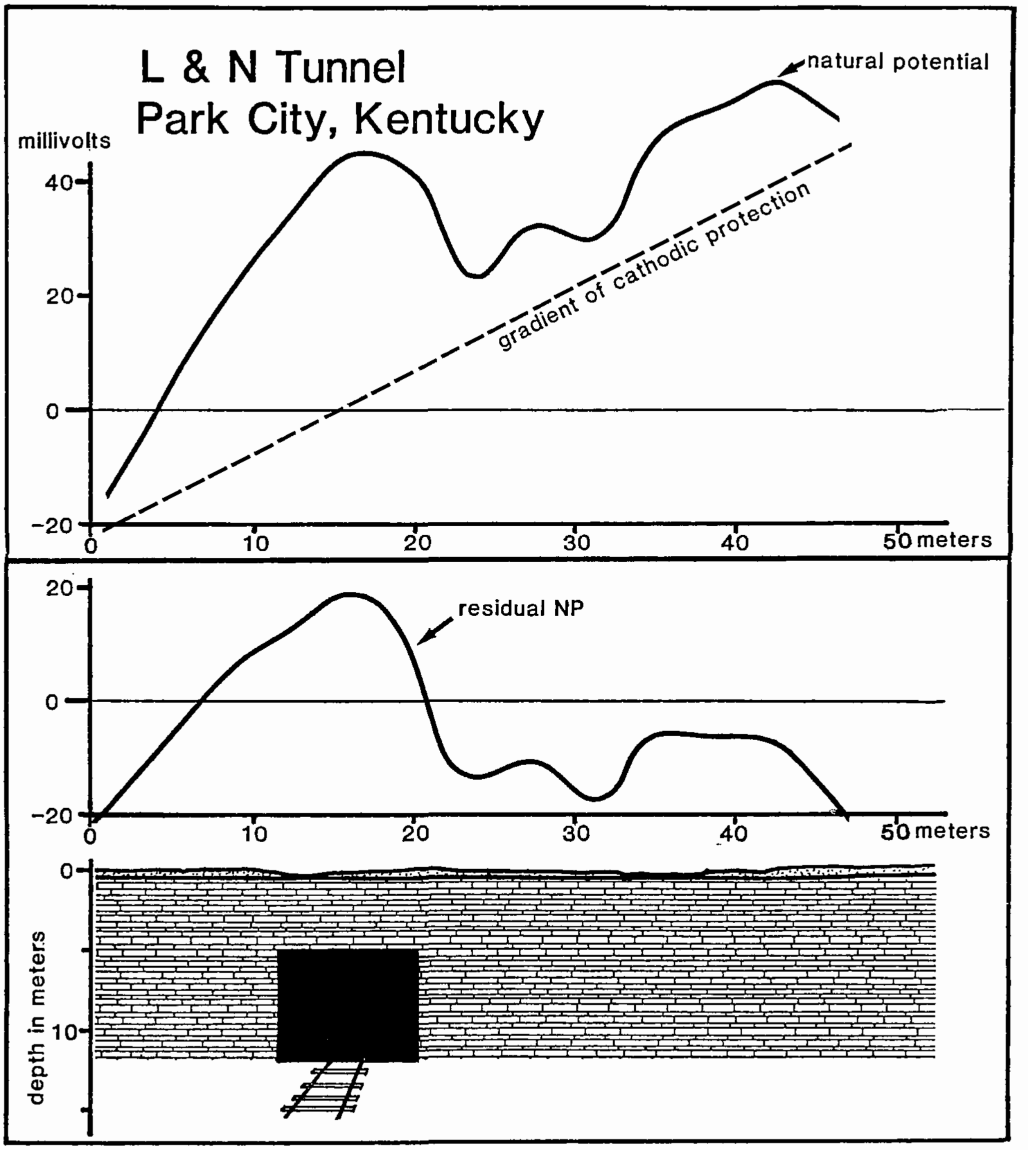


Figure 5. Potential profile measured over the Louisville & Nashville railroad tunnel, Park City, Kentucky. Electric currents from nearby gas mains refract around the tunnel, producing a sinuous anomaly superimposed on the artifical electric field. Filtration through the roof is visibly evident and can produce an additional NP component peaking over the tunnel.

Infiltration over a cave

Localized infiltration can occur over a cave passage owing to the enhanced permeability brought about by the presence of the void and associated fractures. Underground this is evidenced by excessive drippage and discharge from cave walls and by actively forming speleothems--stalactites, stalagmites, etc. The result on the surface can be an NP anomaly, peaking (either positive or negative) over the locus of maximum filtration, which, on a hillside, need not necessarily occur directly over the cave axis. The attendant fringe effects, of opposite polarity, may or may not be evident in the data. According to Scherer & Ernstson (1986), the polarity of an infiltration response in carbonate rock can be dependent on the pH of the electrolyte; thus, in the case of basic water, negative anomalies can be expected; and with acidic solutions, positive anomalies. Many field measurements both above and below ground will be necessary in order to test this polarity assertion under natural conditions. In any event, we have observed both positive as well as negative anomalies over cave passages showing evidence of active filtration (*V.* also Ishido & Mizutani, 1981).

Natural Bridge Caverns, Texas, presents strong negative as well as positive expressions over different portions of the system. Negative anomalies appear over the recently discovered west region of the cave;

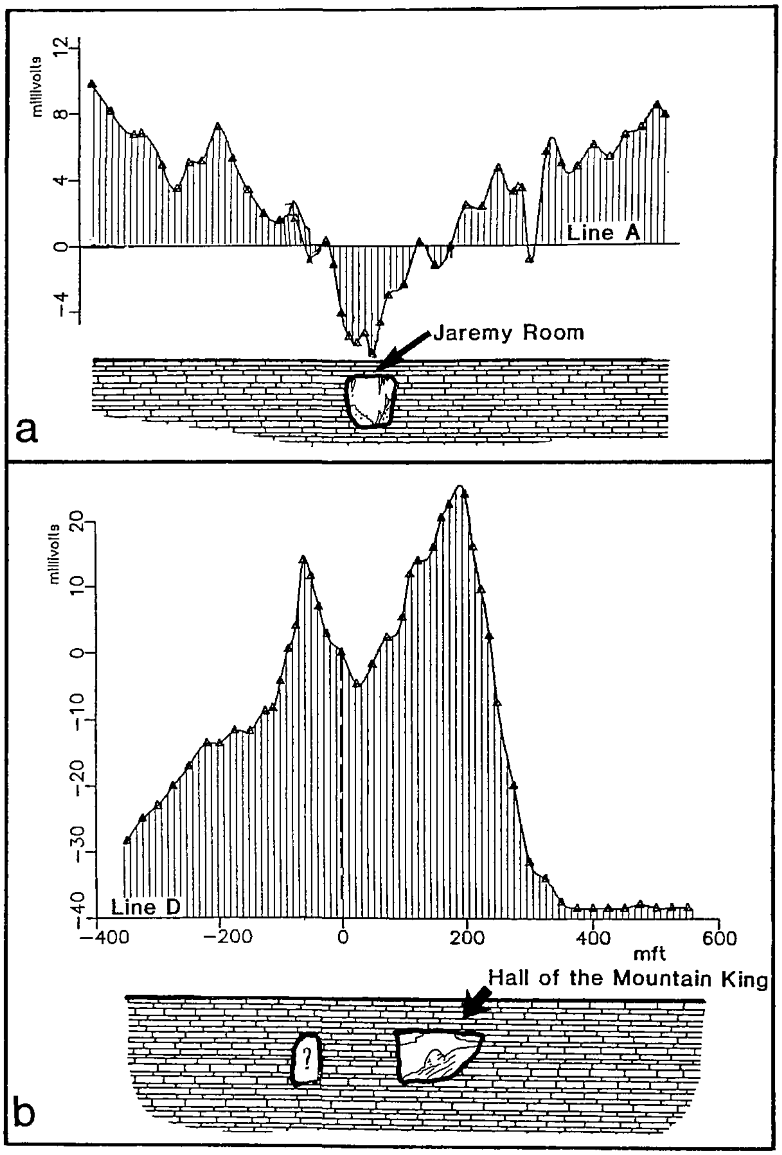


Figure 6. Filtration anomalies over different regions of Natural Bridge Caverns, Texas. **a)** Sharp lows occur in profiles over the recently discovered Jaremy Room, where downward vadose water is presently evident underground. **b)** A 60mV positive anomaly over the largest chamber of the caverns is believed to be the result of water moving from the cave depths towards the surface under capillary flow. A second, undiscovered passage is deduced from the lesser peak.

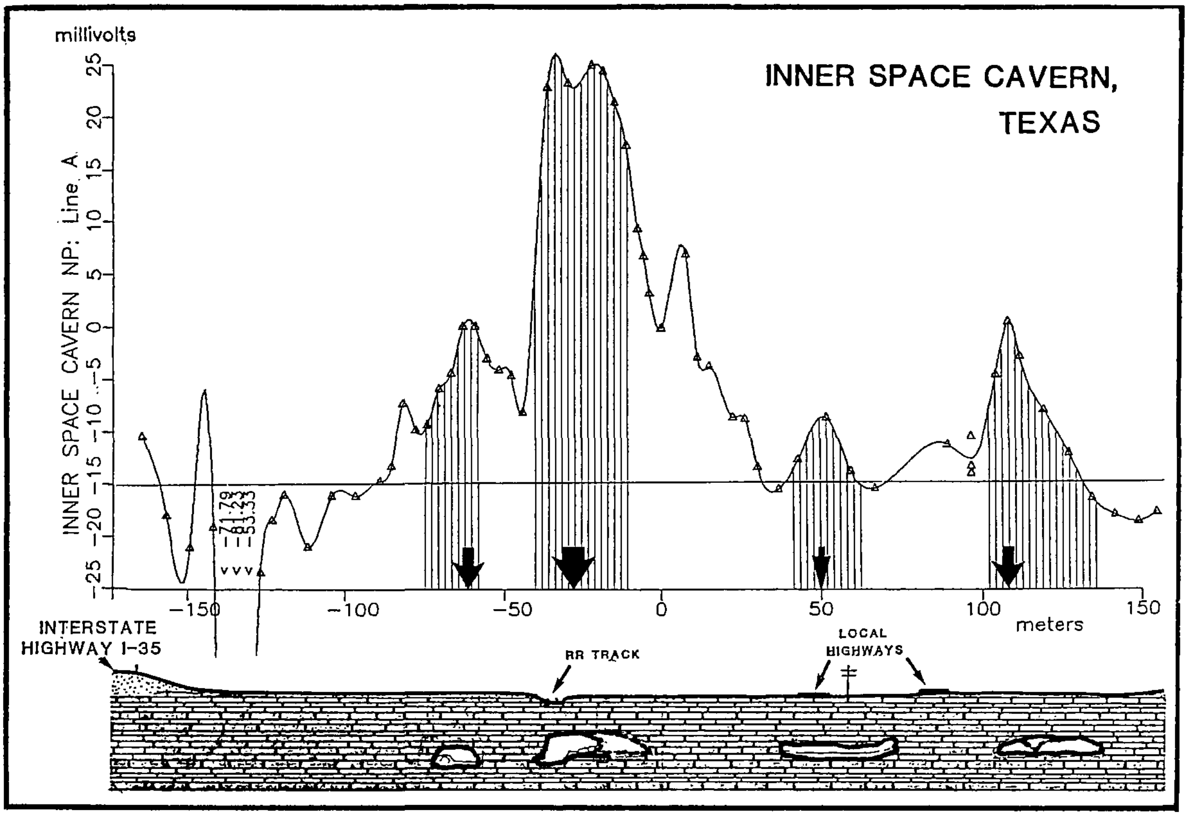


Figure 7. Positive NP anomalies corresponding to passages in Inner Space Cavern, Texas. The expressions are typical of cave anomalies to be expected from downward filtration of acidic water. (The extreme negative anomaly alongside the highway is due to a buried pipeline.)

while the 60mV positive anomaly in the tour portion of the cavern occurs over a relatively dry cave ceiling (Figure 6). One logical explanation is that the negative expressions over the wet Jaremy Room are the effect of filtration; while the positives over the Hall of the Mountain King arise from upward water movement (See the section on capillarity, below).

Inner Space Cavern was discovered during test drilling for highway construction at Georgetown, Texas. Presently, an interstate highway junction, a railroad line and city roads pass over the tourist cavern. A test line run over four typical cave passages produced four corresponding and definitive positive anomalies (Figure 7). Though drippage underground exhibited a pH of 7.0 at the time, the management reports that drip water is generally acidic, a fact that agrees with the rule that infiltrating acidic water produces positive anomalies.

The preceding Texas examples illustrate the effects of infiltration over shallow, individual cave passages. When we proceed to deeper systems--50 to 100m or more--we can expect to lose resolution of particular passages, but observe instead the compound expression of the "cave-as-a-whole" (Figure 1). This is the case at Jewel Cave in the Black Hills, whose lower levels occur at depths greater than 120m. The result is that 93% of the labyrinth crossed by the NP profiles were electrically low; that is, they were negative relative to the surrounding country rock. Measurements of drippage in the cave yielded pH's of 8.3 to 8.4, definitely basic (Bakalowicz et

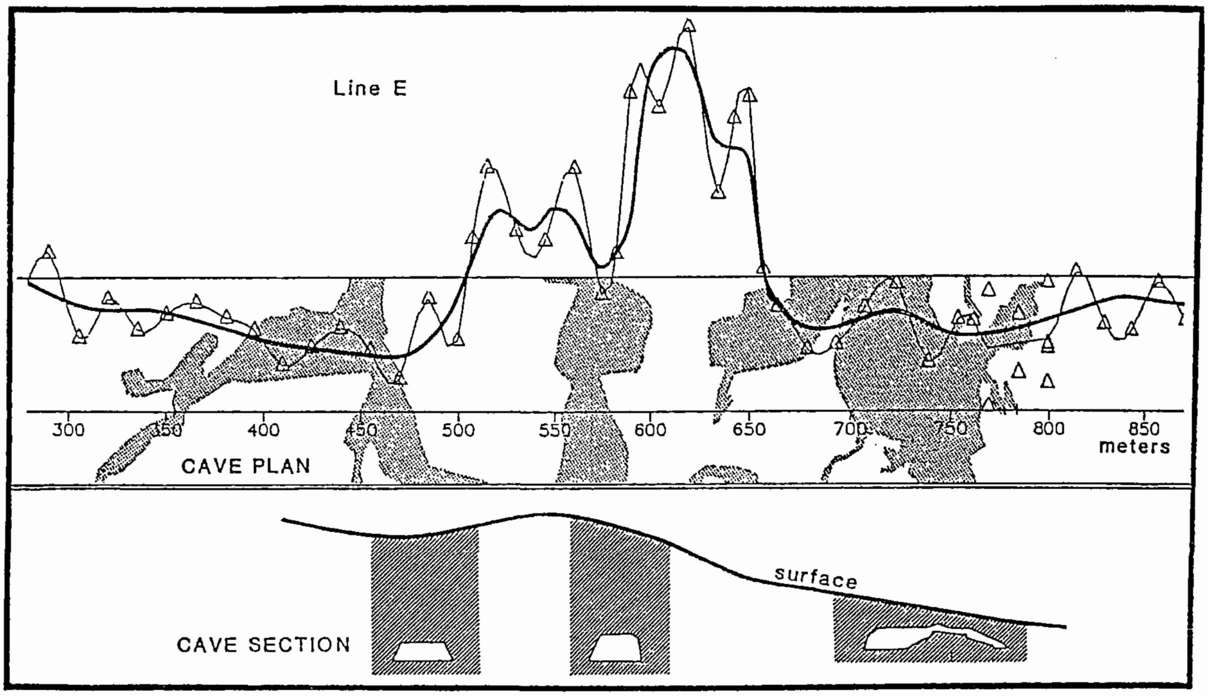


Figure 8. NP response over the main passages of Kartchner Caverns State Park, Arizona. The anomaly here is a compound feature consisting of a double positive flanked by lows, interpreted to be the result of upward migration of water from the very moist cave environment. (From Arizona Conservation Projects, Inc., funded by the Arizona State Parks Department).

al., 1987), and are commensurate with the idea of *downward* moving water, though flow in places can be directed horizontally by local aquitards.

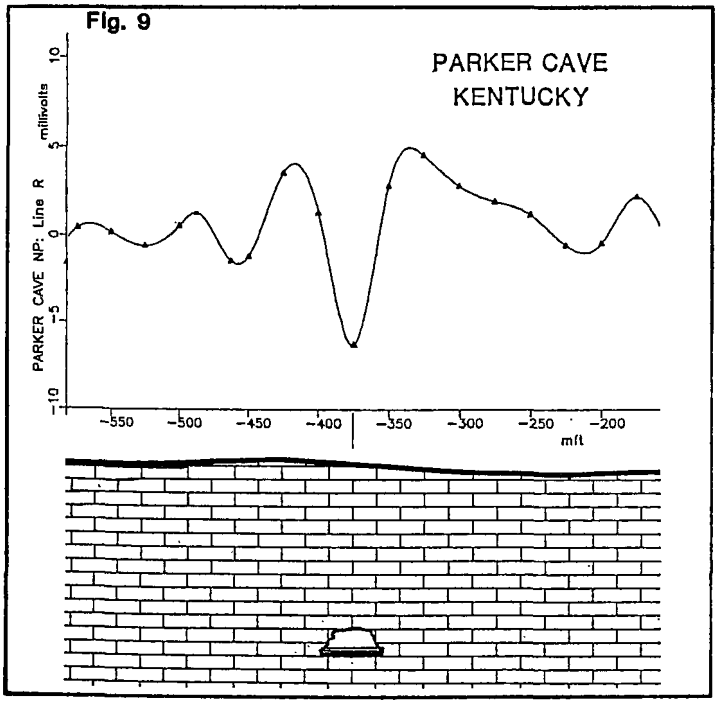
Upward water movement by capillary action

During a dry summer in the arid environment of Kartchner Caverns State Park of southeast Arizona, one might have difficulty imagining any appreciable downward filtration into the parched rock surface; in fact, even during the occasional thundershower, runoff flows rapidly from the rock outcrop towards the alluviated arroyos. Bulk resistivities measured in the massive carbonate over the cavern exceeded 1000Ωm, compared with about 400Ωm in the vicinity of Parker Cave, Kentucky (Lange, et al., 1990). Nevertheless, 25 to 50m below the surface, the cavern is very wet. Its floors are muddy, speleothems are actively forming and portions of the cave flood, following major storms.

Natural-potential anomalies over Kartchner Caverns showed both positive as well as negative expressions. We see typical negative filtration anomalies over the arroyos containing likely underflow; however, on the carbonate outcrop over the main portions of the cave system, the principal profile produced a sombrero-like, mainly positive anomaly (Figure 8). The NP signature was considerably more prominent than that obtained from gravity measurements along the same line. In the Big Room of the cave, ceilings proved to be consistently more positive than the floor. These observations all point toward upward wick-like movement of water by capillary action via interstices in the cave walls and ceiling, transporting water upward from some likely underlying reservoir towards the surface where evapotranspiration likely predominates over any downward filtration. Capillary effects on natural potential have previously been suspected by Hoover (1976), and the result of evapotranspiration observed in German forests by Ernstson & Scherer (1986), though not over caves. The phenomenon may likewise account for the large positive anomalies seen over the main chambers of Natural Bridge Caverns (Figure 6) and Sonora Caverns in Texas.

Electrokinetics of cave streams

In a stream of an alluvial environment, we can expect some leakage into the banks and the stream bottom. Likewise in a *porous* carbonate, such as the Floridian Ocala limestone, we foresee partial water movement into or out of the cave walls in addition to the main flow within the tube. In a *dense* carbonate medium, on the other hand, flow is primarily confined to the conduit. Thus, if we are to see an NP anomaly crosswise to a cave stream, it must come about from flow along the axis of the passage, where the mobile ions give rise to an electric convection current. The return current through the surrounding rock walls comprise a rather distributed conduction current. The result to be expected at the surface is an axial



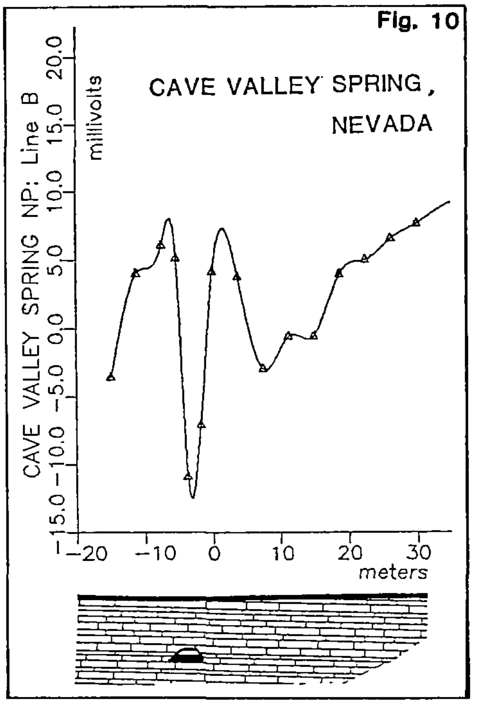


Figure 9. NP profile along a road, crossing over Brown River of Parker Cave, Kentucky, recorded in June 1991. This is a typical M-shaped negative anomaly to be expected over underground streams, and contrasts in polarity with the W-shaped positive observed farther downstream in January 1988 (cf., Figure 4).

Figure 10. Sharp NP stream-type anomaly recorded over a small limestone spring in Cave Valley, Nevada.

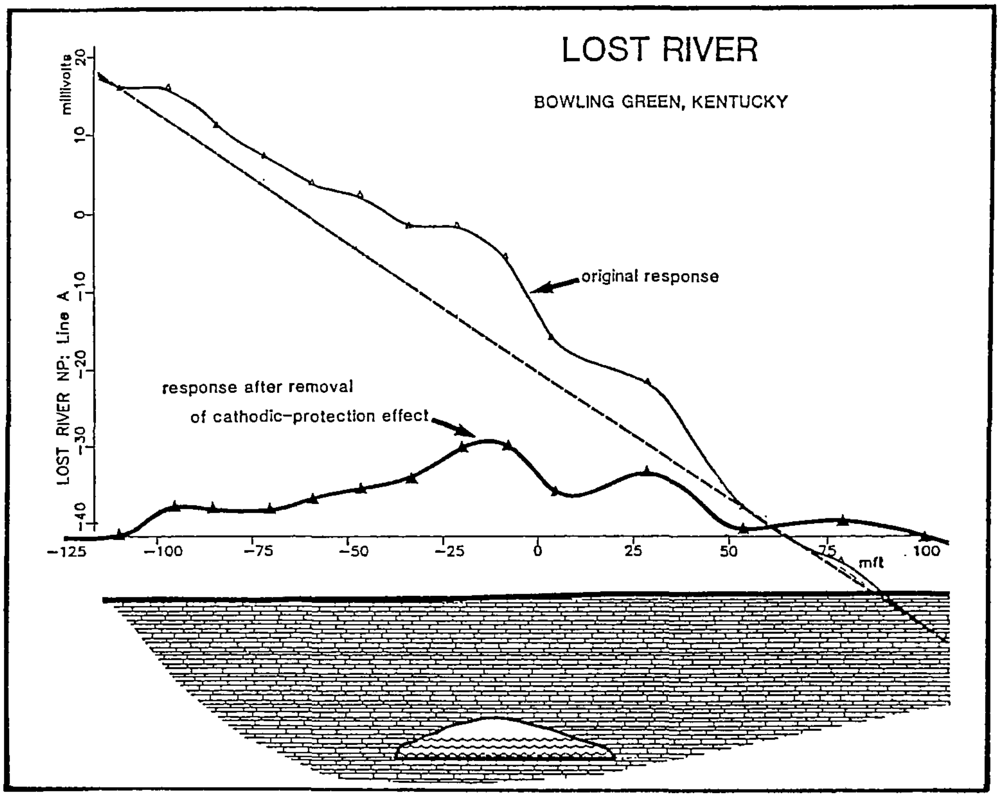


Figure 11. Positive response over the cave passage near the discharge of Lost River, Kentucky. The cave here was almost tube-full, so that its cross-section is drawn schematically rather than from survey data.

peak corresponding to the underlying stream, and parallel limbs of opposite polarity and lower amplitude on either side. Here again, we encounter the sombrero-type anomaly; though the hat may be right side up or upside down. Thus under certain conditions, stream anomalies may become more positive in the downstream direction, so that the discharge end, or rise, will measure positive relative to the upstream end, or swallet. On the other hand, under a different chemistry, the reverse may occur; that is, the downstream end could become increasingly negative. Somewhere about halfway, no anomaly may be observed.

Over natural cave streams we observe both positive and negative sombrero anomalies. Brown River of Parker Cave, Kentucky exhibited a positive expression (Figure 4) during January 1988, yet in June 1991 a negative expression was measured along the roadside a hundred meters or so farther upstream (Figure 9). While the main cavern in Cave Valley, Nevada produced a 60mV positive sombrero anomaly (Lange & Quinlan, 1988), a nearby karst spring yielded a sharp negative (Figure 10). Other positive expressions have been recorded over Lost River Cave, Kentucky and Lost River, Indiana (Figures 11 & 12, resp.). Meanwhile, in the Texas hill country, we see remarkably sharp positive M-shaped anomalies over passages of Honey Creek Water Cave. at locations separated by several kilometers (Figures 13,14), yet another part of the system appeared negative, while still another channel produced no measurable response.

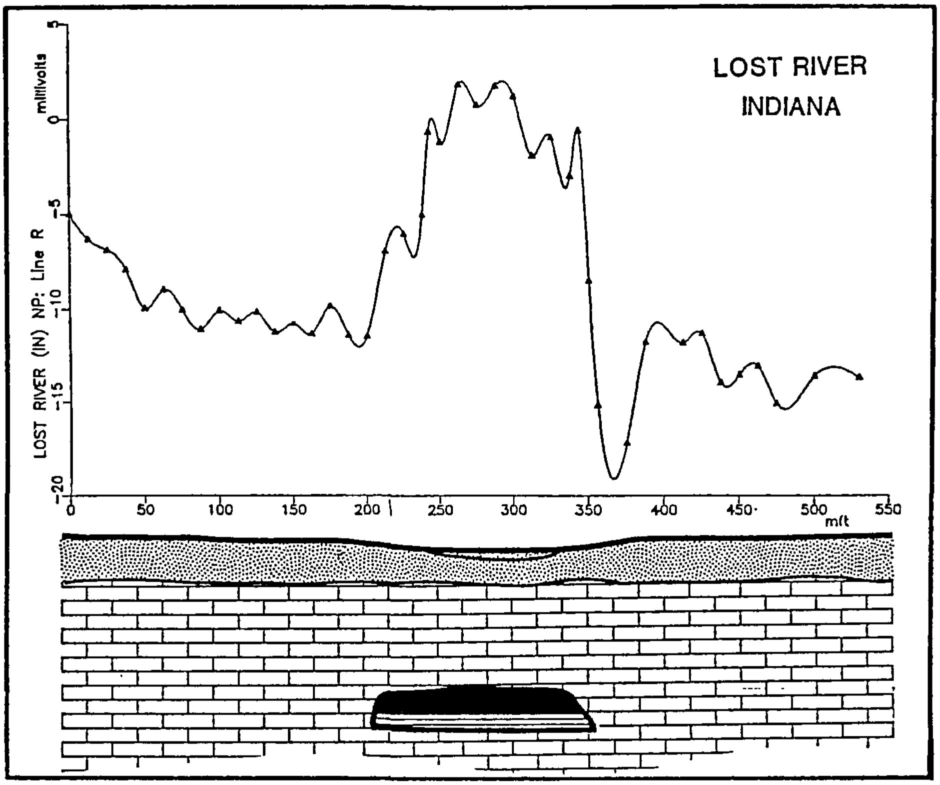


Figure 12. Natural potential along Highway 37 crossing the dry channel of Lost River, Indiana. The cave depth and location are estimated.

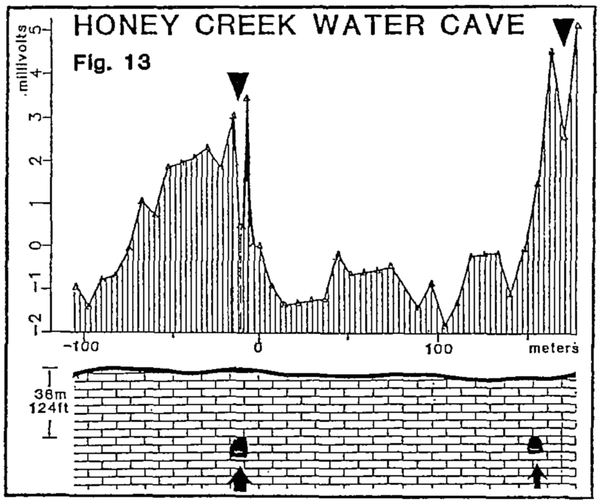
To properly test the electrokinetic stream mechanism, we will require one or more systematic surveys over a discrete mapped stream passage--with measurements conducted at one time of year. Only in this way, will we come to fully confirm the surface potential distribution both transverse to the stream as well as in the axial direction.

Summary

Evidences of four different mechanisms postulated for generating natural-potential expressions over caves and karst conduits have been observed in the field. The mechanisms are the following:

1) Refraction of ambient d.c. currents around a void;
2) Downward filtration through the cave roof and walls;
3) Upward migration of flow from a moist cave environment towards the surface by capillary action; and
4) Electrokinetic effect of flowing water in tubes.

We do not expect these mechanisms necessarily to be operating independently; for example, a stream cave having air space can also receive drippage from the surface. The resulting surface anomaly is then a compound effect. Similar anomalies can be expected from artificial tunnels and aqueducts. Furthermore, other natural processes may be acting as well to produce NP effects; so that in time, other mechanisms will need to be evaluated.



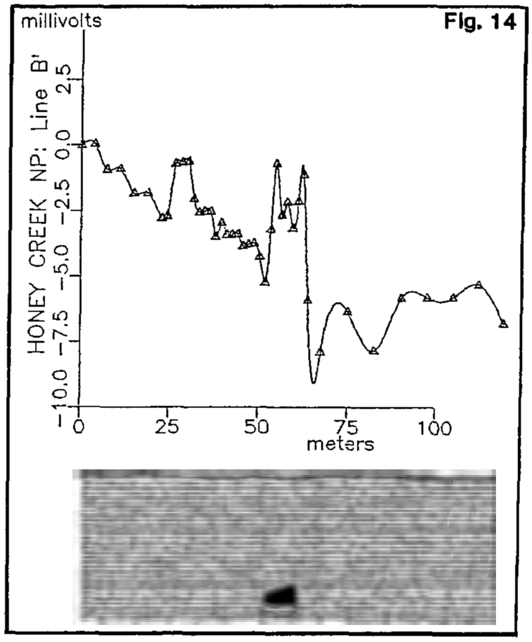


Figure 13. Particularly sharp stream-type anomalies are observed over segments of Honey Creek Water Cave, Texas. Here each conduit is expressed as a sharp M-shaped positive.

Figure 14. Detail of a stream-type anomaly over another segment of Honey Creek Water Cave, remote from those of the preceding figure.

Finally, in addressing an audience of practicing karst hydrogeologists, we need not emphasize the environmental and engineering applications of an effective tracking system for caves and karst conduits. We have already tested the technique for the siting of monitor wells in karst (Lange & Quinlan, 1988); for the delineation of subsurface flowpaths and the mapping of drawdown and recovery patterns associated with pumping (Lange, 1991). Tests have been made for the purpose of avoiding construction failures and environmental misadventures over cave systems, and for avoiding the possible contamination of a major cave resource by nearby drilling. Perhaps the demonstration most needed at this point is the mapping by the natural-potential method of a karst pathway connecting the endpoints of a successful dye trace. We can know where the dye goes in and where it comes out, but to track the intervening flowpath, we must rely on a remote sensing tool such as that of natural potential, in effect, a rapid and cost-effective method for solving environmental problems in karst.

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Lange & Kilty/Natural-Potential Responses

1. You summarized by saying that natural potential (NP) can detect both shallow caves and deep caves, ranging from small to large. Relative to this statement, I ask:
   A. What are the limits on cave size detectable as depth increases?
   B. Under what geologic conditions might the NP method be inappropriate?

2. Please discuss the following as related to the sensitivity of your instrumentation:
   A. What is the sensitivity and noise level of the instrumentation?
   B. What are the electrode noises?
   C. What are the typical levels of field noise?

3. Do you think that NP is the best geophysical technique for identifying the presence of a cave? What are the advantages and disadvantages of the technique?

Lange & Kilty/Natural-Potential Responses: Answers to questions

1a. We have detected caves to depths as great as 250m (Western Borehole of Lechuguilla Cave); however, conduits less than about 3m are unlikely to be resolvable below about 50m. We must keep in mind that, in the case of infiltration into an empty void, much of the response may be coming from the region between the surface and the cave ceiling. Where the response derives from flowing water, its amplitude at the surface will depend on several factors; particularly flow velocity, velocity gradient, and the coupling coefficients (electrochemistry) between the carbonate solid and the electrolyte (water). Limits of detectability, therefore, will be dependent on the overall environment and flow conditions.

1b. We have not yet found situations in which the method is inappropriate; however, there are some in which resolving power is considerably reduced. Several examples are a) Sites having a very electrically-conductive overburden or wet shales near the surface; b) Hot, dry and rocky desert environments, where ground contact resistance is high; c) Frozen ground or ice and deep snow cover; d) Paved areas of concrete and asphalt. There are ways to solve these difficulties; for example, the drilling of small holes in the pavement; however, they generally require more time in the field, and, of course, increased cost.

2a. The instruments can be read to ±0.05 millivolts; although this sensitivity is seldom needed. Instrumentation noise level is less than that.

2b. Using copper-copper-sulfate electrodes, the reading changes by about -360µV per °C; however, by maintaining both the base and roving electrodes in similar environments, in terms of shade and sun, this effect can be minimized. Inherent electrode noise has been measured in the lab as <3µV in one hour.

2c. Field noise varies greatly. Using lines less than about 500m in length, typical variations of 1 to 3mV are seen in pastoral environments. Telluric noise during severe magnetic storms can be as great as ±10mV for line lengths on the order of 500m in a granitic or carbonate terrane--a condition that may require suspending operations for an hour or more until the noise abates. In suburban settings, local artificial noise is typically around ±4mV; whereas in an urban locale having electric trains, as in Staten Island, N.Y., sudden jumps of 25mV or more are encountered. The latter problem can be surmounted by utilizing an averaging, signal-processing receiver which samples as many as 16 readings at each position.

3. While NP anomalies can be produced by a variety of processes; it has proved to be a definitive indicator of caves and karst streams in about 95% of the controlled cases studied. The problem rests in distinguishing cave anomalies from those produced by metal conductors, lithologic changes, localized mineralization, etc.--all of which can produce anomalies. A companion survey using an EM device can help in eliminating these. As a rule the NP cave-resolving power seems to be much higher than other geophysical methods, such as gravity, resistivity, and seismic; furthermore, it is the only standard

geophysical tool that responds to the *movement* of water rather than its mere presence. The method is more economical to employ than the aforementioned and is considerably faster. A good strategy would involve running an NP reconnaissance survey, followed by NP detailing of significant anomalies and, finally, localized gravity traverses over the most promising targets as a confirming technique. The main disadvantages lie in the interpretation; that is, in distinguishing cave anomalies from other types and in evaluating these in terms of depths and dimensions.

WATER TEMPERATURE VARIATION AT SPRINGS IN THE KNOX GROUP NEAR OAK RIDGE, TENNESSEE

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Oak Ridge, Tennessee

ABSTRACT

Water temperatures were measured during 1991 at several springs draining Cambrian-Ordovician age, Knox Group dolomites near Oak Ridge, Tennessee. The coefficients of variation for temperature at each spring indicate, there are two different groups of springs, one group with essentially constant temperatures, and a second group with more variable temperatures. The maximum temperature occurrence at the springs is observed to lag at least 50 days behind the reported maximum mean air temperature for Oak Ridge for 1991 (NOAA, 1991). Although the amplitude of temperature variation is different for the two groups of springs, the lag-times are similar. The lag-time is related to the thermal diffusivity of the system, and the amplitude of temperature variation is related to the relaxation length or temperature damping factor within the system. The only model that best explains the flow system is one where there are different proportions of quick-flow and slow-flow components at each spring. This is because the groups of springs have similar lag-times but different temperature variability and all drain the same aquifer. Analysis of carefully collected temperature data can yield meaningful insights into aquifer mechanics.

INTRODUCTION

Ground-water flow in karst aquifers was characterized by Shuster and White (1971) as a continuum, the two end members of which are, diffuse flow or conduit flow. Smart and Hobbs (1986) further defined the characteristics as a three-dimensional continuum where, the relations between recharge, ranged from concentrated to dispersed, aquifer storage, ranged from low to high, unsaturated to permanently saturated, and flow ranged from conduit-flow to diffuse-flow.

For systems dominated by conduit flow, hydrograph response to rainfall is relatively rapid and return to base-flow conditions also rapid. For diffuse-flow systems, hydrograph response to rainfall is slow and return to baseflow likewise. Chemograph response to precipitation for a mostly conduit-flow-dominated system is flashy and highly variable and for mostly diffuse-flow dominated systems is much less so. Ford and Williams (1989, p. 204-210) provide an overview of how spring chemograph interpretation has been used to study karst aquifers. Quinlan and Ewers (1985) and Quinlan et al., (1991) suggest that several measurements of specific conductance at a spring before, during, and after several storm events is a simple way to determine whether that spring is fed by mostly conduit-flow, mostly diffuse-flow, or is a mixed type, partly conduit-flow and partly diffuse-flow. Water temperature measurements have also been used as a method of assessing the flow characteristics of a spring by, for example, Meiman et al., (1988).

When karst aquifers are characterized using the three-dimensional cube model of Smart and Hobbs (1986) and as modified by Quinlan et al., (1991, Fig. 1) involving the independent affects of recharge, storage, and flow, many different systems can be conceptualized. At one extreme a predominantly fast-flow (conduit) system would have discrete recharge via swallets, low aquifer storage, and a relatively straight connection with the spring, a diffuse-flow component would also exist in such systems with the exception of systems with unique characteristics. An example of one such unique system would be a conduit-flow system where the diffuse-flow component would be essentially negligible (e.g., Castleguard Cave, British Columbia, Canada) where during the winter months and most of the year, the diffuse-flow component could be considered to be effectively zero (Smart and Ford, 1986). At the opposite end member of the continuum a purely slow-flow (diffuse-flow) system, would have dispersed recharge, high aquifer storage, and a complicated pathway to the spring.

It is known that the pulse-through time (hydraulic response) is often different from the flow-through time in phreatic systems (Ford and Williams 1989, p. 227). If they are similar then, as Brown (1972) concluded, open channel (vadose) conditions probably exist within the system. When observing water temperature variation, arrival of a thermal signature at a spring is often associated with a lag-time in response to with interaction of the recharge-water thermal signature and the aquifer's ambient signature. If we consider the movement of a slug of recharge water carrying its temperature signature through a simple cave system, for heat exchange to occur by mostly conduction, the velocity of the water and the amount of surface area it is in contact with (the passage size) would control the rate at which heat is lost by the water and gained by the rock. So in a fast-flow system of large conduits, the temperature signature would probably be transmitted as far as the spring. In a slow-flow system, the thermal signature of the recharge water would be damped or thermally equilibrated with the rock before reaching the spring.

Studies of temperature variation of spring waters in the Pennines of Yorkshire, Great Britain, have used temperature measurements to estimate ground-water contributions to surface flow (Pitty, 1976, 1979). He also noted temperature lag-times at springs, where the maximum temperature at the spring occurred after the maximum mean air temperature or the maximum measured air temperature.

Roy and Benderitter (1986) studied temperature and discharge variation during a year at a spring in the Loire Valley of France; there is a maximum temperature lag-time of between 90 and 120 days. Aley (1970) measured the temperature of a small Ozark spring and observed a temperature lag-time of about 28 days. Meiman et al., (1988) collected temperature data during a study of flood pulse through a karst aquifer at Mammoth Cave, Kentucky, and in this mostly conduit-flow dominated system short-term lag-times were measured in hours.

STUDY AREA

The study area, a valley cutting through the Cambrian-Ordovician Knox Group dolomites near Oak Ridge, Tennessee, is in the Valley and Ridge province in the southern Appalachians. The geology of the valley is summarized in Figure 1. Older clastic rocks of the Rome Formation form Pine Ridge, to the northwest, with overlying interbedded clastic and carbonate rocks of the Conasauga Group outcropping to the southeast, with the Maynardville Limestone forming the floor of Bear Creek Valley. Further to the southeast, Chestnut Ridge consists of the Knox Group dolomites with a repetition of the sequence which lies to the northwest cropping out to the southeast, because of faulting. Throughout the area, beds dip between 38 and 50 degrees southeast (McMaster, 1963).

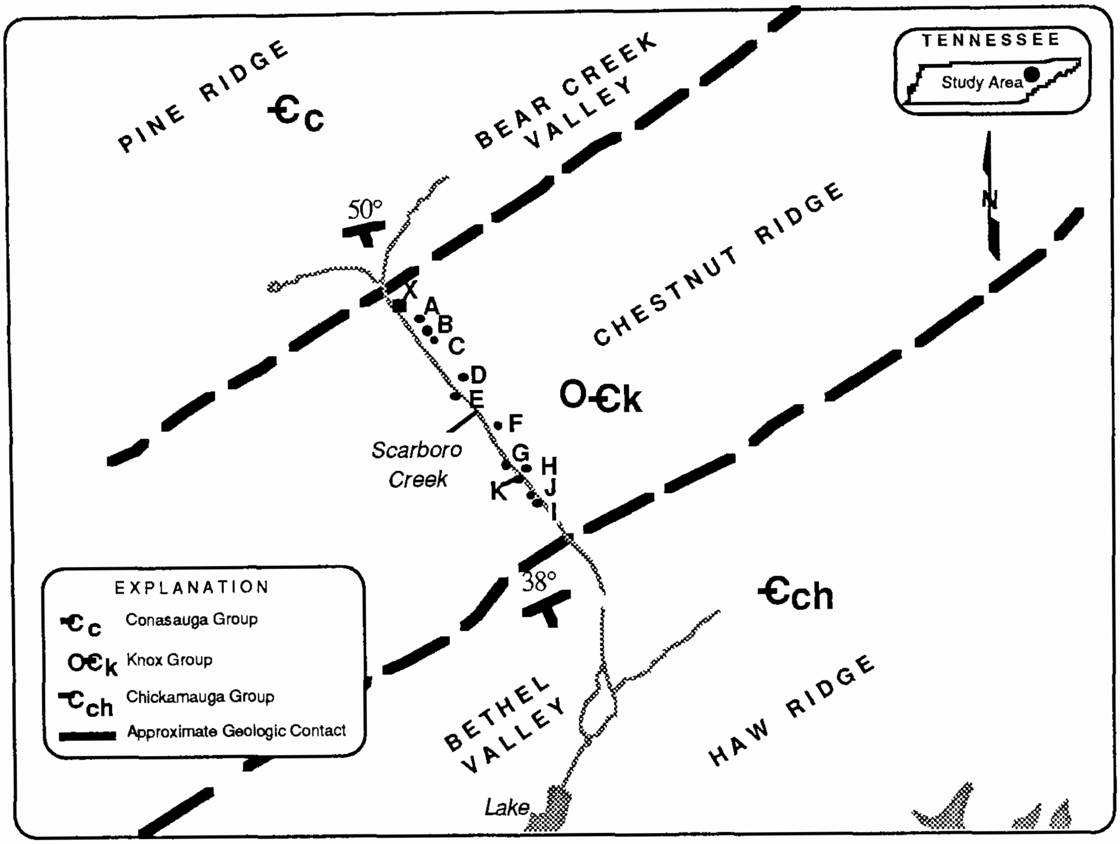

Figure 1. Geologic Map of the Study Area
(after McMaster, 1963)

Each carbonate sequence is karstified. Dolines occur on the Knox Group dolomites near the crests of the ridges, and springs emerge on the dip slopes, some of which sink again into swallets, finally emerging at lower elevations. Springs also emerge from along the strike in valleys indented into the ridge. The entire carbonate sequence is mantled with 15 to 20 meters of residual soil that contains numerous chert fragments. Vegetation on the ridge consists of a combination of hardwoods and pines with some deadfall and covering of leaf litter.

Scarboro Creek is a perennial stream that flows southeastward cutting through Knox Group dolomites on Chestnut Ridge. The springs included in this study all emerge near the channel of Scarboro Creek; on Figure 1 they are labelled A through K and a surface-water measuring point is labelled X. Although these are not the only springs that have been observed to flow near Scarboro Creek during flood conditions, the study included six springs that drain the ridge from the east, A, B, C, D, F, and H, and five springs drain the ridge from the west, E, G, I, J, and K. These springs have discharges estimated to be between 1 and 30 liters per second. Springs A, D, and H have the highest discharges and every spring except Spring F is perennial.

TEMPERATURE MEASUREMENTS

Each temperature measurement was made with a digital thermometer capable of measuring to 0.1 degrees Celsius. The accuracy of the thermometer was checked with a mercury reference thermometer every few weeks. Spring temperatures were measured generally every 2 or 3 days. Each measurement was always made at the same location in the spring pool or flow. Although a coarser resolution has been obtained in this work when compared to the resolution that would be possible with a thermometer measuring continuously or to 0.01 degrees, and using a data-logger as described by Meiman et al., (1988) a large data set for many different springs, all of which drain the same aquifer has been compiled. This allows a comparison of temperature variation for different springs and a relative assessment of likely flow geometries to be made.

RESULTS

Table 1 shows the mean temperatures, their standard deviations, and their coefficients of variation. The table represents data collected from January 1991 and summarizes more than 1,500 measurements.

Table 1. Sample Means, Sample Standard Deviations (SD), and Coefficients of Variation (CV)

SPRING	MEAN	SD	CV
A	14.2	0.1	0.9
C	13.9	0.2	1.2
D	13.9	0.2	1.2
E	14.5	0.2	1.2
H	13.9	0.2	1.2
J	14.6	0.3	2.0
B	14.1	0.3	2.4
I	14.6	0.8	5.4
G	15.3	1.4	9.3
K	16.1	1.5	9.5
F	14.4	1.7	11.7

BASED ON MORE THAN 1,500 DATA POINTS

The coefficients of variation (CV = Std. deviation x 100/mean) show that the springs seem to fall naturally into two groups. One group includes springs A, C, D, E, H, and J. In this group the coefficients of variation of temperature are less than 2% and in most cases close to 1%. The mean temperatures range from 13.9 to 14.6 degrees and standard deviations range from 0.1 to 0.3. It is interesting to note that Springs C, D, and H have practically identical statistics, are all located on the east side of Scarboro Creek, but are not located close to or adjacent to each other. Spring J is located on the west side Scarboro Creek. Although the eleven springs seem to fall into two groups, they are probably members of a continuum that would probably be more obvious if additional springs in the Oak Ridge area were to be sampled.

Spring A has the most constant temperature of all the springs. The spring outlet is a large pool and the flow discharges through a what appears to be a conduit partially blocked with an alluvial plug. The spring outlet is located beneath a large tree and flow is from obliquely along the strike along a bedding plane. Only in the absolutely wettest conditions has the flow from Spring A become turbid.

Spring C is located about 25 meters down the valley from Spring A. This spring emerges from along a bedding plane in a small steephead (steephead = a spring-sapped, steepened area) near the spring outlet. Spring C has a mean temperature of 13.9 degrees and a standard deviation of 0.2. The flow is always clear, even after the most prolonged precipitation.

Spring D is located about 250 meters downstream and is a relatively large spring. The perennial flow emerges from a pool, the alluviated outlet, but in wet conditions flow also emerges from a blow-hole nearby, which itself then discharges to the pool. Spring D shows very little temperature variation and becomes slightly turbid in the wettest conditions. The temperature of this spring responded in a similar way to Springs A and C to heavy, prolonged precipitation events, and flow at the blowhole becomes slightly turbid during floods.

Spring E is located about 40 meters downstream from Spring D, emerging from a steephead on the west bank of Scarboro Creek. The variation of temperature at Spring E is also small, a standard deviation of 0.2, but the mean temperature is higher at 14.5 degrees. The spring water does not become turbid in even the wettest conditions.

Spring H is located on the east side of the creek, about 150 meters downstream. Flow emerges from what appears to be the direction of strike, from a small conduit. This spring has a constant temperature with a coefficient of variation of 1.2% and a mean temperature of 13.9 degrees. The water remained clear throughout most of the year and became only slightly turbid during and after heavy storms.

Spring J is located about 150 meters downstream from Spring H, on the west side of the channel. Spring J emerges from a steephead very close to the creek. The coefficient of variation of this spring is 2% and the mean temperature 14.6 degrees. The temperature at this spring became highly variable during the summer and is thought to have been affected by the initial flow from an orifice near the channel of Scarboro creek becoming plugged and thereafter mixing with flow from Scarboro Creek, but eventually flow from the orifice resuming.

Another group of springs with more variable temperatures includes Springs B, I, G, and K. Coefficients of variation are greater than 2% but less than 10%. Mean temperatures of this group ranges from 14.1 to 16.2 degrees. The standard deviations range from 0.3 to 1.5 degrees.

Spring B is located within a few meters of Spring C, has a very small discharge, and its flow is always clear. Spring B has a coefficient of variation of 2.4 and its temperature variation is quite dissimilar to nearby Spring C. Springs B and C each emerge from steepheads on the bank of Scarboro Creek.

Spring I is located about 10 meters downstream from Spring J, also on the west side of the creek. Flow is from a steephead along the strike at a bedding plane surface. The coefficient of variation is 5.4% and the mean temperature of the spring is 14.6 degrees. Flow from Spring I has never been observed to become turbid.

Spring G is on the west side of the creek about 60 meters downstream of Spring F. The flow joins with the flow from Scarboro Creek beneath a tree, but actually originates in a collapsed area a few meters away. Spring G has the second highest mean temperature of all the springs, 15.3 degrees, its coefficient of variation is also high, 9.2 %. The flow emerges from a steephead and was clear throughout 1991.

Spring K is a shallow pool located on the west side of the Scarboro Creek, about 20 meters downstream of Spring H. It was not originally recognized as a spring but, after observing the pool for several days, it was realized that it could be a spring as it was always clear and persisted as a pool even when the tributary upstream of it dried up. The temperature of the pool was also sometimes similar to the springs nearby and was considerably different from the creek. The mean temperature of the pool is 16.0 degrees. The coefficient of variation is high at 9.9%. The only flow that obviously enters the pool is from the small tributary, in flood conditions; there is a constant amount of water in the pool at all times and when the water in

Scarboro Creek is turbid, the pool remains clear. During the wettest of conditions, overland flow occurs between the closest section of Scarboro Creek and the pool.

Spring F emerges from beneath a tree about 50 meters downstream of Spring E. Spring F has a mean temperature of 14.4 degrees, a sample standard deviation of 1.7, and a coefficient of variation of 11.7%. Water flows from beneath a small bedding plane. Flow ceases during dry conditions.

Thermographs representing measurements at the eleven springs from January 1991 through early December 1991 are combined and shown in Figures 2 and 3.

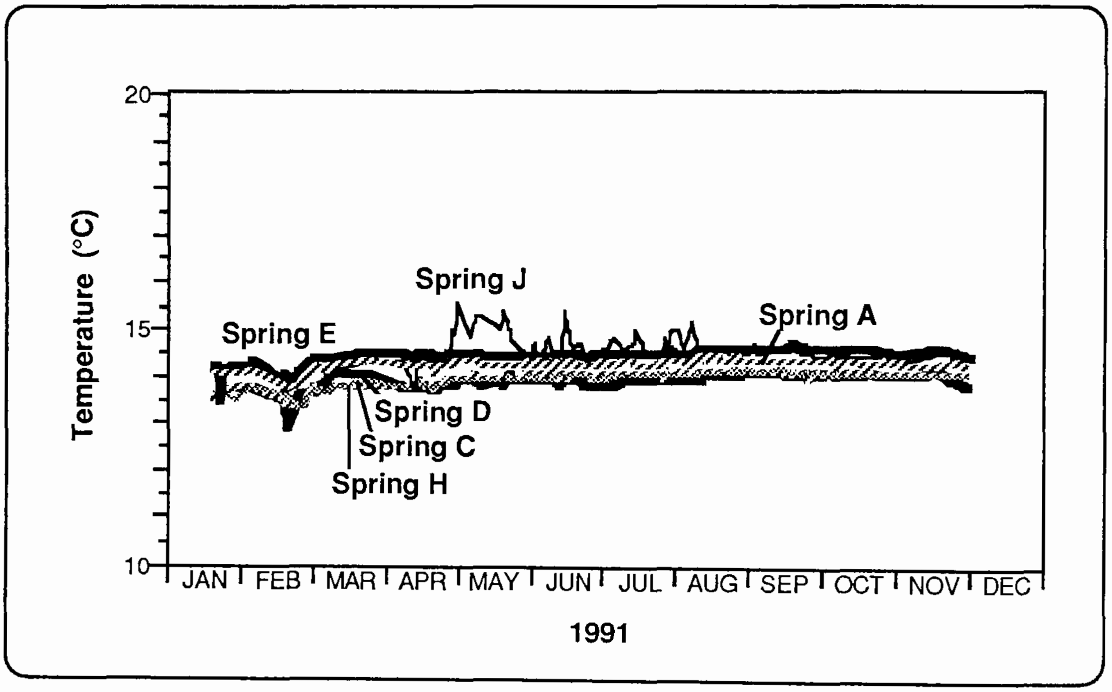

Figure 2. Thermograph of Constant-Temperature Springs

DISCUSSION

The CVs of temperature in this study suggest that all the springs are fed by a mostly diffuse-flow component. This is consistant with the CVs of specific conductance at four springs A, D, E, and H, which also suggest that the springs are fed by mostly diffuse-flow (CV less than 5%) or a mixture of conduit and diffuse-flow (5%<CV<10%), based on the suggested CV limits of Quinlan and Ewers (1985) and Quinlan et al., (1991). Observations that none of the springs become turbid in flood also suggest, again based on suggestions by Quinlan and Ewers (1985) that they are fed by mostly diffuse-flow components.

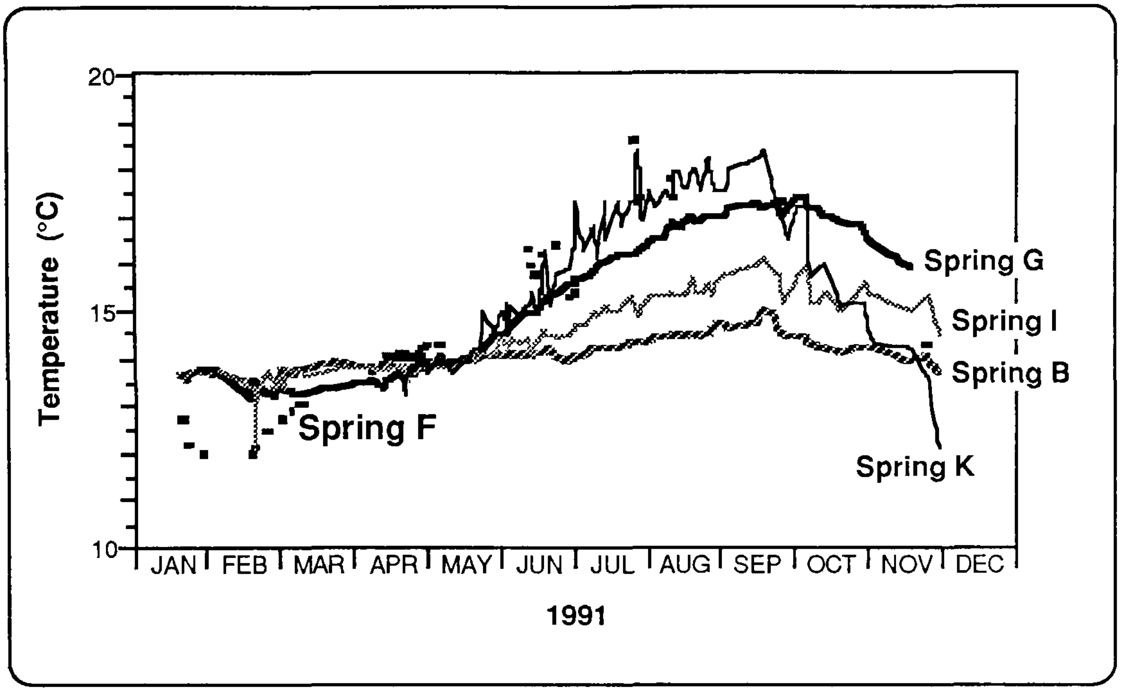

Figure 3. Thermograph of Variable-Temperature Springs (Figures 2 and 3 are plotted with the same temperature scale)

However, there are similarities and differences in the data for all the springs. The similarities are mostly related to the lag-times, and the differences are mostly related to the amplitude of temperature variation through the year. It is important to point out that, from observations in the study and the CVs of temperature, there is no evidence of any relationship (similarity or difference) that is related to relative locations of springs, or the amount of their flow, or the physical appearance of the spring outlets.

The temperature lag-time, when it occurs, provides information about the thermal diffusivity of the system or the time required to displace the resident thermal signature of the bedrock. The thermal diffusivity of a substance can be defined as the thermal conductivity (cal/sec. cm. ^{0}C) divided by the product of the heat capacity (cal/g. ^{0}C) and the density (g/cm^3) (Oke, 1978, p. 39) which can be reduced to and expressed in units of cm^2/sec. The degree of thermal diffusivity is described as the depth of temperature penetration into the rock (or the system) of each pulse of recharge water temperature as it flows through the aquifer. Another way of relating to thermal diffusivity would be to recognize that it is a measure of how far each molecule of recharge water can "carry" its temperature signature into and through the system. A purely conduit-flow system would have a high thermal diffusivity and would result in a short lag-time. A purely diffuse-flow system would have a low thermal diffusivity and would result in a long lag-time.

Wigley and Brown (1976) describe a parameter called the relaxation length, which is associated with the decay of, or damping, of outside air temperature to an asymptotic (constant) value within a cave system. The relaxation length is related to the size of the passage and the velocity of the fluid moving through it, and its roughness factor. More simply put, the relaxation length is a distance inside a cave that the outside air temperature effect can be detected (or measured). Figure 4 shows a representative thermograph for a variable-temperature spring (Spring G), and a curve representing the daily mean air temperature for Oak Ridge in 1991 (NOAA, 1991). The daily mean air temperature curve has been smoothed by using a seven-point moving-average procedure for each data point. The thermograph shows that maximum temperatures at the springs occurred about 50 days after the maximum daily mean air temperature for Oak Ridge (NOAA, 1991). Fifty days is thought to be more reasonable than 415 days (50 + 365); there are several possible temperature oscillation cycles as explained by Lange (1954).

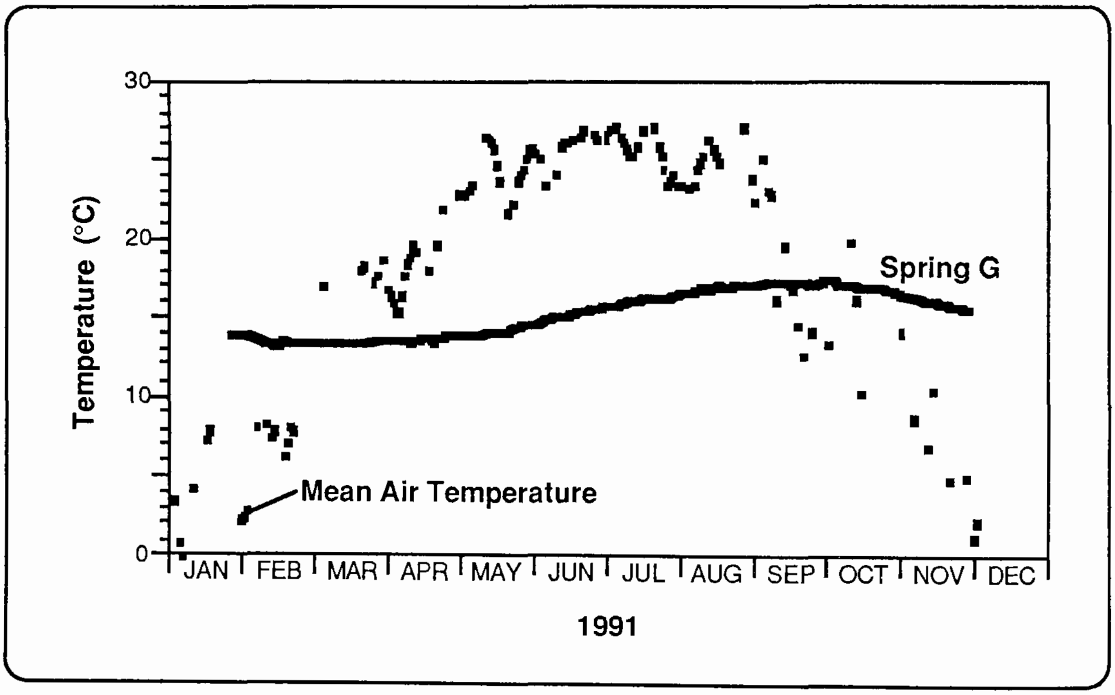

Figure 4. Thermograph for Spring G Compared with Mean Air Temperature for Oak Ridge, 1991

From the relaxation length theory of Wigley and Brown, assuming a system with flow pathways of roughly equal dimensions, a spring draining a conduit-flow system (a quick-flow system) would transmit a temperature pulse deeper through the system, possibly to the spring; there would be a long relaxation length. In the opposite way, a diffuse-flow system (a slow-flow system) would not transmit the temperature pulse as far, or would have a short relaxation length; the temperature pulse would decay or be damped within the system before reaching the spring.

As previously stated, the lag-times for all the springs are essentially the same (with the exception of the "noisy" data for Spring J); as Figure 5 shows, the constant temperature springs have lag-times similar to the variable-temperature springs, but, the amplitude of temperature variation is smaller. Interpretation of various conceptual models of the aquifer system suggests that the differences in the data could be related to the systems being either conduit flow or diffuse flow systems.

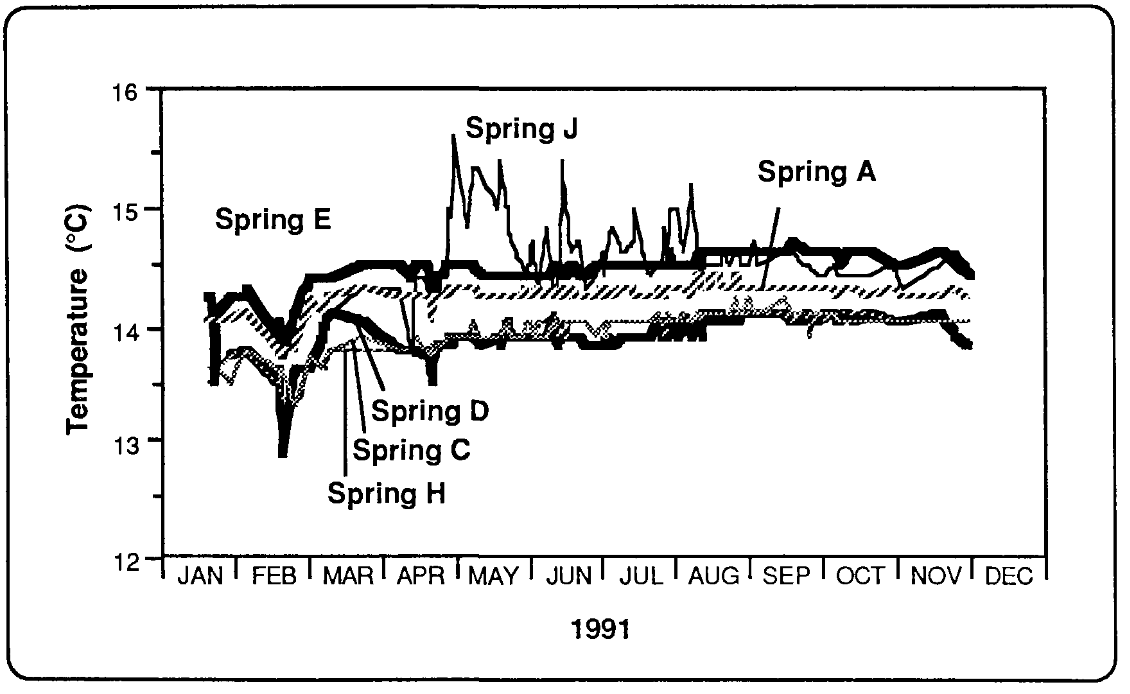

Figure 5. Thermagraph of Constant-Temperature Springs (Showing expansion, for clarity of temperature scale used in Figures 2 and 3)

Relating thermal diffusivity to the lag-times, if the systems are different (i.e., conduit-flow or diffuse-flow) then the thermal diffusivities of each system should be different, so lag-times would be different also. The data do not fit such a model, the lag-times are similar, so this hypothesis must be rejected. Also, the lack of significant turbidity in the springs after rain makes it obvious that they are predominantly diffuse-flow springs.

A second hypothesis applies the theory of relaxation length. If it were valid here, then the differences in each system could be explained by the different ratios of aquifer length versus relaxation length. With such a hypothesis, the variable-temperature springs would be closer to the recharge area as compared to the constant-temperature springs. But as previously stated, thermal diffusivity is a time-and distance related unit, so a shorter aquifer length would cause a change in thermal diffusivity, and would probably result in different lag-times and different ratios between aquifer length and relaxation length. Again, the variation in the data cannot be explained by this hypothesis.

A third hypothesis involves mixing a slow-flow (diffuse-flow, and at a constant temperature) component with different amounts of a quick-flow (conduit-flow - variable-temperature) component. This mixture would always include a diffuse-flow component with a variable proportion of the conduit-flow component present within each system. A mixture of different proportions of the two components could produce a certain temperature at the output, and still maintain acceptable thermal diffusivity and relaxation length conditions for the system.

Multiple temperature measurements at several springs can aid in characterizing flow at a spring, or within an aquifer, but is it likely that whatever impression is gained about flow at them, i.e., whether a spring or an aquifer is dominated by mostly diffuse flow or mostly conduit flow, the flow is probably always a mixture of quick-flow and slow-flow components. The response of a spring or springs to rainfall or temperature change is only the end-product of responses of the whole system and is only a gross generalization of the sum of those responses; they include individual and different responses to recharge, storage, and flow. It is highly likely that no system is predominantly conduit-flow or diffuse-flow. Rather, all systems are different forms of mixed-flow systems.

Temperature variation should be studied at springs because it is a physical parameter and would be expected to vary somewhat differently to specific conductance, which is a chemical parameter, but would provide equally useful information and would give insight into aquifer mechanics. Measurements of specific conductance were not made concurrently with the 1991 temperature measurements and in retrospect they should have been. The temperature measurements and study of their variation should be supplemented with concurrent measurements of specific conductivity and study of its variation. Such data are now being collected for all the springs to allow comparison between the CV of temperature with that of specific conductance. These additional data will be available from the author in mid 1992. However, it can be seen from the (temperature) data described in this paper that if many measurements are made at several springs that drain the same or adjacent geologic unit(s) and/or aquifer(s), such information can yield valuable information about the characteristics of that aquifer. Such data that can not be obtained from studying temperature variation at only one spring, and probably can not be obtained at randomly located wells, in part because of scale differences between monitoring-well effectiveness and typical karst aquifer organization, as suggested by Quinlan and Ewers (1985) and Ford and Williams (1989, p. 210-211).

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BIOGRAPHICAL SKETCH

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WATER TEMPERATURE VARIATION AT SPRINGS IN THE KNOX GROUP NEAR OAK RIDGE, TENNESSEE

By Gareth J. Davies

Do you think the time lags you observed are actual travel times? If so, why? If not, why not?

The lag times are not travel times. Rather, they provide information about the thermal diffusivity of the system. The lag-time is an average for the centroid of the whole basin that drains to a particular spring. A dye-trace provides information about travel time from one point to another, and is not necessarily representative of the whole basin. If a lag-time happens to coincide with a travel time inferred from a dye-trace, e.g., Pitty (1976, 1979) the relationship is just that, a coincidence. Two different processes are being monitored.

THE EFFECT OF PETROLEUM HYDROCARBONS ON THE SORPTION OF FLUORESCENT DYES BY ACTIVATED COCONUT CHARCOAL

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ABSTRACT

The use of fluorescent dyes to delineate ground water flow-routes and basin boundaries in karst aquifers has dramatically increased in the past decade. Dye tracing techniques have been used in conjunction with contaminant assessment investigations to determine the extent of soil and ground water contamination caused by leaking underground storage tanks.

Fluorescein and rhodamine WT, the two most common tracers, are passively collected on activated coconut charcoal at discharge points or in wells . The presence of petroleum hydrocarbons in ground water may retard the sorptive efficiency of the charcoal by competing for carbon attachment sites or by reducing the actual dye concentration due to chemical alteration of the dye.

Solutions were prepared from ground water containing known concentrations of benzene, toluene, ethylbenzene and xylenes (BTEX) and known concentrations of both fluorescein and rhodamine WT. Half of the solutions were analyzed for florescence intensity particular to these dyes, using a scanning spectrofluorophotometer. The second half of the solutions were used to immerse a sample of activated coconut charcoal (6-14 mesh), mimicking a passive detector. After a contact period of 48 hours, the charcoal was eluted with a mixture of 42% 1-propanol, 38% ammonia hydroxide, and 20% distilled water for a period of 24 hours. The elutriant was also analyzed for dye concentration intensity characteristics of the dyes using a scanning spectrofluorophotometer. Comparison of the fluorescent intensity in the dye/hydrocarbon solutions and the elutriants enabled a determination of sorption retardation and chemical alteration caused by the presence of the petroleum hydrocarbons.

INTRODUCTION

Fluorescent dyes are commonly used tracing agents for hydrogeologic investigations. Heightened environmental awareness and expanding government regulations have increased the need to conduct dye tracing investigations in contaminated hydrologic systems.

Rhodamine WT (RWT) and fluorescein,two of the most commonly used dyes, are often employed in conjunction with passive activated charcoal detectors. Dye is introduced into a hydrologic system at an input point and is collected at discreet discharge points (springs) or wells. Historically, tracers have primarily been used to amass information concerning the flow velocity and basin boundary characteristics of free flow karst aquifers. Investigations were rarely conducted in response to the loss of organic contaminants. The need to conduct tracer investigations in the presence of petroleum hydrocarbon contamination has increased with the promulgation of CFR 40, Part 280, requiring hydrologic investigations in response to releases from underground storage tanks (UST's).

Activated charcoal is useful in tracer studies as it readily adsorbs organic molecules. However, the charcoal is indifferent to which organic it will sorb in the presence of multiple organic compounds. Rhodamine WT and fluorescein are organic compounds that adsorb readily to charcoal in a contaminant free environment. An environment riddled with organic contaminants may compromise the sorption efficiency of the charcoal with respect to the dyes. The result of a tracer study conducted in the presence of organic contaminants may be inaccurate because the charcoal may preferentially sorb petroleum hydrocarbons, rather than fluorescent dyes.

The presence of organics, particularly petroleum hydrocarbons, may also chemically alter RWT and fluorescein producing erroneous tracer results versus if the organics were not present. The purpose of this study was to assess the sorption capabilities of activated charcoal in the presence of light petroleum hydrocarbons (gasoline) and how hydrocarbons may possibly chemically alter the respective dyes.

METHODOLOGY

Representative ground water and liquid phase hydrocarbon (LPH) samples were collected from a gasoline contamination site in the coastal plain of North Carolina. Samples were immediately cooled to 4°C for shipment to a certified laboratory. Samples were analyzed according to EPA SW-846 Method 602 modified for benzene, toluene, ethylbenzene and total xylenes (BTEX). The samples contained varying concentrations of total BTEX as listed in Table 1. Although a sample was also collected from a monitoring well which contained floating LPH, its total BTEX concentration was not determined.

Table 1. Total BTEX concentrations (mg/L)

TOTAL BTEX CONCENTRATIONS	
BTEX Free	0.0
Low Dissolved	3.17
High Dissolved	9.95
LPH	132*

* Based on similar samples from monitoring wells containing LPH.

Representative ground water samples were split during collection. One was analyzed by a laboratory and the other was used to prepare the dye mixtures. Dye solutions of 50,000, 1,000, 1.0 and 0.5 ug/l were prepared from ground water samples and distilled water. The various dye solutions were split into two bottles. Half of the solution was transferred into a bottle and kept in total darkness, at a constant temperature of approximately 4° C. The other half of the solution was transferred into a bottle with a packet containing approximately 3.0 grams of activated charcoal and kept in total darkness, at a constant temperature of 4° C. After a period of 48 hours the charcoal packets were removed and shipped with the dye/ground water solutions for dye analysis.

Groundwater/dye solutions were analyzed on a Shimadzu scanning spectrofluorophotometer with respect to RWT and fluorescein. The 50,000 ug/L solution was not analyzed because its fluorescent intensity exceeded the capacity of the instrument. Results of the analyses are shown in Table 2.

Table 2. Concentration of rhodamine WT and fluorescein elutriant.

RHODAMINE SOLUTION

		(Concentration total BTEX (mg/L)			
Dye Concentration	Distilled Water	0.0	3.17	9.95	LPPH
500	0.988	0.898	0.946	0.892	0.916
0.5	0.006	0.005	0.005	0.004	0.007
0.25	0.003	0.002	0.003	0.003	0.004

FLUORESCEIN SOLUTION

Dye Concentration	Distilled Water	0.0	3.17	9.95	LPPH
500	0.772	0.313	0.063	0.073	0.081
0.5	0	0	0.001	0.001	0.001
0.25	0	0	0.001	0.001	0.001

Activated charcoal packets were placed in a mixture of 42% 1-propanol, 38% ammonia hydroxide and 20% distilled water for a period of 24 hours. The resultant solution, known as elutriant, was analyzed on the Shimadzu scanning spectrofluorophotometer with respect to RWT and fluorescein. Results of the analyses are included in Table 3.

Table 3. Concentration of rhodamine WT and fluorescein solutions.

RHODAMINE ELUTRIANT

		(Concentration total BTEX (mg/L)			
Dye Concentration	Distilled Water	0.0	3.17	9.95	LPPH
500	1.455	0.892	1.599	1.036	1.85
0.5	0.004	0.008	0.005	0.006	0.01
0.25	0.004	0.003	0.004	0.003	0.004

FLUORESCEIN ELUTRIANT

Dye Concentration	Distilled Water	0.0	3.17	9.95	LPPH
500	5.91	4.141	6.353	5.389	3.473
0.5	0.009	0.008	0.007	0.009	0.1
0.25	0.006	0.007	0.005	0.006	0.007

RESULTS AND DISCUSSION

The 500 ug/L dye concentration used in this study is a higher concentration than commonly occurs at aquifer discharge points. Dye concentration of initial solutions versus dye concentration measurements of resultant solutions were plotted as bar graphs for each BTEX value. Graphs are presented in Figures 1 through 4. Review of these figures indicate that at low dye solution concentrations (0.5 and 1.0 ug/L), resultant dye concentration is insignificant for both the dye solutions and for the elutriant. Measured dye concentration appears to be within the range of instrument fluctuation. Dye solutions at these chosen concentrations were not useful in determining the chemical alteration of the dye or the sorption capability of the charcoal. Concentration from elutriant though cannot be considered purely quantitative. Although care was taken to standardize charcoal volume, no two dye packets will have the same sorptive characteristics.

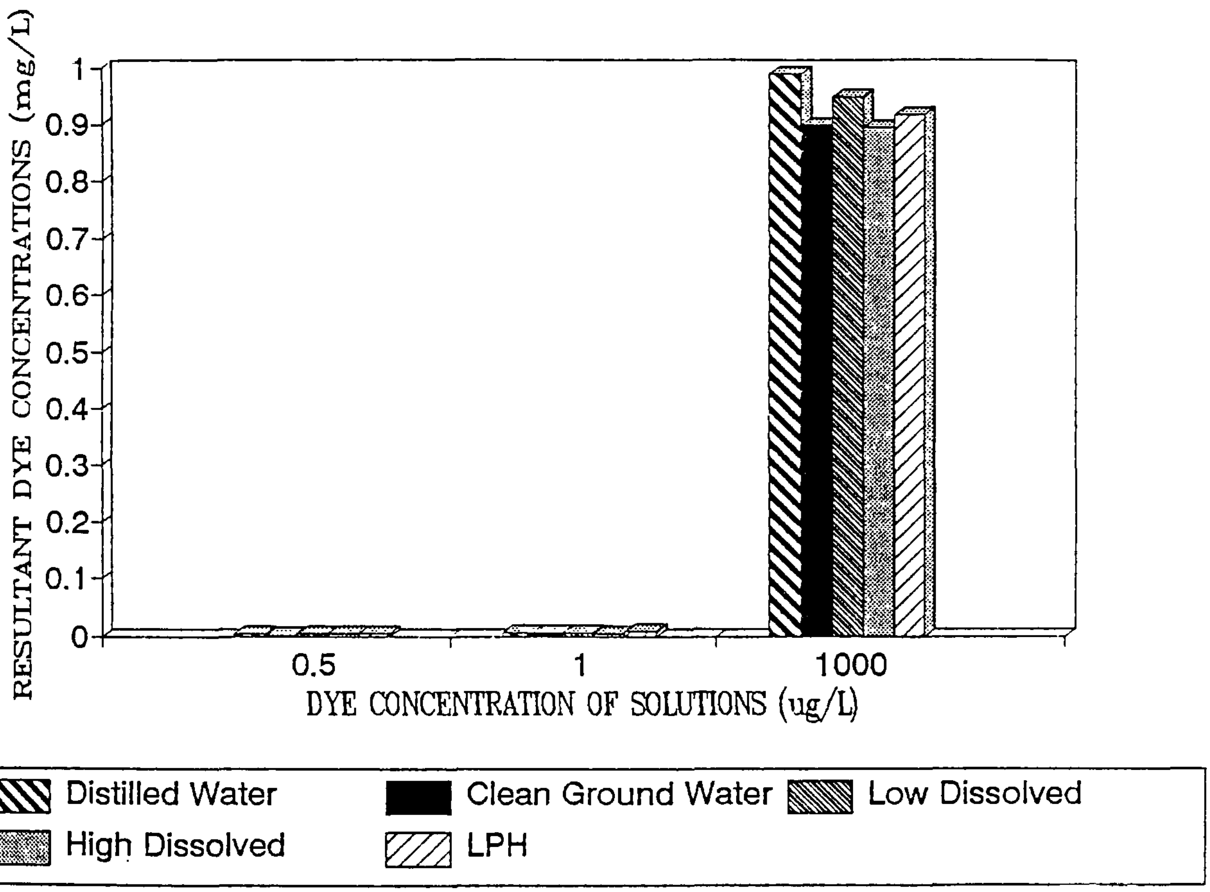

Figure 1. Resultant dye concentration versus dye concentration in prepared RWT solutions.

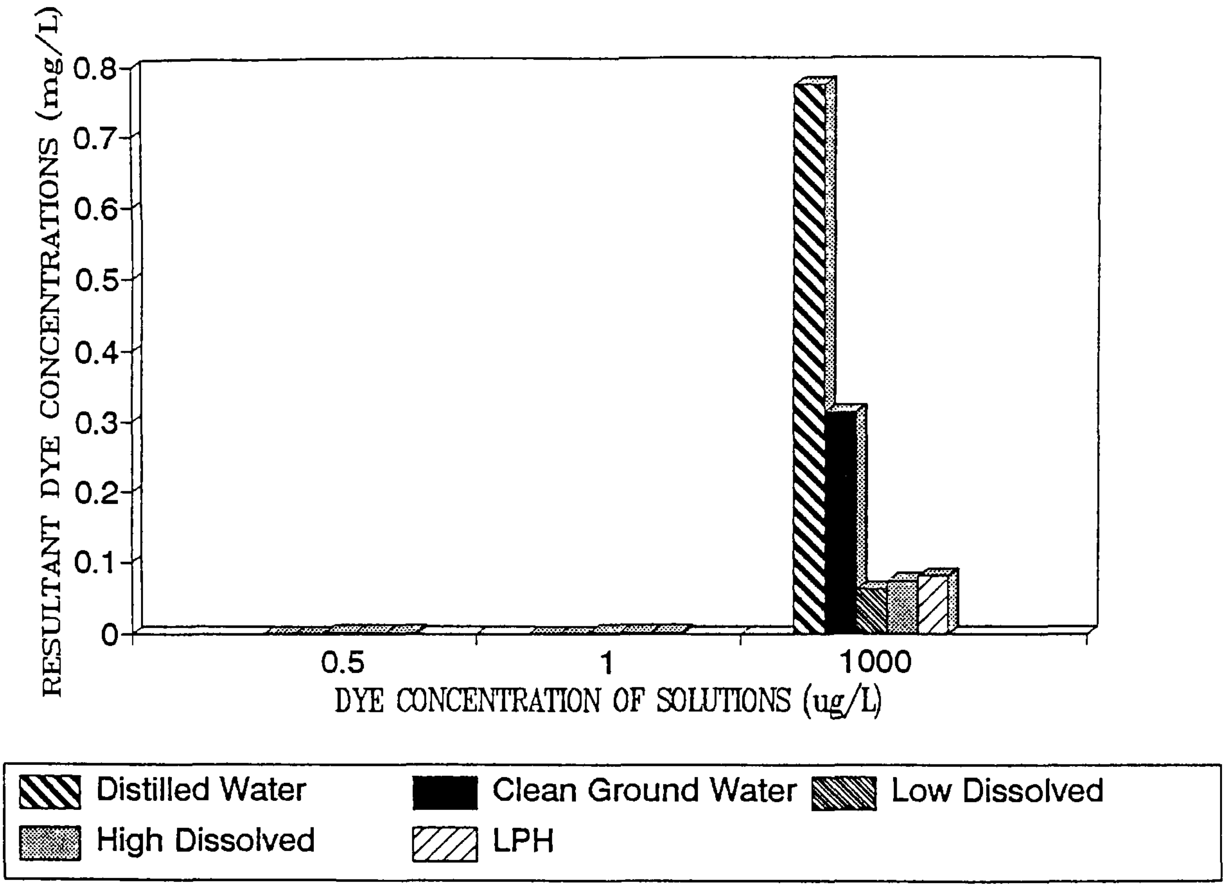

Figure 2. Resultant dye concentration in prepared fluorescein solutions.

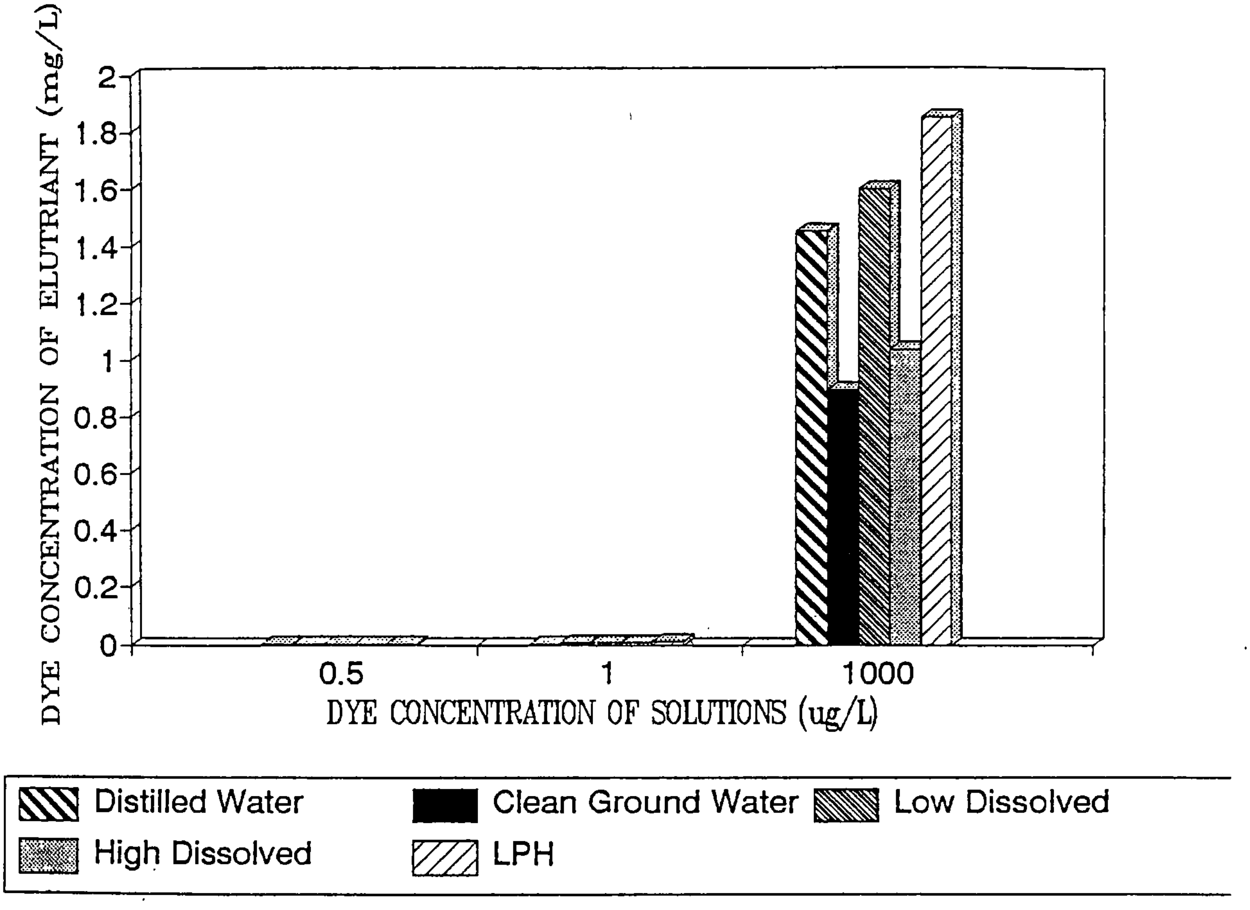

Figure 3. Dye concentration of elutriant versus RWT concentration of prepared solutions.

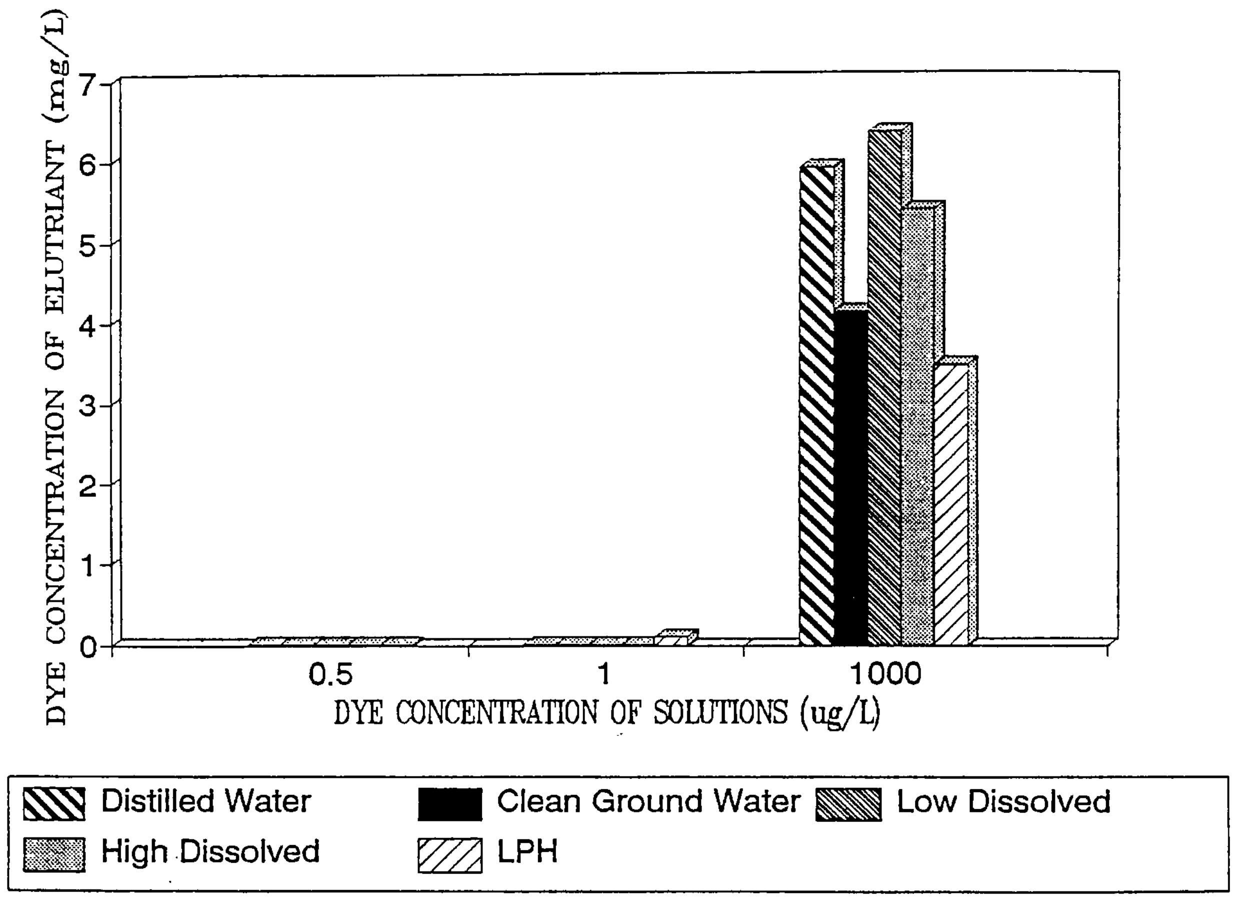

Figure 4. Dye concentration of elutriant versus fluorescein concentration in prepared solutions.

RWT Solution

Dye concentration versus BTEX concentration was also plotted for the 500 ug/L RWT ground water solutions. The plot is shown by the bar graph which is Figure 5. Figure 5 illustrates the relationship of chemical alteration of RWT with increasing concentrations of BTEX. Rhodamine WT concentrations for these solutions range from 0.988 mg/L in distilled water to 0.892 mg/L in the 9.95 mg/L BTEX ground water solution. The deviation in dye concentration between these samples is not significant indicating little to no chemical alteration of the dye with increasing concentration of dissolved BTEX. The highest concentration of BTEX shows an intermediate RWT concentration, also indicating that chemical alteration of RWT is not a function of BTEX concentration.

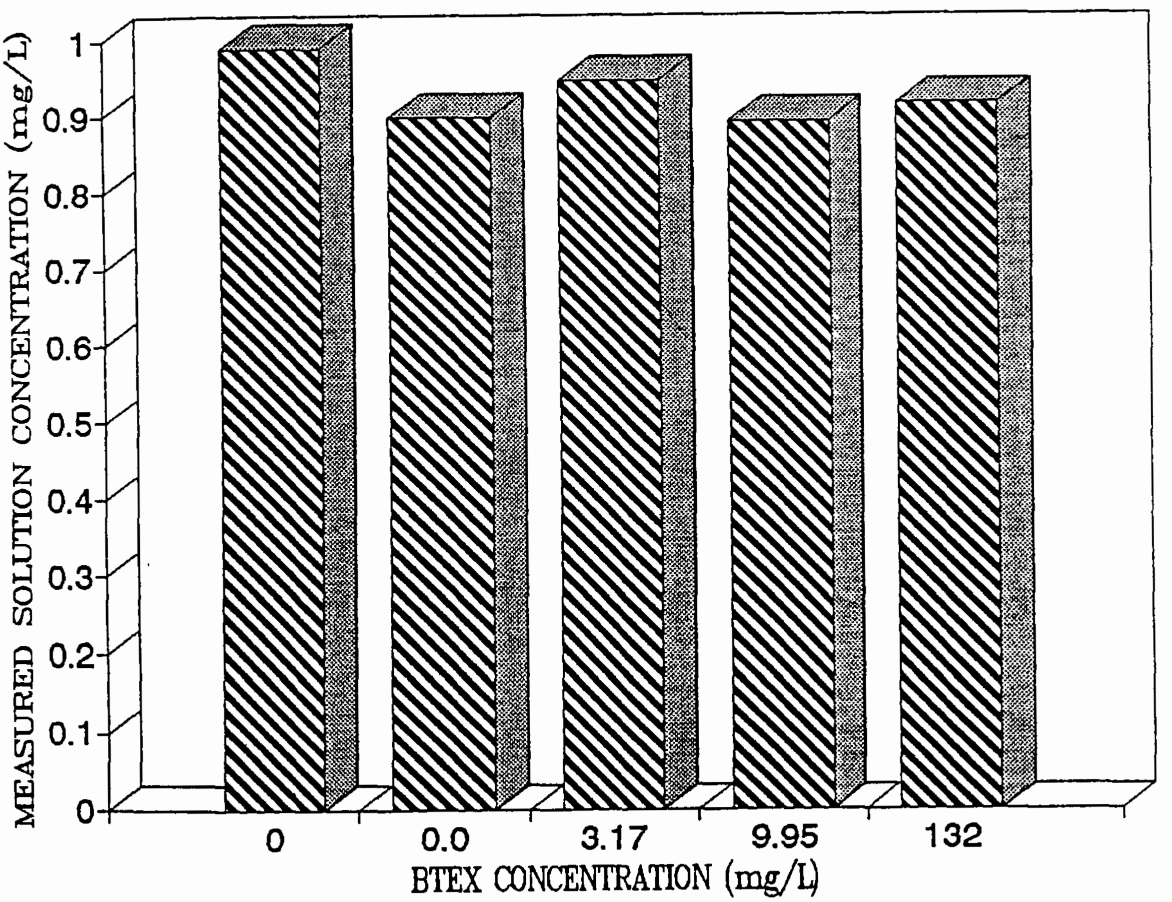

Figure 5. Dye concentration versus BTEX concentration in 1000 ug/L RWT solution.

RWT Elutriant

Results relating to the concentration of 1000 ug/L of each dye with differing concentrations of BTEX were quite visible. Figure 6 illustrates a bar graph of dye concentration versus BTEX concentrations for the 1000 ug/L elutriant. This figure illustrates that dye concentration appears to be independent of BTEX concentration. The sorption capability of the activated charcoal, therefore, does not appear to have been affected by an increase in BTEX concentration. This is intuitively contradictory with the theoretical situation where charcoal sites should be used up by the presence of additional organics other than RWT. The highest dye concentration is evidenced by the elutriant from charcoal in the highest BTEX concentration and the lowest in the "clean" ground water sample.

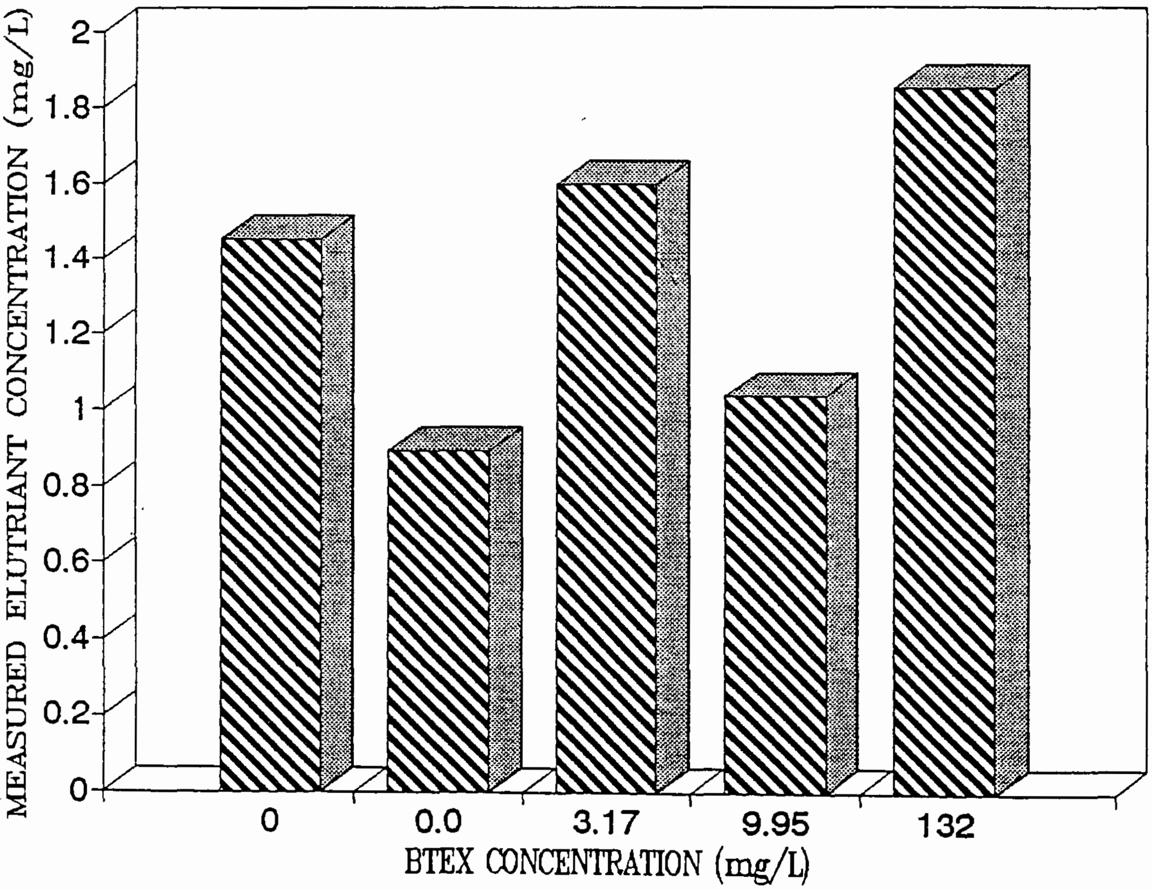

Figure 6. Dye concentration versus BTEX concentration in 1000 ug/L RWT elutriant.

Fluorescein Solutions

Figure 7 is a plot of dye concentration versus BTEX concentrations for the 1000 ug/L fluorescein dye/ground water solutions. A decrease in dye concentration is noted between the solution of distilled water and dye versus the solutions of ground water and dye. The difference in dye concentration may be an indication that naturally occurring organics in ground water may act upon the dye and reduce its concentration through chemical alteration.

Dye concentration ranges widely with a maximum of 0.772 mg/L in the distilled water/dye solution to a minimum of 0.063 in the 3.17 mg/L BTEX ground water/dye solution. Notice the marked decrease in dye concentration in the presence of detectable concentrations of BTEX. This drastic decrease signifies that the presence of petroleum hydrocarbons may have a significant effect on the chemical alteration of fluorescein in ground water. In other words the concentration of fluorescein may be decreased as a result of the chemical breakdown due to the presence of elevated levels of petroleum hydrocarbons in the ground water.

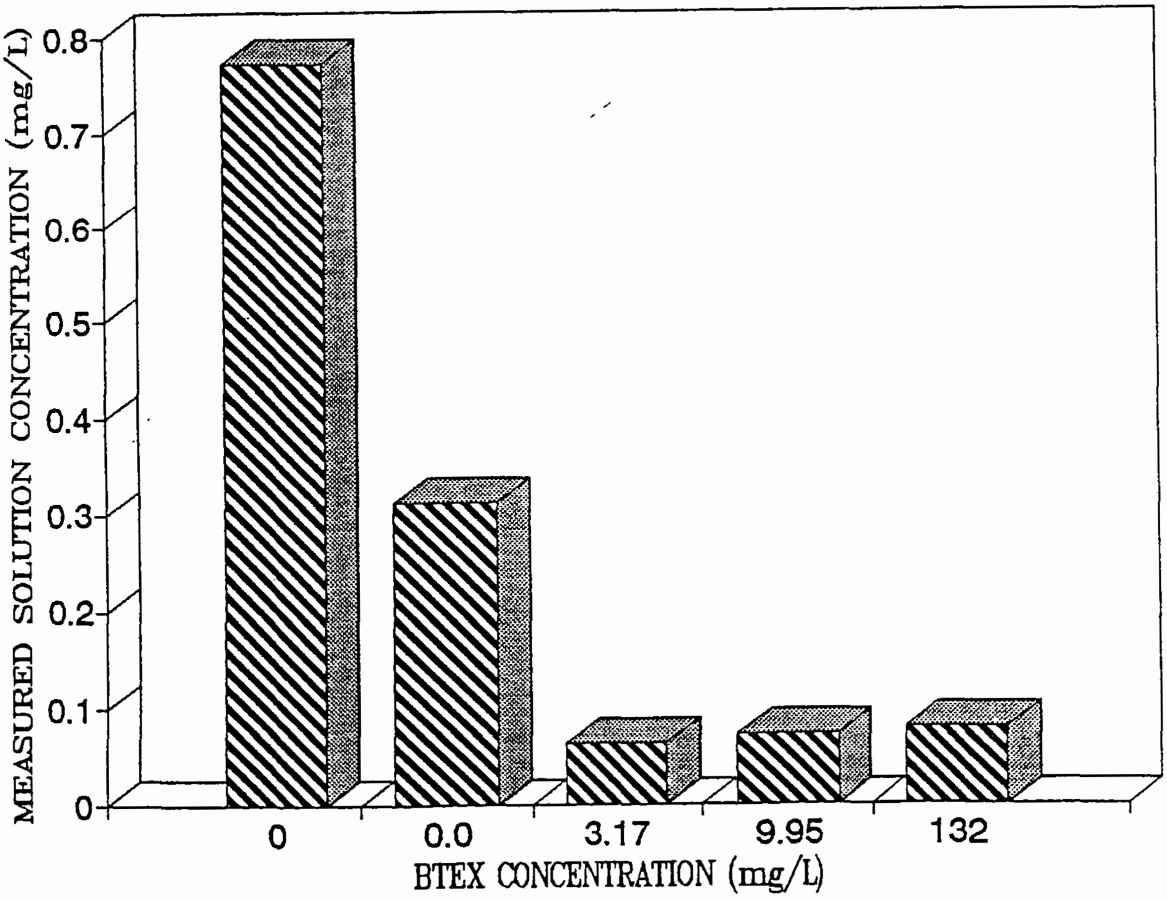

Figure 7. Dye concentration versus BTEX concentration in 1000 ug/L fluorescein solution.

Fluorescein Elutriant

Figure 8 shows the plot of dye concentration versus BTEX concentration for the elutriant from charcoal soaked in the 1000 ug/L dye/ground water solutions. Values for dye concentration in elutriant range from a maximum of 6.535 mg/L in the 3.17 mg/L total BTEX solution to a minimum of 3.473 mg/L in elutriant from the solution containing LPH. Review of Figure 7 also indicates a decrease in dye concentration of 2.880 mg/L from the highest to the lowest BTEX solution concentration. This decrease in dye concentration with an increase in BTEX concentration suggests that the efficiency of charcoal decreased with an increase in BTEX concentration. This is contradictory to results from RWT. Each dye, however, has a different molecular composition and structure and may, therefore, be sorped to the charcoal at different rates and/or different efficiencies in the presence of multiple organics. In other words the charcoal may "prefer" RWT over petroleum hydrocarbons. Fluorescein, conversely, may take a "back seat" to the hydrocarbons and be sorped less efficiently had the hydrocarbons not been present.

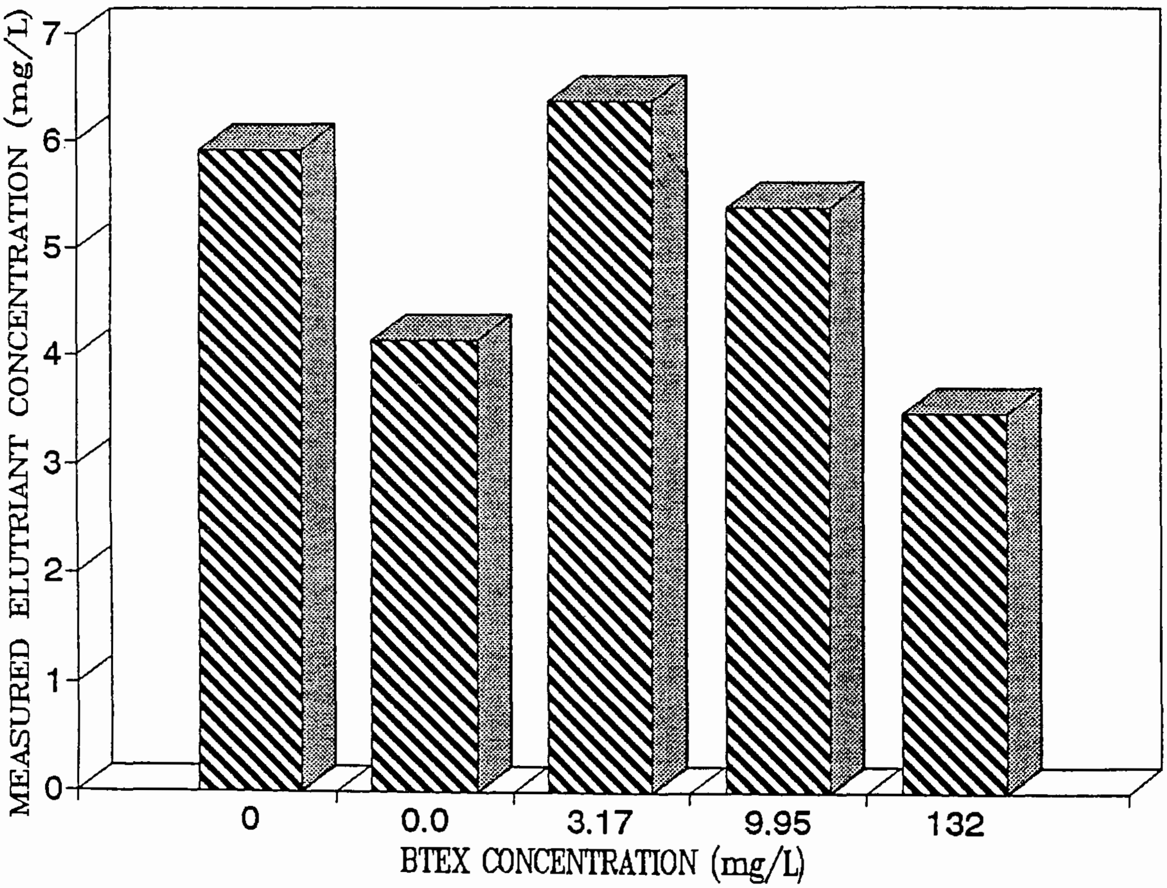

Figure 8. Dye concentration versus BTEX concentration in 1000 ug/L fluorescein elutriant.

CONCLUSIONS

Rhodamine WT

The 1000 ug/L RWT dye/groundwater solution shows little fluctuation in dye concentration. The presence of increasing concentrations of petroleum hydrocarbons does not appear to affect or chemically alter RWT. Elutriant from charcoal immersed in the 1000 ug/L RWT solutions shows no distinctive dye concentration pattern. It is unlikely, therefore, that the sorptive effectiveness of charcoal has been compromised by the presence of the petroleum hydrocarbons. Even in the presence of hydrocarbons it appears that carbon effectively sorbs RWT. Charcoal sorption sites may "prefer" RWT over constituent compounds of the petroleum hydrocarbons.

Use of RWT as a ground water tracer in an environment affected by petroleum hydrocarbons is recommended. There appears to be little if any chemical alteration of the dye and charcoal readily sorbs the dye in the presence of hydrocarbons.

Fluorescein

Elutriant from charcoal immersed in the 1000 ug/L fluorescein solutions exhibited a distinctive decrease in dye concentration with increasing BTEX concentrations. This observation may be significant in evaluating the sorption capabilities of the charcoal with respect to this dye; however, the apparent chemical alteration of the dye must first be considered. The 1000 ug/L fluorescein dye/groundwater solutions displayed a drastic reduction in dye concentration with increasing concentrations of BTEX. Based on this information the dye seems to be drastically compromised in the presence of even low concentrations of petroleum hydrocarbons.

Use of fluorescein as a ground water tracer in environments affected by petroleum hydrocarbons is not recommended. Chemical alteration of the dye may result in a negative trace that may have been a positive one had the contaminant not been present. The contaminant, therefore, may affect an area of the aquifer that was previously traced as "safe" using fluorescein. A dye concentration of 1000 ug/L is perhaps a higher concentration than is commonly recovered at collection points. Perhaps at lower concentrations more common to real situations, petroleum hydrocarbons may have a more detrimental alteration effect on the dyes.

FUTURE STUDY

As with any decent real world scientific investigation, this study seems to raise more questions than it has answered. Future research using dyes in contaminated environments is necessary to promote accurate, dependable, and reproducible tracer tests to ensure protection of the environment and human health. Here are several suggestions for further studies:

- A similar study should be conducted using a wider range and number of dye concentrations. A range from 100 ug/l to 1000 ug/l in increments of 100 ug/l is recommended;

- Studies should be conducted using a wide range of organic contaminants such as heavier hydrocarbons (diesel fuel, fuel oil, jet fuel, kerosene, waste oil) organic solvents, and organic metals complexes. The fluorescent wavelengths of the contaminants should be assessed prior to the study; and

- An actual ground water trace should be conducted in an aquifer affected by a contaminant that was previously traced while unaffected to assess the differences in dye interaction with contaminants at various stage. Flow routes should be retraced and confirmed using several different dye types.

ACKNOWLEDGEMENTS

The authors would like to extend their thanks to Paula Recker for providing the clerical support to make this publication a reality. We would also like to thank Hiram Rutledge who took time out of his already overloaded schedule to collect duplicate ground water samples necessary to perform this research.

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THE EFFECT OF PETROLEUM HYDROCARBONS ON THE SORPTION OF FLUORESCENT DYES BY ACTIVATED COCONUT CHARCOAL

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1. You exposed the charcoal to dye and BTEX simultaneously. Would int not be more relevant to expose the charcoal to the BTEX for several days and then expose it to the dye?

Yes, we agree that BTEX exposure may be the preferred method as it would better simulate a real world situation. During every dye trace it is necessary to place the detectors into contaminated ground water prior to the dye drop. The exposure of the detectors to BTEX during the dye's travel time is certainly a factor which should be considered.

2. The fluorescence of fluorescein is widely known to be greatly suppressed at pH's below about 4.5 and maximized at pH's above 9.0. These reactions are reversible, with no destruction of the dye. What data, if any, do you have on the pH of your mixtures of BTEX in water? Without such data is it possible that at least some of your apparent loss of fluorescein is due to lowered pH? Also, why not dilute samples with high concentrations of dye -- as is standard analytical procedure, with high concentrations of any substance with any analytical technique?

Monthly pH measurements have been recorded in ground water samples from this location at monthly intervals for a period of three years. The range of pH is small and generally remains between 7.6 and 8.0. It is unlikely that the prepared solutions exhibited pH below this expected range.

Yes, the higher concentration samples should have been diluted to allow for quantification. This information may have provided more concrete results on the reaction of the dyes in these environments.

3. I am distressed by the sloppiness of the design of the experiments described. so many variables are unaccounted for that it is highly questionable whether there is any significance or reliability in any of your conclusions. Why

did you present this paper NOW, rather than after a properly designed series of tests? By presenting possible erroneous results, you may inspire someone to conduct the testes with adequate rigor, but you may also mislead both investigators and administrators to perpetuate your possibly false conclusions and innocently make wrong decisions concerning spill investigations -- how to conduct them and how to interpret them. Rigorous experimental design requires more thought than money or time.

Any and all constructive criticism is welcome, however, this question does not appear to fall into the constructive category. This paper was submitted at this time because of our strong belief that these questions need to be addressed in the dye tracing community. We have simply touched on the fundamental issues using the resources we have available. Scientific study is evolutionary and many workers fail to present experimentation which yield anything but presupposed results. We felt it necessary to break this mold and present our information at this time. We would hope that our work would not mislead any workers in that our conclusions are tentative and we have simply raised questions for other workers to consider.

CONTINUOUS-FLOW FLUOROMETRY IN

GROUNDWATER TRACING

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ABSTRACT

In dye tracing, continuous flow fluorometry limits contamination, significantly improves precision and accuracy and allows immediate response to tracer appearance. However, it is logistically more complex than simpler methods, and provides data rather than actual water samples which may be critically important should corrective processing be required. Disadvantages outweigh advantages for most tracer applications. However, an attempt to investigate the detailed evolution of tracer dye through a defined water-filled conduit lent itself to continuous flow fluorometry and led to some developments of the method.

The Jordtulla system in Svartisen, northern Norway drains a 27 km^2 subarctic catchment through a 580 m largely phreatic conduit averaging 5 m in diameter. The system drains directly from a large lake, has clear well-buffered water and has been mapped by cave divers. It thus provides an ideal site in which to determine the relationship between conduit hydraulics, morphology and tracer behaviour. Sampling occurred at the resurgence (Fiskepole), and at a window (Mellomgrotta) part way through the system. Portable alternators and automobile batteries powered pumps and standard Turner Designs Model 10 Series Fluorometers. Water temperature, fluorescence and stage were recorded on Campbell CR21X Microloggers. Thirty three traces were made over 9 days, providing a total of 50 breakthrough curves.

Fluorometer response was inadequate on the unusually steep

rising edge of the breakthrough curve, demanding a complex logging algorithm. The resulting data required substantial processing before satisfactory breakthrough curves could be defined. Fiskepole breakthrough curves are consistent and reproducible, but the Mellomgrotta data showed a distinctive structure indicating incomplete mixing. Subtle detail in individual curves appears to reflect the style of tracer injection.

The heavy power demand of pump and fluorometer limit the portability of the present system. Sophisticated data loggers permit excellent control and measurement capability, but add yet a further level of logistic complexity. Continuous flow fluorometry remains justified only by stringent experimental or legal demands.

INTRODUCTION: SAMPLING STRATEGY AND PROJECT DESIGN

Strategies adopted in groundwater tracing are usually a compromise between objectives and resources available. The tracer preferred by most workers in karst terrain is slug injected dye because it meets the fundamental requirements of safety, availability (including legality in most cases), cost and detectability. A range of sampling strategies are available for determining dye return characteristics. These range in order of increasing technology and information content from visual observation, through charcoal detectors, discrete water sampling and continuous flow fluorometry. The analytical system similarly ranges from visual detection, through filter fluorometers to spectrofluorometers.

As a general rule, the more advanced technology can deliver more precise and accurate data, but only at greater logistical and financial cost. The fundamental question to be posed at the inception of any tracer experiment is the information required. Resource availability is a secondary question, although it may prevent investigation of some questions.

The resources available should not dictate the methodology adopted. On the one hand this may result in data inadequate to answer the question being posed. On the other hand, the data may be obtained at excessive expense. In addition, an over-technical trace may be less reliable, because more complex tests present more opportunity for error.

The majority of important tracer tests will be simple point-to-point detector traces with the hydrogeographical objective of determining tracer trajectories. More sophisticated traces demand prior knowledge of the hydrogeography derived from point-to-point tracing or well-judged hydrogeological reconnaissance.

However, there are legal and scientific situations where very high quality data may be required. First, as hydrogeological litigation evolves, demonstrably higher quality data will become more important. This will provide a means of assuaging the increasingly expensive confrontation, scrutiny and derogation of "expert" witness. The technology required, along with simple quality-assurance-quality-control requirements, may make groundwater tracing considerably more expensive. However, it seems more rational to expend resources on a high calibre trace than in vacuous litigation.

Scientific work may also demand high quality data either to verify the technique or the system under scrutiny. In field experiments, the difficulty of exerting effective scientific control makes it critically important to minimise any error masking the experimental variance. The remainder of this paper is a report on the design and implementation of a continuous-flow fluorometric system for field studies in remote areas.

CONTINUOUS-FLOW FLUOROMETRY

Modern field fluorometers are designed to allow water to pass directly through the analytical cuvette, permitting a continuous reading of water fluorescence. Normally, this water is drawn directly from a spring or stream and a continuous trace of fluorescence is printed on an analogue recorder (e.g. Turner Designs 1978). There are problems and advantages inherent in this system. These will be addressed with special reference to the Turner Designs Model 10 Series Fluorometer which is the most common and, despite its age, arguably the most suitable instrument currently available for field fluorometry.

Calibration of the continuous flow cell requires rather large quantities of dye standard, perhaps a litre or more. Usually, a short hose length is used to fill the flow cell from the bottom. Several rinsings may be required until a stable reading is obtained on successive flushes. With high temperature coefficient dyes (e.g. the "Rhodamines") it is essential to record the temperature of the calibration standard.

In remote areas without A.C. line voltage, power consumption is a serious consideration, which can significantly increase logistical load. The Model 10 draws some 2 amps of power when operating from a 12 volt source, and thus requires a rechargeable battery source (usually one or more automobile batteries) and a portable alternator with fuel (usually gasoline). Such instrumental energy consumption is appalling by contemporary standards, but is dictated by the "chopper" and high voltage photomultiplier technology which gives the instrument its impressive analytical capability.

Instrument power demands are often overshadowed where it becomes necessary to pump water up from a stream at a suitably rapid rate. A heavy duty 12 v marine "bilge" pump will often draw 3-5 amps. A siphon can supply water at a suitable rate for zero power, but such systems are inherently unreliable and their flow rate is strongly dependent on stream stage. A combination pump-assisted siphon is very efficient, allowing (once primed) a much lower power pump, providing a priming system is incorporated. It is generally suggested that pumps should be placed downstream of the fluorometer to avoid over-pressurisation. However, we have found that in small diameter pipe (e.g. ½" garden hose), this can result in degassing of water. Instrument noise increases and dye detectability suffers greatly from the stream of bubbles produced.

The lightest fluorometer-alternator package with a lightweight motorcycle battery weighs at least 30 kg. The system may operate for only an hour or so without recharging. It is imperative in these circumstances to have prior knowledge of likely travel times. Where A.C. line power is available, the fluorometer will run indefinitely. Line power instability may cause some interference, e.g. low pressure mercury lamps (Alexander pers. comm. 1991). It may be worthwhile to run the fluorometer from a 12 volt battery continuously charged from an A.C. battery charger. This isolates the fluorometer from line noise and improves precision. Fouling of the analytical cuvette and calibration drift become the primary long-term problems. These can be corrected on an itinerant basis. Although it is possible to collect an excessive quantity of data, over long duration tests, or those requiring high frequency sampling, continuous flow fluorometry may be much less expensive and less prone to error than equivalent discrete sampling procedures.

RECORDING

The simplest recording system is an analogue chart recorder, matched to the instrument telemetry. A chart record provides real-time "hard-copy" of a trace. This is valuable as a statutory record. It also allows an immediate impression of trace progress and is invaluable in managing sampling strategy, e.g. chart speed, regional sampling patterns, sampling frequency and sampling termination. However, the chart record includes all instrument noise, can be mis-matched to the fluorometer and is often unreliable.

Modern digital data loggers are less expensive than analogue chart recorders and considerably more reliable and powerful. They are no more technically demanding than chart recorders, although they do not produce any real-time hard copy; their memory and recording media can be volatile. Work with both Campbell Scientific and Aanderaa data loggers suggests that with suitable memory back-up facilities, this is a mature technology with any modern field logger system.

The single greatest problem with continuous fluorometry occurs where the direct fluorescence reading is dependent on other variables than tracer concentration. For example, cross-fluorescence from other dyes, background variations, temperature, pH and turbidity controls on fluorescence. These problems normally require a discrete sample so that some sample preparation or further analysis can be undertaken before fluorescence can be determined. Many data loggers also permit data processing, logical process control and data management. It may thus be possible to correct readings to obtain a direct record of tracer concentration. For example, it is a simple matter to record sample temperature and correct to standard temperature. In general, it is advisable to obtain required data on site, but to avoid on-site processing because errors become more probable as processing becomes more complex. In the worst case, this may invalidate the entire record. Providing sufficient memory is available, it is better to record essential parameters such as temperature, mass injected etc. and to process the data after downloading to a suitable computer system or spreadsheet.

A great advantage to logger processing is the capacity to smooth the data by averaging. Measurements can be made with very high frequency (perhaps every 0.1 s) and the average and standard deviation recorded (perhaps every 5 minutes). Each recorded value in this example is thus the average of 3000 readings. The signal noise which obfuscates analogue chart traces is removed and, providing there is no significant trend over the recording interval, a very smooth (precise) record is generated. This facility is of such great value at low tracer concentrations that it is now being applied to the analysis of discrete samples.

Loggers can also be programmed to respond to measured data. Typically, this will involve higher frequency measurement during rapid evolution of tracer concentration and lower sampling rates in the tail of a breakthrough curve.

Continuous flow fluorometry can generate a very large data base compared to that possible with discrete sample analysis. This is advantageous, unless there is a need for collection of physical samples. Such a need may arise for reasons of legal evidence, or more commonly because some form of sample processing is necessary to determine correct dye concentrations. Background, turbidity, pH, temperature and cross-fluorescence are examples of factors which may require some pre-analytical processing.

The Model 10 combined with a data logger can provide excellent field data in a typical trace. However, the Turner Designs fluorometer is unable to automatically switch over its full dynamic range, a major constraint. It is also possible to use only a single filter combination with each fluorometer which limits its value in multi-tracing. Fountain (1989 pers comm.) has developed a logger-controlled hydraulic switching system which can route water from a number of intakes into a single fluorometer. In applications with very short rise times, the instrument is unable to respond sufficiently quickly to rapid increases in fluorescence. The need to switch through all ranges on rising fluorescence while only a single range change is necessary on falling fluorescence is a fundamental design flaw, albeit one of limited impact.

Logger data collected from continuous-flow fluorometers provides the best data currently available on tracer breakthrough curves. In sites with high background or complex traces requiring careful interpretation, spectrofluorometric analysis of grab samples is advisable. Data acquired from very rapid sampling and very rapid changes in fluorescence can be extremely complex to process. It has proven worthwhile to record at two levels during critical rapid changes; the first is the typical averaging approach, the other is a direct recording of readings which can be invaluable in interpreting the reliability of the averaged data. A visual interactive editing programme has been found the most useful way to process fluorometric data.

FIELD EXPERIMENTS

Earlier work on continuous flow fluorometry in the Canadian Karst systems at Castleguard and Maligne are reported in Smart (a,b). These and other (multi-tracer) experiments showed clear advantages in continuous flow fluorometry, especially for revealing precise detail in the breakthrough curve and for eradicating contamination.

More recent work on has taken place on the Jordtulla conduit, Svartisen, Norway (Lauritzen et al. 1985), an exceptional natural laboratory for experiments on conduit flow. It is a 5 m in diameter, 580 m long, singular conduit, which has been completely explored and mapped by divers and (draining an extensive lake) varies gradually in discharge between 1 and 50 m^3s^{-1}. A series of 33 tracer experiments were made in May 1989, a period of transition from winter base flow to spring floods (Figure 1). Injections of ~20 g Rhodamine WT (Acid Red 338) were made at the sink point or at one of two karst windows, Mellomgrotta and Waltergrotta. Fluorescence was measured continuously on two Turner Designs Model 10 Series Fluorometers powered from 12 v automobile batteries. Samples were collected at Fiskepole, the resurgence and at Mellomgrotta a karst window 150 m from the sink point. Fluorescence and sample temperature data were recorded on Campbell Scientific CR21X Microloggers as one minute averages or 60 readings and as 10-second direct readings during periods of high fluorescence. The data were segregated into discrete traces, corrected to a standard injected mass and standard temperature and edited using an interactive sorting routine.

Figure 2 shows an example of tracer breakthrough curves from Mellomgrotta and Fiskepole. Note the extremely rapid rise of the Mellomgrotta curve and clear instability compared to the Fiskepole trace. When inspected in detail (Figure 3), the Mellomgrotta trace shows a rapid decline in fluorescence background prior to arrival of the dye. The latter problem results from desorption of dye from the vinyl hose used to feed the fluorometer. It is significant because of the high dye concentrations involved and the cessation of pumping between traces. Adequate flushing time is essential before dye arrival. The impact of such adsorption can be minimised by continuous flushing of the shortest possible length of intake hose. Provided each subsequent dye impulse is comparable in

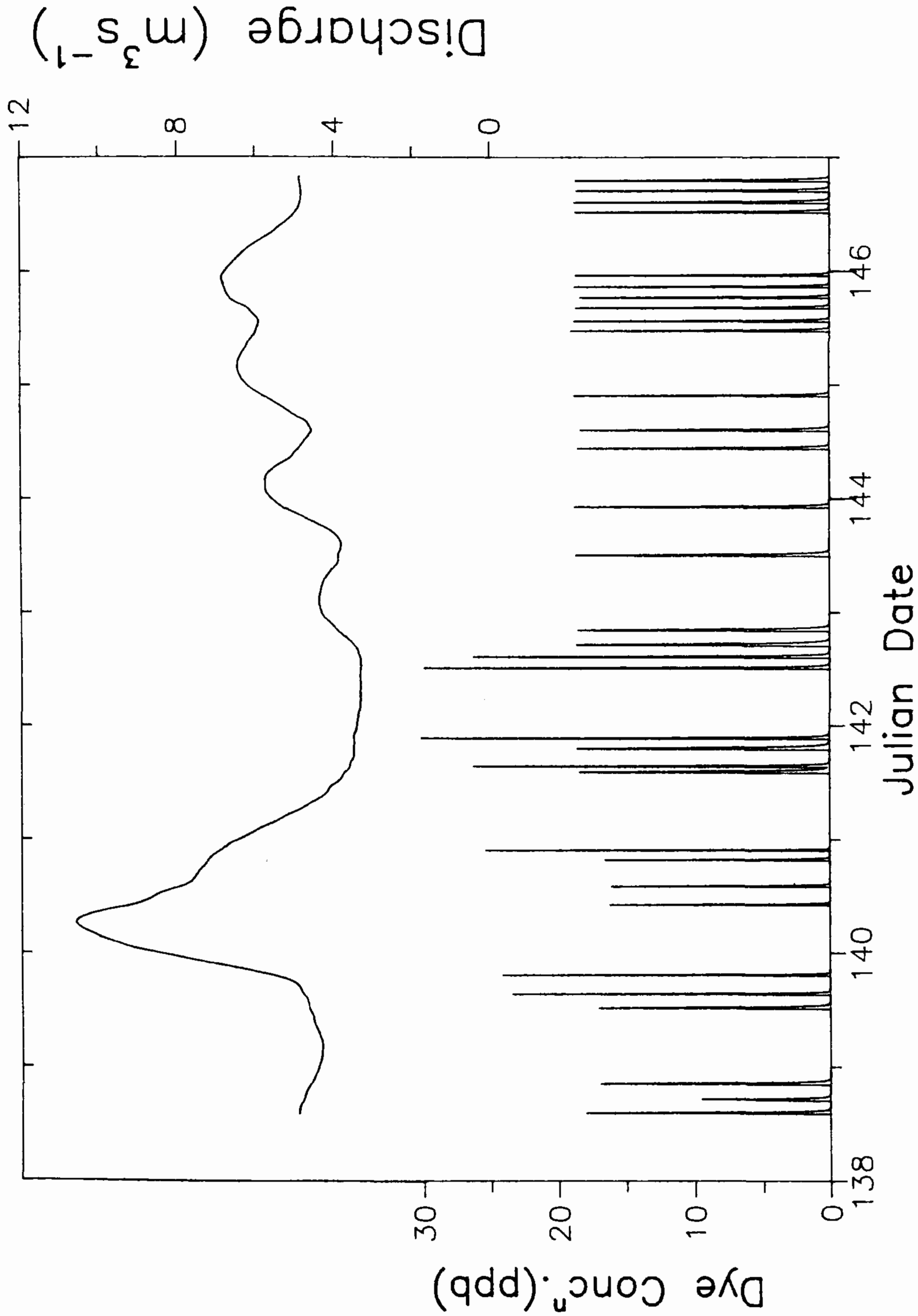

Figure 1. Discharge and fluorescence at Fiskepole, the outlet of the Jordtulla System. Obvious differences in peak fluorescence reflect injections made at intermediate points in the conduit. Trace two was incompletely documented.

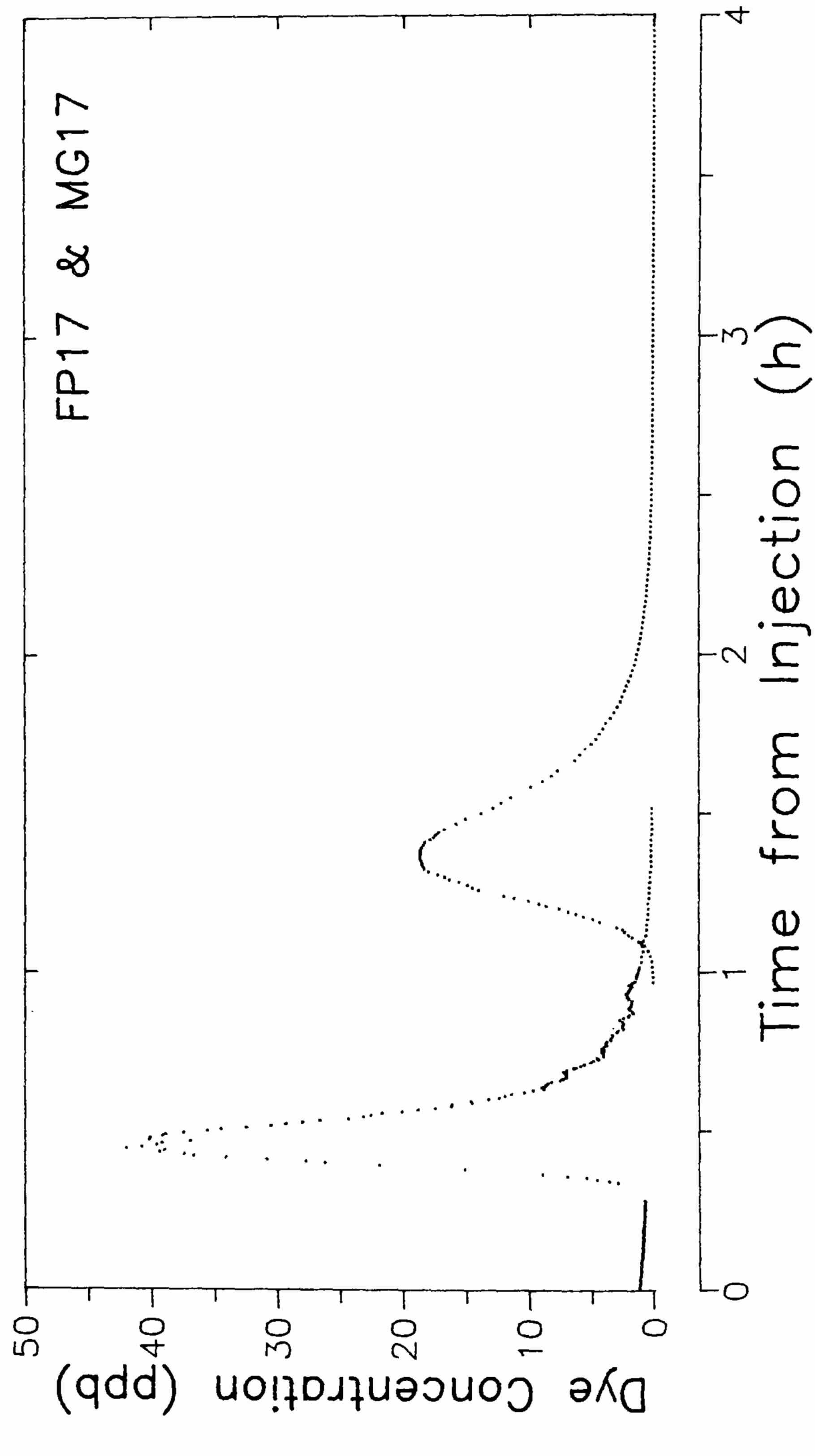

Figure 2. Comparison of the Fiskepole and Mellomgrotta breakthrough curves for a single trace. The former curve shows considerable noise and is much less well defined than the latter.

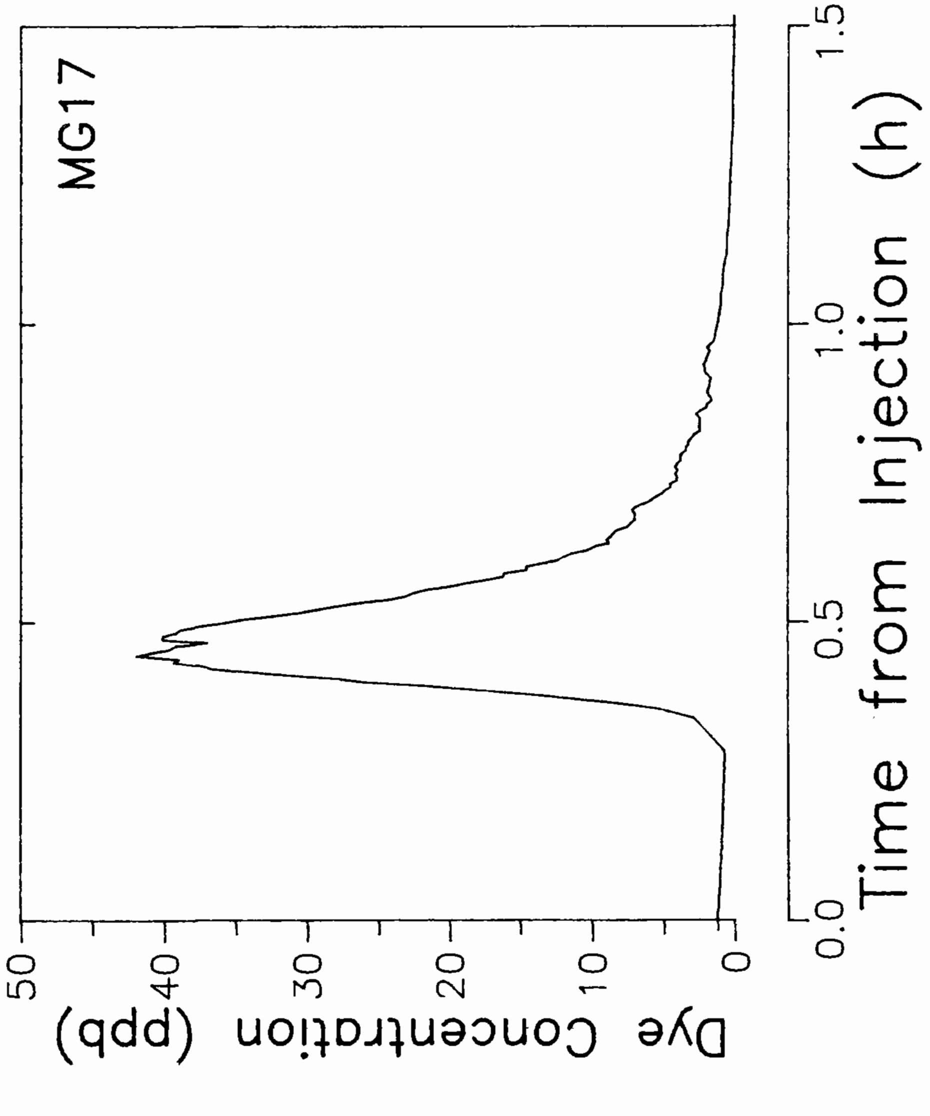

Figure 3. Detailed plot of the Mellomgrotta trace. Note the decline in fluorescence before dye breakthrough and the 5% oscilation in the record representing turbulent structures in the stream and caused by inadequate mixing.

magnitude to its predecessor, there should be no significant impact. However, it may be possible to mimic subtle variations in the detailed form of the breakthrough curve by adsorptive exchange.

The gross instability of the Mellomgrotta curve was reproducible in frequency and amplitude, but not phase on all traces. It is considered to be an indication of incomplete mixing in the system. It is thus a measure of the scale of turbulence in the stream. Such irregularity in a continuous record can be used to indicate inadequate mixing length in a tracer study.

The Fiskepole breakthrough curves were in general well-mixed and showed little deviation from the anticipated form. Apart from discharge, the single largest source of variation in form results from the injection style. Although injection technique was reasonably standardised, the exact pattern of secondary flows at the injection site varied somewhat with discharge. A dye-trapping eddy at the sink thus resulted in slight shoulders on the tracer recession under higher flows.

Our technical conclusions from this experiment are that despite the excellent technical procedures, there remain practical limitations to the precision of field fluorometry, although the error resulting from imprecision is far from significant relative to overall variance. However, changes in the breakthrough curve in response to discharge are very subtle over the range of discharges investigated. For this system, therefore, extremely precise procedures are required. Smaller diameter conduits or greater changes in discharge are necessary for more satisfactory experiments, but of course this may prevent complete exploration.

Further work will investigate the relationship of tracer dispersion and advection relative to injection site and discharge. Preliminary studies indicate that the limited free-surface associated with the karst windows have a detectable effect on the breakthrough curve. Models developed for a pure conduit, will thus require modification to accurately represent the system.

CONCLUSIONS

A continuous flow fluorometer coupled to a good data logger can provide excellent tracer breakthrough curves. The high resolution of the record and lack of sample handling provide precise curve definition and can minimise labour. However, in remote sites, or where tracer concentration is not simply measured the technique can become logistically complex, expensive and inaccurate. However, recent development of less precise, but quite adequate submersible fluorometers (1991) suggests that multi-tracing to multiple springs may soon become affordable, although still requiring considerable logistical effort.

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PERSONAL RESUMES

Chris Smart obtained a BSc from the University of Bristol (UK) in 1974 where his interest in karst and hydrology was sparked. The statistical properties of meandering cave passages provided the focus of an MSc gained from the University of Alberta in 1977. A PhD. was awarded by McMaster University in 1983 following a study on the hydrology of the partially subglacial Castleguard Karst aquifer in the Canadian Rockies. Present research interests focus on instrumented field studies in alpine terrain, including karst, till, glacier and fracture aquifers. This work is complimented by work on instrument development, data processing, mapping and numerical modelling.

An organic chemist by training, Stein-Eric Lauritzen converted an amateur interest in caves and karst into a profession. He is currently involved in projects in karst hydrology and hydrochemistry, quaternary geology and cave archaeology, glacier hydrology and karst resource conservation. He runs the Norwegian Uranium-Thorium dating facility in the University of Bergen.

Chris Smart - Continuous Flow Fluorometry

1. Given that most applications of continuous flow fluorometry will probably be at sites where there is ready access to electric power from the mains, I think you have overemphasized the difficulties of the technique. Please comment.

   Most springs and cave streams are remote from mains power. Therefore, the arbitrary application of continuous flow fluorometry will be constrained by the logistics of power supply. At those few sites with <u>safe</u> mains power, the technique is straightforward; however, other problems remain. First are technical problems of robust data recording. Second are irrecoverable <u>analytical</u> problems which may not be evident in the data record. Finally, problems associated with lack of physical samples will also remain unless a sampler is operated concurrently. Therefore, provision of mains power only overcomes the most superficial, albeit immediate difficulties of continuous flow fluorometry.

DEVELOPMENT OF A FLOW-THROUGH FILTER FLUOROMETER FOR USE IN QUANTITATIVE DYE TRACING AT MAMMOTH CAVE NATIONAL PARK

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ABSTRACT

A series of quantitative traces were completed in the Buffalo Spring ground water basin in Mammoth Cave National Park as part of the field testing of a newly developed filter fluorometer. The RME flow-through filter fluorometer is an inexpensive, labor-saving, battery-operated, submersible device that, when interfaced with a digital datalogger, is capable of precisely measuring the travel time of two dye slugs (rhodamine WT and fluorescein) simultaneously. It is also able to measure the approximate dye concentrations passing a recovery point. Interpretation of RME data yielded unprecedented information concerning the hydrology of the Buffalo Spring basin--including the unanticipated discovery of a major flow-route.

INTRODUCTION

Fluorescent tracer dyes are commonly used in the study of ground water movement in karst terranes. Qualitative dye tracing, using passive dye-detectors like cotton and activated charcoal to recover the dye, is frequently employed to approximate ground water flow-routes and define ground water basin boundaries. Quantitative dye tracing, which requires the measuring of changing dye concentrations at a recovery point, is useful in the determination of ground water velocities, conduit condition (phreatic or vadose), unexpected flow routes, and water "budgets" (for basins with multiple discharge points). If the flow of dye through an aquifer is closely documented using quantitative tracing, models of soluble point source contamination events may be generated. Such models may be used by ground water managers to aid in drafting contingency plans for dealing with acute ground water pollution.

Quantitative dye tracing--much more expensive and labor intensive than qualitative tracing--is generally performed only as a supplement to qualitative tracing. Typical methods used to recover dye include grab sampling, automatic sampling, and flow-though fluorometry. Each of these methods has inherent drawbacks. Grab sampling is enormously labor intensive. Automatic sampling is moderately labor intensive and costly--automatic samplers cost over $2000 each. Samples obtained using grab or automatic sampling must be quantitatively analyzed on a fluorometer. Fluorometer prices start at around $7000. Flow-through fluorometers are able to directly measure concentrations of dye at a recovery point; however, in addition to being expensive, they require a pump to generate flow through the instrument. The energy requirements of the pump and fluorometer make this method impractical in remote areas where electrical service is not available. A need exists for the development of cheaper and easier techniques capable of obtaining results of similar precision.

THE RME FILTER FLUOROMETER

An alternative to conventional dye recovery methods has been developed and is being used extensively at Mammoth Cave National Park. The RME flow-through filter fluorometer is an inexpensive, battery-operated, submersible probe, supported by a digital datalogger. It may be deployed in the field for extensive periods of time, and requires only occasional servicing. The RME uses 6 volts DC and draws less than 100 ma/hr. Since it is submerged into the spring and is designed to slowly draw water through itself, no pump is required. The RME is capable of continuously measuring small concentrations of two dyes simultaneously--rhodamine WT (C.I. Acid Red 388) down to 0.5 ppb and fluorescein (C.I. Acid Yellow 73) down to 5 ppb. The material cost of building an RME is approximately $175 per unit. A datalogger with versatile programming is required to execute and record the data measurements. The datalogger and RME battery power supply, attached to the RME through waterproof wire, must be placed above the highest possible water level in a weatherproof enclosure.

The RME fluorometer is two filter fluorometers in one package. It has one light source, a 4-watt clear quartz mercury ultraviolet lamp, sandwiched between two flow-through sample tubes (Figure 1). The flow-through tubes are made of 6 inch sections of 1-inch ID aluminum box tubing. Two elongate windows are milled into each tube at right angles to each other. Clear microscope slide glass is mounted across each window from the inside using a silicone sealant.

LIGHT FILTER SETS

Situated between the lamp and the rhodamine sample tube is an

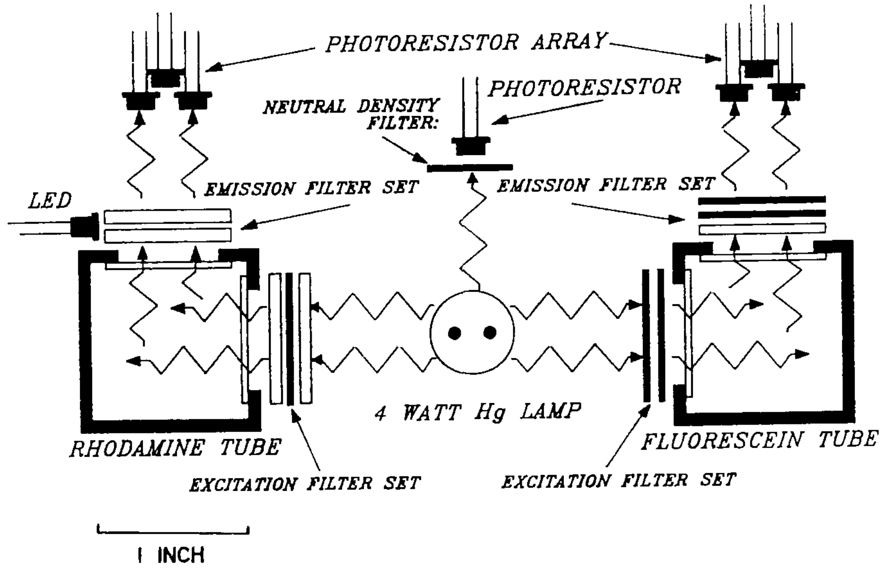

Figure 1. Diagrammatic cross-section of the RME filter fluorometer.

excitation filter set, composed of a Kodak Wratten 61 gel filter sealed between two Corning I-60 colored glass filters (recommended in Smart and Laidlaw, 1977). This filter set is designed to allow only the 546nm mercury line light to illuminate the inside of the sample tube. The other major spectral lines emitted by the mercury lamp (578nm, 436nm, 405nm, 365nm, and 254nm) are absorbed by the filter set. The 546nm light illuminating the interior of the rhodamine tube is within the excitation spectrum for rhodamine WT (its excitation maximum is about 555nm), so if that dye were present in the sample tube, it would be induced to fluoresce. An emission filter set, composed of a Corning 3-66 and a Corning 4-97, is located between the other window and the photodetective array. The secondary filter set is designed to transmit a spectrum that has peak nearly coinciding with the emission maximum of rhodamine WT (about 580nm); the filters are nearly opaque to wavelengths outside this relatively narrow spectrum.

The excitation filter set for the fluorescein tube is a combination Wratten 2A and a Wratten 47B. It transmits the 436nm mercury line, which is within the excitation spectrum of fluorescein (the excitation maximum is 490) and is nearly opaque to the other lines. The emission filter, located between the emission window and a photodetective array, is composed of a Wratten 2A, a Wratten 12, and a Corning 4-97 (recommended in Turner Designs, 1983). This filter set transmits a portion of the excitation spectrum of fluorescein (the maximum is about 520nm) to the photodetective array. All Wratten filters are sealed inside clear glass to help preserve them.

THE PHOTODETECTORS

The RME uses cadmium sulfide photoresistors as photodetectors. The electrical resistance of a CdS photoresistor varies--in an inverse log relationship--with the intensity of light striking it. The photoresistors are extremely sensitive even to tiny changes in light intensity--especially in the 500nm to 600nm range. Both of the RME's sample tubes have an array of three photoresistors, connected in parallel and located outside the emission filter (Figure 1). Another photoresistor, along with a protective neutral density filter, is used to monitor intensity fluctuations in the mercury lamp.

PROGRAMMING THE RME

To conserve battery power and to extend the life of the heat sensitive Wratten filters, the lamp, and the lamp circuitry, the lamp is only operated periodically. The datalogger (Campbell Scientific 21X microloggers were used by this investigator), via a relay, switches the lamp on for one minute out of every ten. At the end of that one minute the resistances of the rhodamine array, fluorescein array, and the lamp reference are measured, using a DC half bridge, and stored. If dye above a pre-chosen concentration is sensed, the sampling interval will change to once per five minutes and then revert back to ten minutes when that concentration is no longer exceeded. This insures a better probability of documenting short duration features.

Cadmium sulfide photoresistors have an undesirable inherent behavior called a memory or light history. If they are placed in total darkness for even a brief period of time (as they are when the lamp is off), they will become "stuck" in this very high resistance dark state; small increases in illumination will not cause any change in electrical resistance. Minute increases in dye concentration above background would therefore go unnoticed. To counter this, an LED that keeps the rhodamine array slightly illuminated is switched on when the lamp is switched off. The fluorescein array receives enough exciting light through its secondary filter (an otherwise negative trait) that, when the lamp is switched on, the array is quickly "snapped out" of the memory state.

RME ENCLOSURE

The electronics and optics of the RME are enclosed in a watertight 4-inch schedule 40 PVC pipe compartment. The sample tubes are connected to 3/4-inch PVC pipes which pass through the endcaps of the compartment. To prevent ambient light from reaching the sample tubes, light baffles made of 45 and 90 degree ells are inserted between the sample tubes and the outside (Figure 2). The front light baffles are removable to facilitate cleaning the sample tubes. All piping and the enclosure itself are painted black as further protection against ambient light. On the downstream end of the RME exterior is an inverted funnel-shaped feature called a drag inducer. When the RME is properly oriented in a flowing stream, the drag inducer produces a vacuum

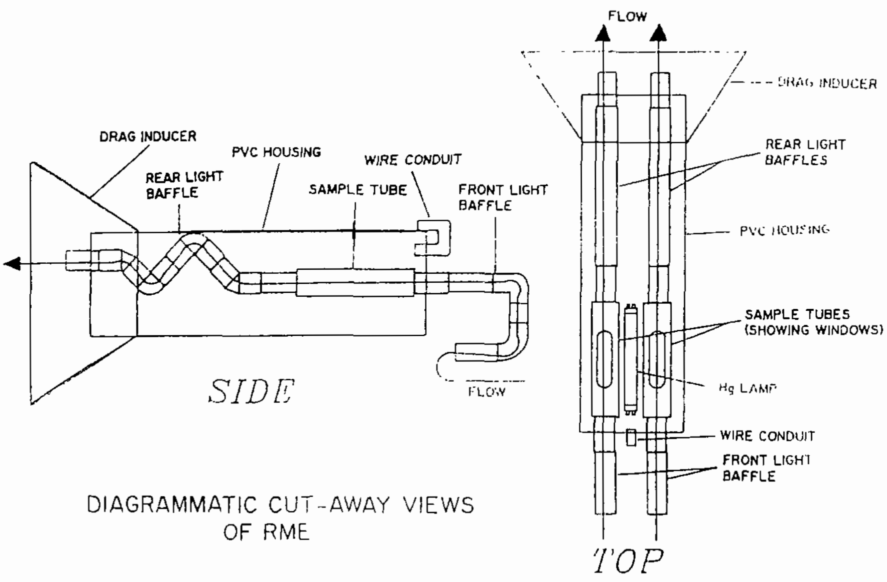

Figure 2. Diagrammatic cut-away views of the RME.

effect which draws water through the sample tubes. This insures that the RME is taking a sample representative of the water around it at any given time.

SUMMARY OF HOW THE RME MEASURES DYE CONCENTRATION

The following is a very basic summary of the physical relationships employed by the datalogger and the RME to measure dye concentration:

1). A change in dye concentration results in a directly proportional change in fluorescence.
2). A change in fluorescence results in a directly proportional change in the amount of illumination striking the CdS photodetector.
3). A change in the amount of illumination striking the photodetector results in an inverse logarithmic change in the electrical resistance of the photodetector.
4). A change in the electrical resistance of the array is measured as a proportional change in the output voltage of a DC half bridge by the datalogger.

The datalogger records the output voltages of DC half bridge measurements, which are downloaded from the logger onto a cassette tape or into a data can and then loaded into a PC spreadsheet where the following transformations may be made to it:

1). Using a conversion formula, output voltages are converted to resistances.
2). Resistances are converted into dye concentrations by interpolating from a calibration curve--calibration curves are created prior to field deployment by plugging one end of an RME's flow-through tubes, pouring in a series of

standards, and recording the resultant resistances.
3). Concentrations are temperature compensated using the formula provided by Smart and Laidlaw (1977).
4). Instrumentational background is subtracted out by "zeroing" data immediately proceeding the leading edge of a dye slug.
5). Temperature compensated dye concentrations are multiplied by the discharge of the spring or stream to determine dye load.
6). The area under a dye load curve is calculated to determine the total amount of dye recovered.

Because of deficiencies in the present RME design (primarily in the light filter sets), fluorescein concentrations may only be roughly determined; therefore, fluorescein may only be reliably used with the RME in ground water time of travel study.

THE DETERMINATION OF THE HYDROLOGY OF THE BUFFALO SPRING GROUND WATER BASIN USING RME FLUOROMETRY

INTRODUCTION

The Buffalo Spring ground water basin occupies about a $20Km^2$ portion of Mammoth Cave National Park, Kentucky. It is located north of the Green River, just west of its confluence with the Nolin River within the Hilly Country of the Chester Upland (George, 1989). Buffalo Spring is stratigraphically located near the middle of the Girkin Formation, the uppermost unit of a thick section of highly karstifiable Mississippian limestone. An alternating sequence of relatively thin Mississippian sandstones and limestones and the basal Pennsylvanian Caseyville Formation are located above the Girkin Formation. The regional dip is a relatively gentle 5 to 15 m/Km to the west-northwest. The Buffalo Creek surface drainage splits into its two main tributaries, the Wet Prong and the Dry Prong, about 1Km from the Green River. Surface flow is absent in both of these branches where the Girkin Formation crops out, except under high flow conditions. The surface streams are lost through a series of sequential ponors downstream from the upper Girkin contact. Many of the tributaries to both the Wet and Dry Prongs also sink into the upper Girkin.

Buffalo Spring is a rise pit type spring. It has a highly variable discharge that ranges between about 60 and 1800 l/s, with an average discharge of about 500 l/s. Qualitative dye tracing by Meiman and Ryan (1990) confirmed that sinking water from the Wet Prong and Dry Prong sequential ponors resurges at Buffalo Spring (Figure 3). A large tributary to the Dry Prong, Mill Branch, and numerous smaller tributaries to both Prongs were also traced to Buffalo Spring. Qualitative dye tracing showed that Confluence Spring was an overflow spring for Buffalo Spring. Fort's Funnel is a cave located on the flank of Collie Ridge just northwest of the Dry Prong (Figure 3) and containing a large

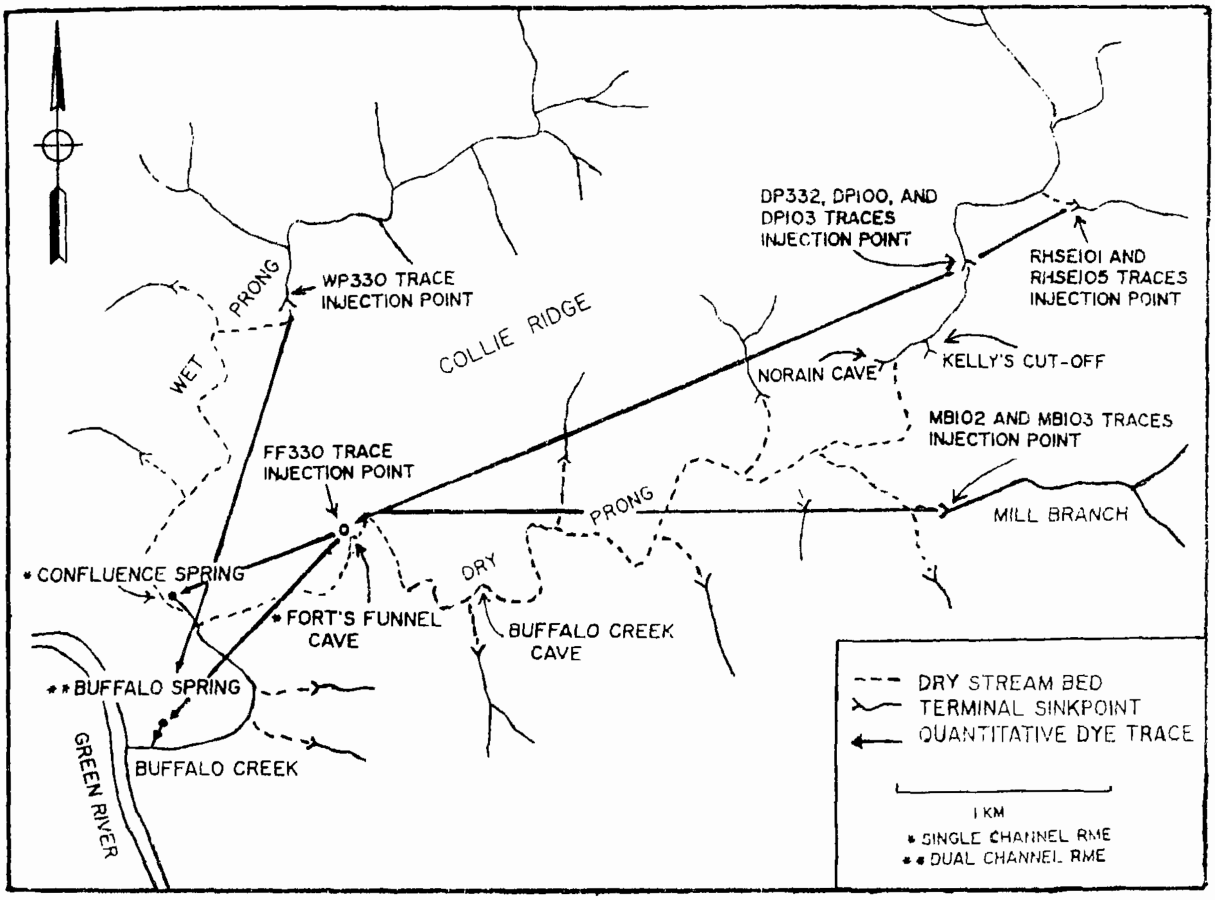

Figure 3. Map of the Buffalo Spring ground water basin.

stream. The discharge of the cave stream is roughly half that of Buffalo Spring. Every qualitative dye trace performed in the basin was recovered with positive results at Fort's Funnel as well as at Buffalo Spring and Confluence Spring (if Confluence Spring was flowing). Since the discharge at Fort's Funnel was considerably less than Buffalo Spring, the exact relationship between the Wet and Dry Prongs and Fort's Funnel remained problematic even after recovering numerous qualitative traces.

QUANTITATIVE TRACES

Table 1 is a summary of the quantitative tracer tests performed in the Buffalo Spring basin using the RME filter fluorometer. A dual channel RME, capable of recovering rhodamine and fluorescein simultaneously, was placed at Buffalo Spring during all the traces. Single channel RMEs, rhodamine sensitive only, were placed in Fort's Funnel and Confluence Spring only during the April, 1991 traces. Figure 3 shows the injection points, recovery points, and straight line travel routes for each trace and Table 2 summarizes the results of each trace.

Table 1. Summary of quantitative dye traces performed in the Buffalo Spring basin using the RME.

Trace number	Injection date	Injection site	Recovery point(s)	Dye used/ Quantity
DP100	4-10-91	Dry Prong base flow sink point	Buffalo Spring Confluence Spring Fort's Funnel	Rhod. WT/ 150g
DP103	4-13-91	Dry Prong base flow sink point	Buffalo Spring Confluence Spring Fort's Funnel	Rhod. WT/ 150g
DP332	11-28-90	Dry Prong base flow sink point	Buffalo Spring	Rhod. WT/ 119g and Fluor./ 150g
MB102	4-12-91	Mill Branch quarry ponor	Buffalo Spring Confluence Spring Fort's Funnel	Rhod. WT/ 150g
MB103	4-13-91	Mill Branch quarry ponor	Buffalo Spring	Fluor./ 150g
RHSE101	4-11-91	Raymer Hol. SE terminal sink point	Buffalo Spring	Fluor./ 150g
RHSE105	4-15-91	Raymer Hol. SE terminal sink point	Buffalo Spring Confluence Spring Fort's Funnel	Rhod. WT/ 150g
WP330	11-26-90	Wet Prong base flow sink point	Buffalo Spring	Rhod. WT/ 119g
FF330	11-26-90	Fort's Funnel cave stream	Buffalo Spring	Fluor./ 100g

Table 2. RME-determined results of the Buffalo Spring basin quantitative traces.

Trace number	*Recovery point(s)	Apparent travel distance (meters)	Discharge (l/s)	Time to leading edge (hours)	Time to peak conc. (hours)	Approx. peak conc. (mg/l)
DP100	BS	4320	875	12.83	14.38	.002
	CS	4000	150	11.13	14.18	.005
	FF	3120	395	8.23	9.38	.004
DP103	BS	4320	950	10.00	11.90	.002
	CS	4000	460	8.90	10.65	.006
	FF	3120	770	5.75	7.50	.010
DP332	BS	4320	280	28.75	34.75	.007
MB102	BS	3980	940	11.27	12.37	.004
	CS	3630	360	9.77	11.02	.011
	FF	2780	700	7.67	8.37	.024
MB103	BS	3980	960	9.50	10.50	-
RHSE101	BS	4850	930	14.72	17.22	-
RHSE105	BS	4850	1090	12.25	16.75	.0005
	CS	4600	330	11.85	14.75	.001
	FF	3750	595	7.25	11.85	.0025
WP330	BS	2000	280	11.92	13.82	.100
FF330	BS	1200	280	18.20	20.75	-

*Buffalo Spring = BS
Confluence Spring = CS
Fort's Funnel = FF

November Traces

Three quantitative traces were recovered at Buffalo Spring in November, 1990. Flow conditions were low and relatively stable during this period, and Confluence Spring was not flowing. Figures 4 and 5 show the recovery curves of traces initiated simultaneously from Fort's Funnel and the Wet Prong terminal sinkpoint. The dye slugs were recovered using both a dual channel RME and an automatic sampler. Surprisingly, the rhodamine injected in the Wet Prong (WP330 trace) arrived at Buffalo Spring more than six hours before the fluorescein from the much closer Fort's Funnel (FF330 trace) (Figures 4 and 5, and Table 2). This shows that a primary flow-route exists between the Wet Prong sink and Buffalo Spring with a gradient that is significantly steeper than the flow-route between Fort's Funnel and Buffalo Spring. Consequently, a difference in head must exist between this newly discovered trunk conduit carrying Wet Prong water and the Dry Prong trunk visible at Fort's Funnel. Enough of the Wet Prong trunk is apparently pirated by the Dry Prong trunk above Fort's Funnel to be detected using qualitative dye tracing methods, but not enough to cause a noticeable secondary dye slug to appear at Buffalo Spring while using quantitative methods.

The RME results (Figure 4) compared favorably with the ISCO sampler/Shimadzu spectrofluorophotometer results (Figure 5). The ISCO sampler, which was programmed to draw a sample hourly, failed to sample the peak rhodamine concentration. The higher resolution RME data shows that the peak rhodamine concentration was considerably greater than what was determined by the ISCO/Shimadzu methods. Figures 4 and 5 prove that the RME is a capable alternative to conventional dye recovery methods. RME and ISCO/Shimadzu results were also similar for a simultaneous two-dye trace from the Dry Prong sinkpoint to Buffalo Spring (DP332).

April Traces

Six quantitative traces--two using fluorescein and four using rhodamine--were performed in the Buffalo Spring basin in April, 1991. Flow conditions were much higher than in November and fluctuated due to several moderate rainfall events received during the study period. During the six day study period the discharge was measured, using a tape measure, a survey stick, and a Marsh-McBirney flow meter, five times at Buffalo Spring, four times at Fort's Funnel, and five times at Confluence Spring. Discharges listed in Table 2 for each dye slug at each recovery site were interpolated from these measurements and are presumed to be only moderately accurate.

If automatic sampling had been used as the dye recovery method at all three sites for six days with a sampling interval of one hour, it would have required changing 432 sample bottles and

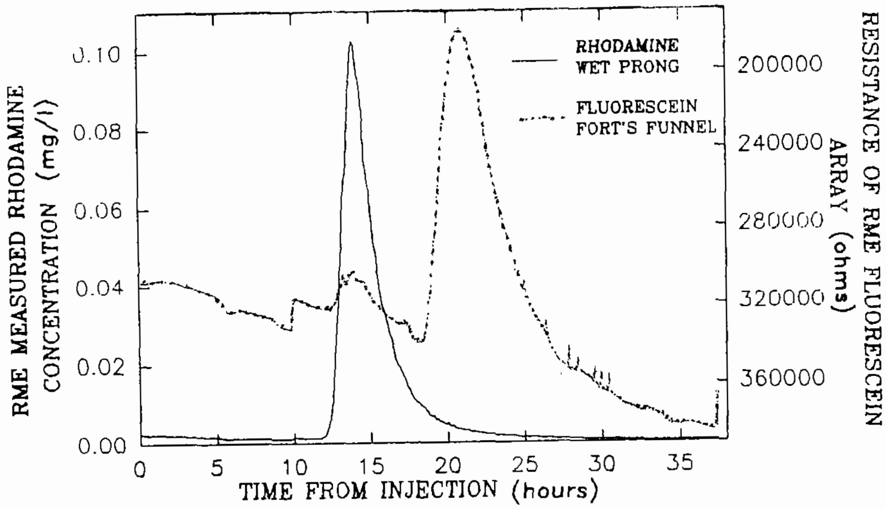

Figure 4. Recovery curves of FF330 and WP330 simultaneous traces created from RME data.

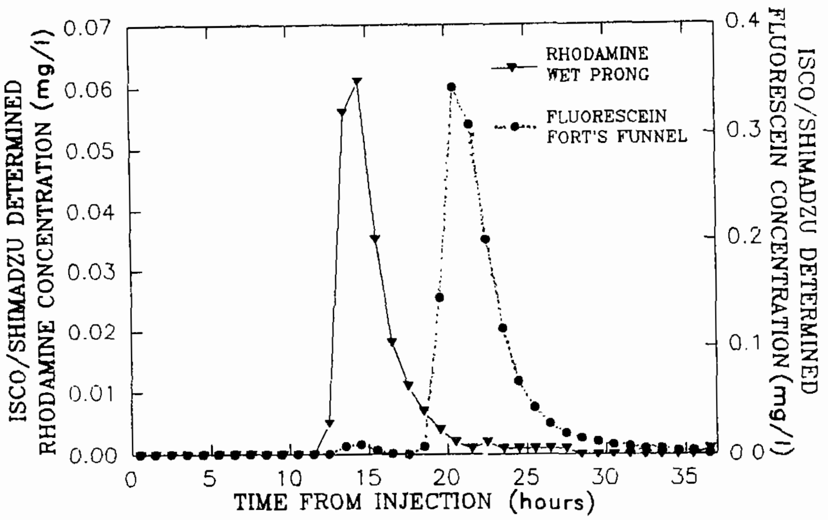

Figure 5. Recovery curves of FF330 and WP330 simultaneous traces created from automatic sampler/spectrofluorometer results.

analyzing 576 samples on the spectrofluorometer. All this toil would have produced only mediocre results because each dye slug would have been sampled only a few times (due to temporal compactness of the slugs) and only a vague picture of a slug's true shape would have resulted. The ten (or five) minute sampling frequency of the RME insures that a more realistic view of the dye recovery curve will be recorded.

MB102 Trace

Rhodamine trace MB102 was initiated at a discrete sinkpoint in Mill Branch and was recovered at Fort's Funnel, Confluence

Spring, and Buffalo Spring. The resultant recovery curves are shown in Figure 6. It is apparent in that the peak concentrations of Confluence Spring and Buffalo Spring are much lower than Fort's Funnel. The Wet Prong trunk converges with the Dry Prong trunk somewhere between Fort's Funnel and Buffalo Spring and dilutes the dye-laden Dry Prong waters. Further longitudinal dispersion was probably also a factor in this reduction. Hubbard et al. (1982) contains an excellent description of dye dispersion in streams during quantitative traces. Evidently, an input from the Wet Prong trunk exists between Fort's Funnel and Confluence Spring as well, and it is responsible for the sizable drop in concentration at Confluence Spring. Helpful "black box" type models of conduit systems like these were presented and discussed by Brown (1973).

The total mass of dye recovered at Fort's Funnel during the MB102 trace was 122g, or 81% of the 150g injected. Only 45% of that 122g was recovered at the two terminal springs; the remaining 55% was unaccounted for. Since the total discharge of the Buffalo Spring system was above average and increasing, a plausible explanation is that dye-laden conduit water moved into diffuse storage adjacent to the conduit--like river bank storage. Atkinson et al. (1973) describes this analogy in some detail. Significantly, very strong positive results for rhodamine and fluorescein were still being found on the passive dye detectors at Buffalo Spring under low flow conditions more than three months later. It is possible that dye in conduit adjacent diffuse storage was slowly released as the summertime base flow condition was approached. Other factors that may have contributed to the apparent dye loss may have been adsorption or the use of inaccurate (too small) discharge values.

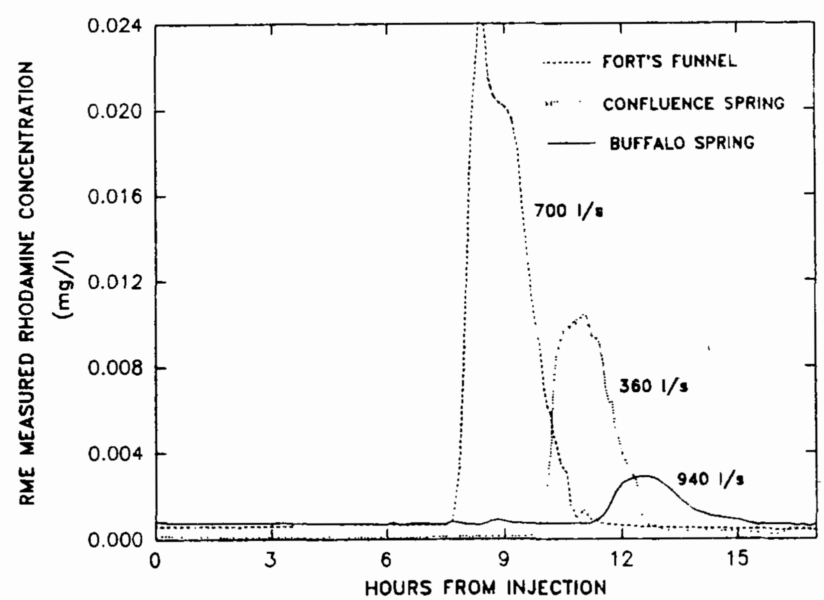

Figure 6. Recovery curves from MB102 trace.

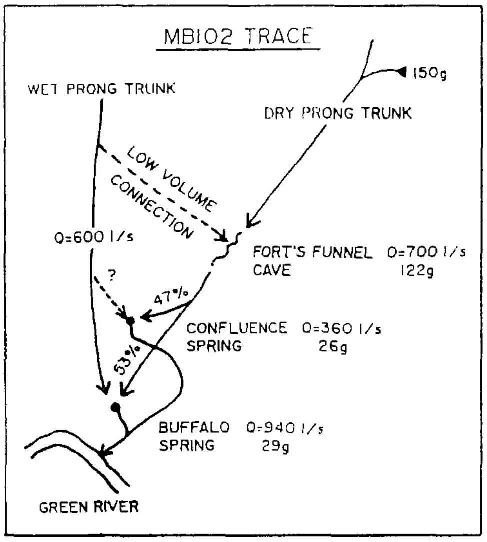

Figure 7. Summary of MB102 trace.

Figure 7 summarizes the dye recovery results for the MB102 trace. The percentage of dye (and flow) going to each spring was computed by considering the total mass of dye recovered at both terminal springs 100%, then the mass recovered at either one of the springs was divided by the total recovered at both springs and multiplied by 100. The discharge of the Wet Prong trunk was computed by subtracting the discharge at Fort's Funnel (which was assumed to be the entire Dry Prong trunk flow) from the combined Confluence Spring and Buffalo Spring discharges. Based on this, approximately 47% of the discharge from Fort's Funnel resurged at the overflow route--so about 329 l/s of Confluence Spring's 360 l/s discharge came from the Dry Prong trunk. The remainder of the Dry Prong trunk flow and nearly all the Wet Prong trunk flow resurged at Buffalo Spring.

DP103 Trace

Figure 8 shows the recovery curves at the three recovery sites for the DP103 dye trace. When compared with the recovery curves from the MB102 trace, in which the same amount of rhodamine was injected, several differences are discernable: the peak concentrations are all lower, the travel times are less, and the slugs are more dispersed longitudinally. Because the discharge was greater during this trace, the first two are believable--even though the reduction in concentration was larger than such an increase in discharge would warrant. An increased longitudinal dispersion, however, is the exact opposite of what would typically be expected for a trace initiated under higher flow conditions.

The reason the DP103 slugs were more dispersed than the MB102 slugs may be related to the fact that DP103 dye entered the subsurface through three widely spaced sequential ponors instead of through one discrete ponor. All three Dry Prong traces were

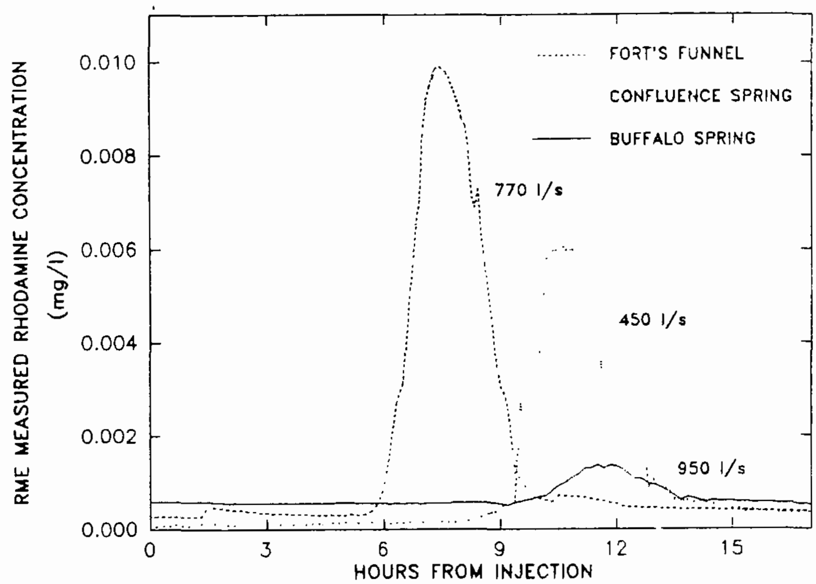

Figure 8. Recovery curves from DP103 trace.

initiated from the same point--just above the Dry Prong base flow terminal sinkpoint. However, the flow conditions were very different for each trace: the terminal sinkpoint for the DP332 trace was the base flow sinkpoint, the terminal sinkpoint for the DP100 trace was about 600 meters downstream at a ponor called Kelly's Cut-off, and for the DP103 trace the terminus of surface flow was a huge ponor 250 meters further downstream called Norain Cave (Figure 3). Dye from the DP103 injection entered the subsurface at all three of these major ponors. As a result of this the injected slug was split into three separate slugs, each with a slightly different route to follow at first. Flow from the three separate inputs eventually reunited and the three slightly out of phase dye slugs were fused back together--slightly more dispersed and with a lower amplitude than a single input slug would have been.

Only 75g of the 150g of dye injected (50%) was recovered at Fort's Funnel. The amount of that dye which was recovered at the terminal springs was 73%. The results, summarized in Figure 9, suggests that 55% of the total discharge passing Fort's Funnel went to the Confluence Spring (about 423 l/s) and 45% went to Buffalo Spring (about 347 l/s). The remainder of each was supplied by the Wet Prong trunk.

When the DP103 trace results (Figure 9) are compared to the MB102 results (Figure 7) several important insights into the behavior of this aquifer may be gleaned: the Confluence Spring waters are mostly derived from the Dry Prong trunk, and the Wet Prong trunk is not well connected to Confluence Spring. Thus the Confluence Spring is predominately an overflow spring for the Dry Prong trunk. If the discharge were increased in both the trunks simultaneously, hydraulic damming by Wet Prong waters, which basically have no place else to go but Buffalo Spring, would cause a decrease in the percentage of Dry Prong water resurging

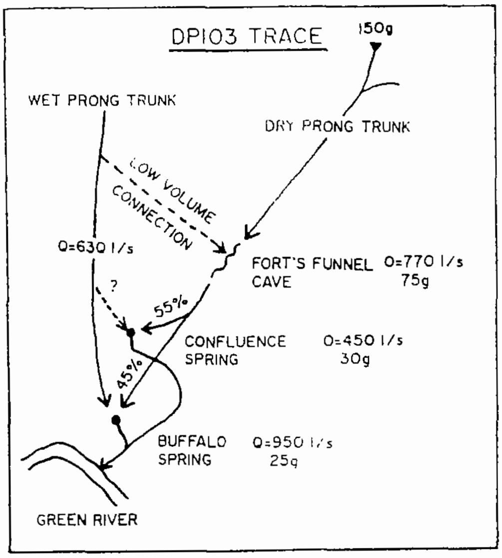

Figure 9. Summary of DP103 trace.

at Buffalo Spring and an increase in the percentage overflowing at Confluence Spring.

Repeated Quantitative Traces as a Predictive Tool

The input-to-resurgence travel time of a dye slug decreases with increasing discharge. Peak concentrations often decrease with increasing discharge because dilution increases, and longitudinal dispersion decreases due to decreased dye slug travel time. Figure 10 illustrates the results of a trace from Dry Prong to the terminal spring(s) repeated three times under different flow conditions. The aforementioned effects of increased discharge are very clear. Using results from these three traces, reliable predictions for almost any set of flow conditions could be made concerning travel time, peak concentration, and dispersion for a soluble contaminant accidentally injected into the Dry Prong. Mull et al. (1988) gives a detailed discussion of this important topic.

Determination of Conduit Condition

Conduit condition may be resolved even if a conduit is inaccessible by using quantitative tracing; this was done for segments of the Dry Prong trunk using the RME. The discharge of a vadose conduit is increased by increasing the flow velocity and/or the cross-sectional area of the channel (by increasing stage). The only way to increase the discharge of a phreatic conduit, since it is completely full and stage cannot be increased, is by increasing the flow velocity. So, when log discharge (X) is plotted versus the log travel time (Y) for a series of traces through a phreatic conduit the result would be a line with a slope of nearly -1 (Smart, 1981). A plot of traces through a vadose conduit would be a line with a slope of less than -1.0 (but probably greater than -0.3). Figure 11 shows first order linear regressions of log discharge versus log travel time for the three Dry Prong traces recovered by the RME for, the entire Dry Prong, the segment of the Dry Prong trunk upstream from Fort's Funnel, and the segment of Dry Prong trunk downstream from Fort's Funnel. Judging from their slopes, which are admittedly based on a paucity of data, the segment downstream from Fort's Funnel is apparently mostly phreatic and the segment upstream is mostly vadose.

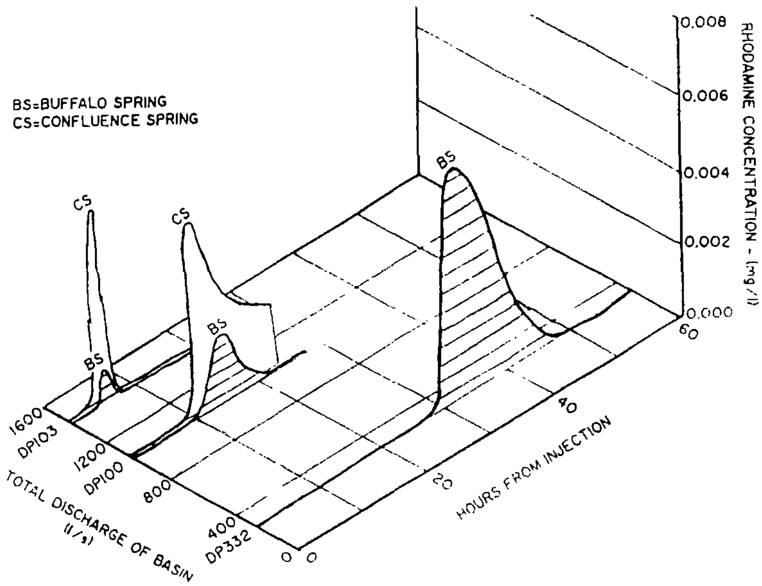

Figure 10. Recovery curves of Dry Prong to the terminal spring(s) traces.

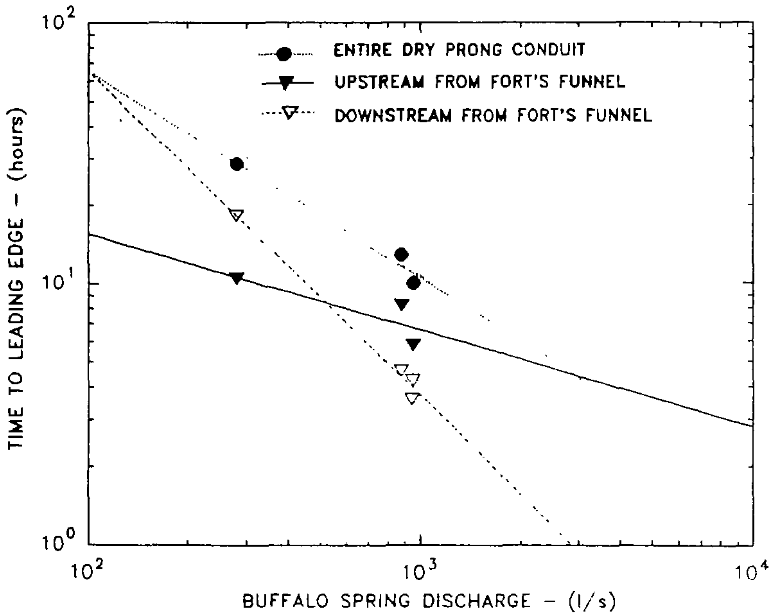

Figure 11. Log-Log plots of discharge vs. time of travel for various segments of the Dry Prong trunk.

Hydrologic Structure of Buffalo Creek

A pictorial summary of the hydrologic structure of the Buffalo Spring karst ground water basin was generated by synthesizing all the qualitative and quantitative trace data and geomorphological data collected (Figure 12). Smart (1988) and Smart and Ford (1986) presented a structural model of the Castleguard conduit aquifer and laid the groundwork for this type of aquifer representation. Models like these could be quite useful to ground water managers charged with determining a course of action during an accidental contamination event.

SUMMARY AND CONCLUSIONS

The RME is an inexpensive alternative to conventional quantitative dye recovery methods. Extensive fieldwork in the Buffalo Spring ground water basin, including some in conjunction with traditional dye recovery methods for comparison, proved that the RME is a useful dye quantification tool for field study. Through use of the RME, subtle details concerning the hydrology of Buffalo Spring basin were recognized and described including several previously unknown ground water flow routes. Also generated was new information about the relationships of the primary spring and the over-flow spring to the two primary feeder trunks and the response of aquifer transmissivity to changes in discharge. Interpretation of RME data helped to identify the phreatic and the vadose portions of the Dry Prong trunk conduit. A structural model of the Buffalo Spring basin was produced using all the available dye tracing data.

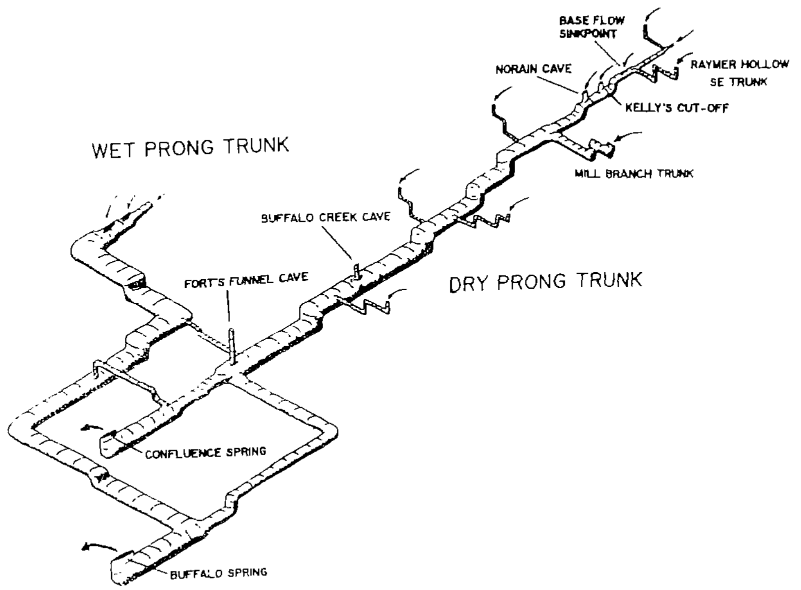

Figure 12. Hydrologic structure of the Buffalo Spring ground water basin.

ACKNOWLEDGEMENTS

Dr. Ralph Ewers of Eastern Kentucky University provided to me the initial idea of developing a submersible fluorometer supported by a digital datalogger. Joe Meiman, my colleague at Mammoth Cave, wrote the datalogger program used by the RME and contributed many hours of assistance during development and testing. Mammoth Cave National Park provided financial support, laboratory facilities, and other resources. I am extremely grateful to them all.

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BIOGRAPHIC SKETCH

Martin Ryan is a Hydrologic Technician at Mammoth Cave National Park. Current research includes developing and implementing new quantitative fluorometric techniques to model flow in karst aquifers, delineating ground water basins north of the Green River in the Mammoth Cave area, and performing water quality monitoring. He received his BS in geology from Eastern Illinois University in 1987 and he is currently pursuing a MS in geology at Eastern Kentucky University.

Development of a Flow-through Filter Fluorometer for Use in Quantitative Dye-tracing at Mammoth Cave National Park

by Martin T. Ryan

I understand that inclusion of a turbidity measurement was contemplated for an earlier model of your flow-through fluorometer. Why did you decide not to monitor turbidity?

The nephelometer/turbidity meter prototype we were experimenting with had severe problems with water leakage. Additionally, laboratory experiments had showed that the design of the instrument, which used a jumbo LED with a Fresnel lens and two photoresistors, was adequate only for approximation of turbidity (which is itself only an approximation of suspended sediment concentration--the maligner of dye fluorescence.) Even if a successful turbidity meter had been deployed alongside the fluorometer, I do not believe that a turbidity-compensating mathematical formula could have been generated. Rather, turbidity data may have been useful in helping to explain anomalies in the fluorometric record by intuitive comparison--for example, if the mass of dye recovered during a quantitative trace is unusually low, a partial explanation may be available if the turbidity data is examined and found to be exceptionally high (eg. because a brief storm event) during the time period including the recovery of the dye slug.

We are still trying to develop a nephelometer/turbidity meter that is interfaceable with a digital datalogger, but it is not urgently required to validate data acquired through RME fluorometry -- rather it may, at times, be a helpful supplement.

Session IV:

Hydrogeology and Processes Occurring in Karst Aquifers

Stable Isotope Separation of Spring Discharge in a Major Karst Spring, Mitchell Plain, Indiana, U.S.A.

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ABSTRACT

Major-ion chemistry and isotopes of oxygen and deuterium where used to identify components of water resurging at Orangeville Rise, a large perennial spring. The Orangeville Rise basin is recharged by precipitation that infiltrates a 125 km^2 sinkhole plain and upland area developed on Mississippian limestone and clastic sequences.

In October, 1990, 53 mm of rain fell on the basin in a 40 minute period. Isotopic composition of the rain was -8.2‰ and -56‰ for $\delta^{18}O$ and δD respectively. Prior to this, at baseflow, $\delta^{18}O$ and δD of Orangeville Rise was -6.2‰ and -40‰ respectively, and specific conductance was 770 uS. Discharge increased rapidly from 0.3 m^3s^{-1} to 3.4 m^3s^{-1}, however major ion concentrations did not show a rapid drop which would have been the case had an influx of rainwater been responsible for the increase in discharge. Rather, the most dilute waters were found 24 hours after peak discharge. Isotopic composition of waters also reflected this trend. Isotopic separation of storm hydrograph into pre-storm and storm water components revealed that 75% of discharging water consisted of pre-storm water.

Similar monitoring was conducted during April, 1991, when 4 rains, totaling 38 mm produced an increase in discharge from 1.6 m^3s^{-1} to 4.9 m^3s^{-1}. δD of these rains ranged from -7‰ to -45‰, and $\delta^{18}O$ ranged from -2.6‰ to -6.8‰. Again, major-ion and isotope chemistry did not reflect a rapid influx of rainwater, and isotopic hydrograph separation showed that rain water comprised only 20% of the total volume of discharge. These results seem to indicate that water stored in the epikarst may discharge at Orangeville Rise prior to the arrival of the mass of rain water.

INTRODUCTION

Discharge from springs in karst terrains responds to influxes of precipitation in many ways. The response is governed by the type of recharge (concentrated or dispersed), as well as the routes of subsurface flow within the basin (conduit or diffuse). It is also highly dependent upon the volumes of water stored in the unsaturated and saturated zones.

Historically, individuals have studied karst systems by mapping cave passageways, and by conducting dye-tracing studies. Analysis of storm hydrographs from karst springs has also been performed, concentrating on recession limb behavior (Mangin, 1974, 1975; Torbarov, 1976; Milanovic, 1976; and Atkinson, 1977).

Major-ion analyses have proven useful in discerning properties of a karst system, and have been used by numerous investigators, (e.g., *Shuster and White*, 1971; Ford and Ewers, 1978; Friedrich and Smart, 1982; Gunn, 1983; and Scanlon and Thrailkill, 1989). Changes in water quality associated with storm hydrographs of karst springs have been helpful in interpreting sources of water which contribute to discharge (Williams, 1983).

Stable isotopes have been used to show that new rainwater can displace old water in storage in karst systems (Bakalowicz et al., 1974). The two were found to mix in different proportions to create fluctuations in $\delta^{18}O$ across the storm hydrograph. Dreiss (1989) separated storm discharge from karst springs in Missouri, into new event water and pre-storm water on the basis of calcium and magnesium ion concentrations.

This study looks at water quality changes associated with storm hydrographs at Orangeville Rise, a perennial spring in a karst region of Indiana. Goals of this study include quantifying the proportion of rainwater contributing to storm discharge through the use of oxygen and deuterium isotopes, accounting for observed water chemistry changes across the hydrograph, and attempting to identify subsurface storage zones which contribute to discharge at Orangeville Rise.

Geologic and Hydrologic Setting of the Study Area

Orangeville Rise is a perennial spring located in south-central Indiana, approximately 150 km south of Indianapolis (Figure 1). The ground water basin supplying water to Orangeville Rise, as defined through dye-tracing studies (Murdock and Powell, 1968; Bassett, 1974), has an areal extent of 115 km^2 and lies within two physiographic provinces, the Mitchell Plain and the Crawford Upland.

The Mitchell Plain is a flat-to-gently-rolling lowland surface developed on middle Mississippian limestones of the Blue River and Sanders Groups (Figure 2). These formations consist of dense, thinly bedded, highly fractured limestones which have experienced profound solutional modification to produce features typical of karstic terrains. Indeed, the Mitchell Plain is known for its extensive karst topography expressed in a myriad of sinkholes, caverns and blind valleys.

The Crawford Upland occupies a region where alternating formations of sandstone, shale and limestone of the Mississippian Chester Series crop out at the surface. These formations are more resistant to erosion than underlying Blue River and Sanders limestones and hence form an upland surface that stands 45 to 60 m above the Mitchell Plain. The Crawford Upland is a rugged, deeply dissected surface. Karst features (i.e., sinkholes and sinking streams) have developed where stream erosion has cut through Chesterian Strata to expose Blue River limestones. As a result, the hydrologic behavior of Orangeville Rise is dictated

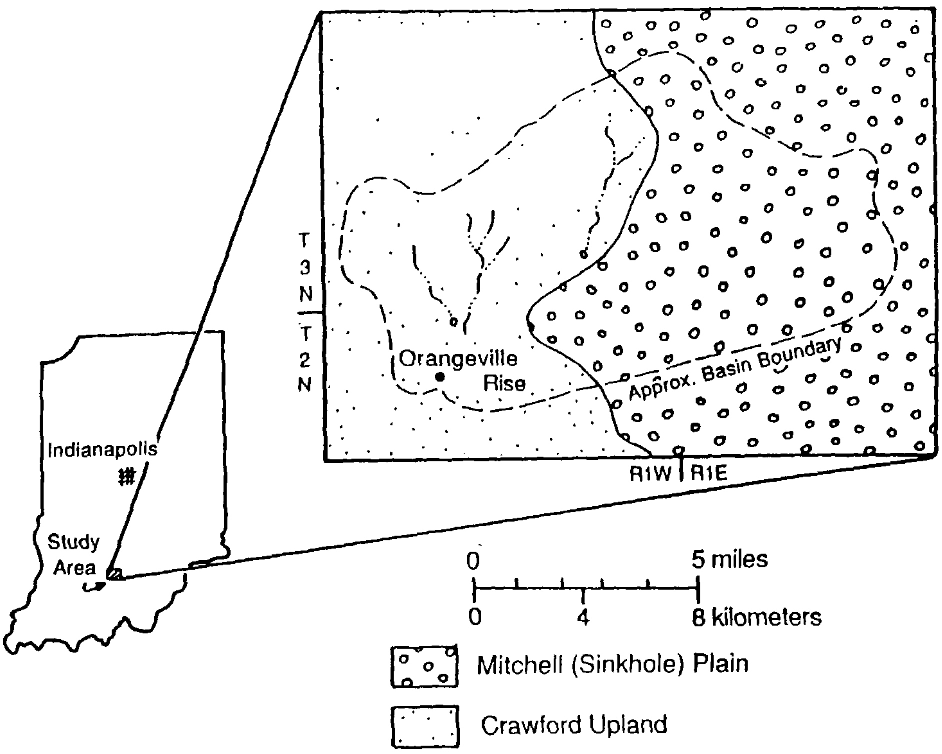

Figure 1. Location of the Orangeville Rise, and physiographic provinces within the basin.

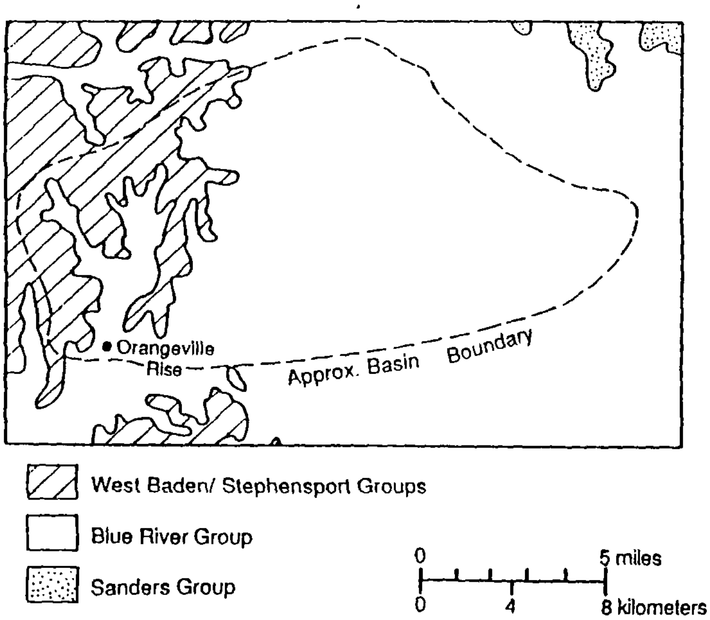

Figure 2. Geologic Map of the Orangeville Rise basin.

by the karstic nature of the Blue River bedrock.

Discharge at Orangeville Rise ranges from 0.06 m^3s^{-1} (2 cfs) to 5.1 m^3s^{-1} (180 cfs). The Rise is situated within a bedrock alcove forming a pool that is approximately 15 m in diameter, and at low-flow is 6 m deep. The Rise is fed by two vertical conduits which are 6 m long. Scuba divers descending these shafts, found them to connect to narrow horizontal conduits at their base. Discharge from Orangeville Rise forms a perennial surface stream that occupies an alluviated channel 5 m deep.

At low flow, when suspended sediment loads are low, waters in the discharge pool take on a greenish cast. After precipitation events, discharge increases rapidly. Depending on the magnitude of the storm, the pool surface may rise 3 m or more. The waters become heavily laden with suspended sediment, and a boiling of the pool surface can be seen as water is forced up through the vertical conduits.

No perennial surface streams discharge at Orangeville Rise. Rather, discharge is derived predominantly from precipitation that infiltrates soils of the sinkhole plain, with a secondary component coming from sinking intermittent streams draining the Crawford Upland.

Soils of the Orangeville Rise basin consist of clay and clay loam derived as residuum from underlying limestones, as well as residual material produced during erosional retreat of the Crawford Upland, and thin loess deposits (Ruhe, 1975). These soils are susceptible to fracturing due to their high clay content, creating macropores for rapid infiltration of precipitation to the subsurface (Wells and Krothe, 1989). Soils are up to 5 m thick, but are locally much thinner. In many places, particularly around sinkholes, soil material has been completely removed by erosion, creating another avenue for rapid subsurface recharge.

Southern Indiana has a humid-temperate climate, with a mean annual rainfall of 1130 mm. Precipitation is distributed through the year, such that there is a pronounced dry time in the fall when monthly average precipitation is 70 mm, and a wetter time in late winter through spring, when average monthly precipitation is 115 mm. July is the warmest month on average, with mean daily maximum and minimum temperatures of 90° and 64° F respectively. In contrast, January is the coldest month, when mean daily maximum and minimum temperatures are 43° and 25° F respectively.

Previous Investigations at Orangeville Rise

Bassett (1974) has identified the waters from Orangeville Rise to be calcium-bicarbonate in character, and he saw an inverse relationship between discharge and total dissolved solids. Subsequent studies have confirmed these observations (Libra, 1981; Tweddale, 1987). The carbonate aquifer(s) supplying water to Orangeville Rise have a strong conduit component, however, it is more complicated than a simple system of conduits. Based on major-ion chemistry and sulfur isotopic composition, Krothe and Libra (1983) differentiated two flow systems within the ground water basin. One is a shallow conduit system, dominated by surface flow entering the subsurface through large fractures. Discharge at Orangeville Rise responds rapidly to precipitation by water entering the aquifer via this route. A second system consists of a deeper diffuse network of interconnected fine fractures and joints which respond more slowly to inputs from precipitation.

BACKGROUND

Conceptual Model for Ground Water Storage

A simplified model illustrating water storage compartments for Orangeville Rise is shown in Figure 3. The model is simple in that it includes no impermeable bedrock or caprock, and recharge occurs only from precipitation that falls within the basin. These identified compartments could exist in many karst systems, and have important ramifications for supplying water as discharge to any karst spring.

Recharging precipitation may be temporarily stored in the vadose zone (unsaturated zone) as soil moisture, epikarst storage, diffuse vadose storage, or vadose conduit storage.

The volume of water stored as soil moisture at any one moment is highly variable and dependent on soil properties of depth, texture and composition, as well as on the timing and volume of precipitation events. After extensive drought conditions, soil moisture will drop to a minimum. When precipitation events are large, or follow in rapid succession, soils may become completely saturated. Recharging water from precipitation moves through soil horizons as an advancing front, acting like a piston to displace and replace previous soil moisture.

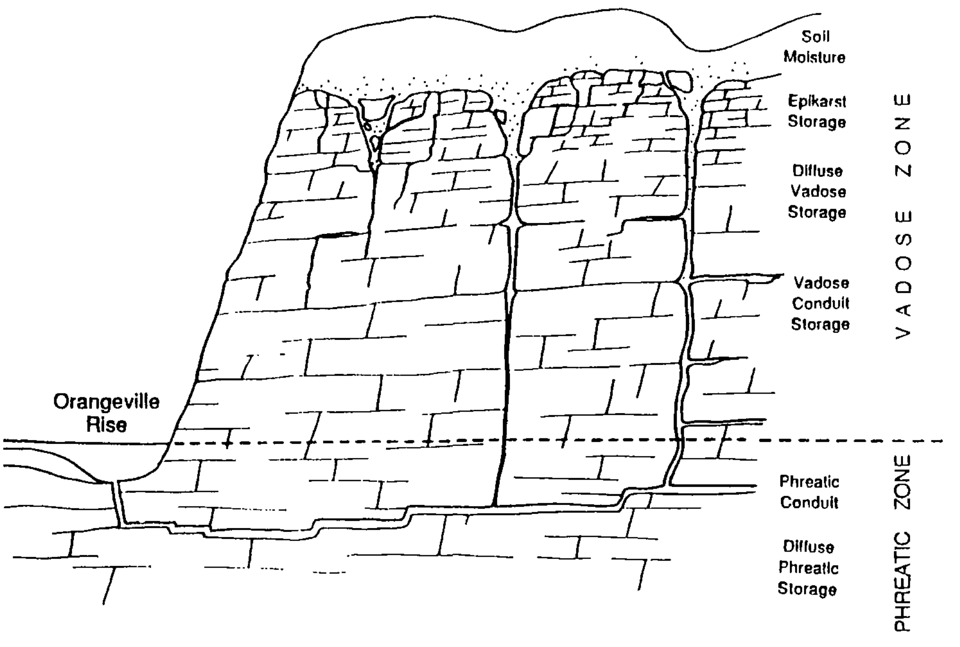

Figure 3. Model of water storage compartments of the vadose and phreatic zones in the vicinity of Orangeville Rise.

Water which passes through the soil zone may reside for some time in epikarst storage. Mangin (1974, 1975) defined epikarst as a region in the upper weathered layers of rock at the base of soil horizons, lying above the permanently saturated (phreatic) zone. Williams (1983) described the formation of epikarst as follows: Water flowing through soil dissolves CO_2 from the soil atmosphere, and becomes aggressive toward underlying limestones. As infiltrating water encounters bedrock, its ability to dissolve the rock is at a maximum. These aggressive waters preferentially flow along prominent fractures, dissolving bedrock along the fracture, making thus a zone of maximum dissolution, and producing substantial secondary porosity. As the water flows deeper into the fracture, its capacity to dissolve is diminished, and the fracture narrows with depth, limiting the rate of water transmission to deeper zones. Following precipitation events of high intensity and/or large volume, a substantial body of water can accumulate within the epikarst, and form a region where water saturation exists, resulting in a perched water table.

Water will drain from the epikarst along solutionally modified fractures, but will also move into the rock mass through the system of interconnected fine joints and fractures. Depending on the amount of primary porosity and the extent of fracturing and jointing within the bedrock, the volume of water within diffuse vadose storage may be minimal or quite large. The degree of interconnectedness of the joints and fractures will also govern the rate at which water can move into and out of this compartment.

There exists a transition zone between diffuse vadose and diffuse phreatic storage, as well as between vadose conduit and phreatic conduit storage. It is well documented that changes in water table elevation in karst terrains can be quite dramatic, and can occur on a seasonal cycle, or following precipitation events. The magnitude of change in conduits can be on the order of tens of meters or more, and can occur at an alarming rate after heavy rainfall. The response is slower and more subdued within the diffuse bedrock compartments. Because conduits flood quickly and drain more rapidly than the diffuse zones, the two areas do not appear to be in equilibrium, and it is therefore difficult to speak of a water table in karst terrains. Nonetheless, it is helpful to visualize water storage in terms of the vadose and phreatic zones since it lends an element of elevation or position within the subsurface.

Stable Isotopes in Ground Water Studies

Stable isotopes of oxygen and hydrogen bound in a water molecule, under low temperature conditions, are altered only by physical processes such as diffusion, dispersion, mixing and evaporation, and may therefore behave in a conservative manner. As a result, these isotopes can act as tracers and their analysis may aid in determining the geochemical history of a water mass. Stable isotopic analysis of waters may also provide information on movement and mixing of water bodies, provided that the isotopic composition of each water body differs by at least the analytical error of isotopic ratio identification.

In this study, stable isotopes of oxygen and hydrogen are used in an attempt to identify two time-source components of discharge at Orangeville Rise: 1) water that existed within the basin prior to a storm event and 2) new rainwater that fell within the basin.

If it is assumed that discharge at Orangeville Rise is derived by simple mixing between two components: pre-storm water (Q_{PS}) and rainwater (Q_R), then two mass balance equations can be written that describe the water flux and the isotope flux at the Rise:

$$Q_M = Q_R + Q_{PS}$$

$$Q_M \delta_M = Q_R \delta_R + Q_{PS} \delta_{PS}$$

where Q_M is the total measured discharge at any instant in time and is defined by the sum of the two components. Delta notation represents the ^{18}O or deuterium (D) composition of the instantaneously measured discharge (δ_M), the rainwater (δ_R), or pre-storm water (δ_{PS}). By combining these two equations, a third equation can be written which identifies the rainwater contribution to discharge at any instant in time, in terms of the isotopic composition of the measured discharge, pre-storm water and rainwater.

$$Q_R = Q_M \frac{(\delta_M - \delta_{PS})}{(\delta_R - \delta_{PS})}$$

To use this equation as a means of separating discharge into rainwater and pre-storm water components, the isotopic composition of rainwater must be significantly different than pre-storm water, and the pre-storm water should have an identifiable and uniform isotopic composition throughout the basin. This technique has been applied to surface streams by Fritz et al. (1976) and Sklash and Farvolden (1979), where the pre-storm water was interpreted to be ground water.

In this report, stable isotopic composition of water is reported relative to standard mean ocean water (SMOW), in parts per thousand (or premil) notation, such that:

$$\delta\,(^{18}O, D)_{SMOW} = \left(\frac{R_{sample} - R_{standard}}{R_{standard}}\right) \times 1000$$

where R is the ratio of D/H or $^{18}O/^{16}O$, depending on which ratio is being identified.

FIELD INVESTIGATION

The field investigation was conducted in two stages. During the first stage, water was sampled on a bi-weekly schedule to establish baseflow chemical characteristics. It was critical during this stage to determine the baseflow or pre-storm isotopic composition to be used in the mass-balance equations. Water from domestic wells was also sampled during this stage of the investigation to establish isotopic composition of the ground water within the basin.

The second stage of sampling concentrated on short-term changes in water chemistry associated with changes in discharge brought on by two storm events in the Orangeville Rise basin: in October, 1990, corresponding to the driest time of the year; and in April, 1991, during the wettest time of the year, when recharge to the carbonate aquifer is a maximum (Bassett, 1974). Sampling frequency during this stage of monitoring was keyed to the rate of discharge increase and decrease: every two to four hours during the time of rapid discharge increase, peak flow and initial discharge recession, and less frequently during later stages when discharge decreased less rapidly.

RESULTS

Results from the bi-weekly sampling confirm the results of previous investigations. Discharge at Orangeville Rise is calcium-bicarbonate in character, and there is an inverse relationship between discharge and total dissolved solids. It was also found that as discharge approached baseflow, the water's isotopic composition repeatedly approached -40 ‰ and -6.2 ‰ for δD and $\delta^{18}O$ respectively. These values were used as the isotopic signature of pre-storm water in hydrograph separation of spring discharge.

Ground water isotopic composition was found to be quite uniform, even though there was variable major-ion chemistry in the basin (Table 1). The average δD and $\delta^{18}O$ of ground water was found to be -44 ‰ and -6.6 ‰ respectively, and is similar to that of the average isotopic composition of rainwater in southern Indiana (Sheppard et al., 1969).

October, 1990 Storm

On October 4, 1990, 53 mm of rain fell on the Orangeville Rise basin in a 40-minuter period. Rain was sampled from two locations in the basin to check for areal variability in the amount and isotopic composition of the rain. The amounts of rainfall at each station were identical, and isotopic compositions were within analytical error, and were averaged.

Table 1. Selected chemical data from domestic wells located in the Orangeville Rise basin. (n = 26)

PARAMETER	RANGE	AVERAGE
Specific Conductance (uS)	408 to 1985	668
Calcium (mg/l)	70 to 388	113
Sulfate (mg/l)	13.9 to 976	125
δD_{smow} (‰)	-43 to -46	-44
$\delta^{18}O_{smow}$ (‰)	-6.4 to -6.8	-6.6

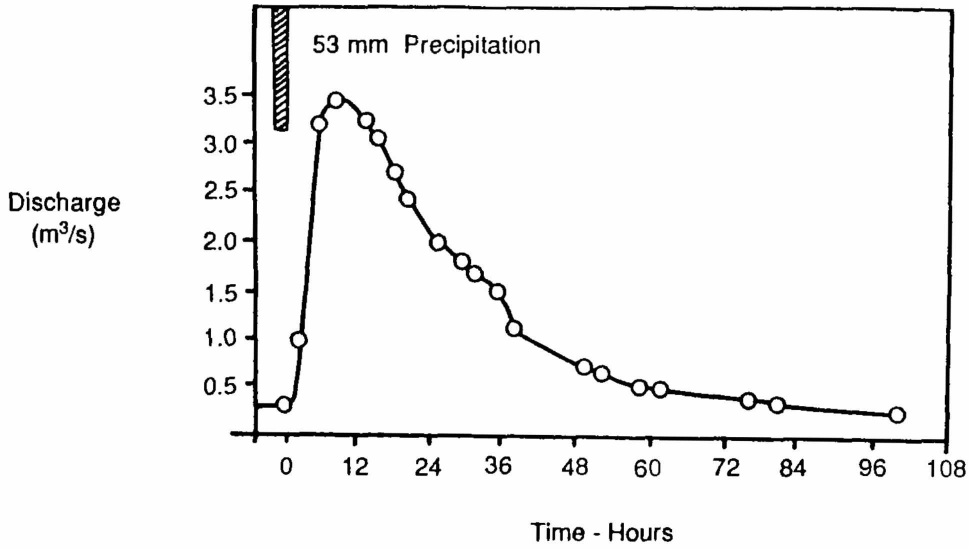

Figure 4. Storm hydrograph for Orangeville Rise, October 4, 1990.

Isotopic composition of the rainwater was −56‰ and −8.2‰ for δD and $\delta^{18}O$ respectively. Isotopic composition of the rain may have shifted during the course of the 40 minute storm, however since the duration of the storm was short in comparison to variable travel time for water in the entire basin, the potential shift was considered insignificant for the purpose of this study, and a bulk isotopic composition was determined.

Prior to this storm, baseflow discharge at Orangeville Rise was 0.3 $m^3 s^{-1}$ and specific conductance of discharging waters was 770 uS. Calcium, magnesium, sulfate, and bicarbonate concentrations were at the highest levels measured at any time during the study.

Discharge increased rapidly to a maximum of 3.4 $m^3 s^{-1}$ within 6 hours after the storm (Figure 4). There was no precipitation during the next four days, and discharge underwent a steady exponential decrease to 0.3 $m^3 s^{-1}$, with on exception between 26 and 36 hours, where there was a small perturbation in the otherwise smooth recession.

Results of chemical analyses for waters collected across rising and receding limbs of the hydrograph are shown in Figure 5. It can be seen that major-ion concentrations slowly decreased during the first 8 hours, even though discharge increased dramatically. Concentrations were not at their minimum until 24 hours after peak discharge, when the HCO_3 concentration had decreased to 75% of its baseflow value and SO_4 dropped to only 21% of its baseflow value. These minimums were preceded by a small yet significant increase in ionic concentration for several species, which corresponds to the small irregularity in recession. Ion concentrations remained low during much of the recession, and only started to recover in the last 24 hours of monitoring.

Isotopic ratios remained fairly constant during the rapid rise in discharge, and showed a small decrease during peak

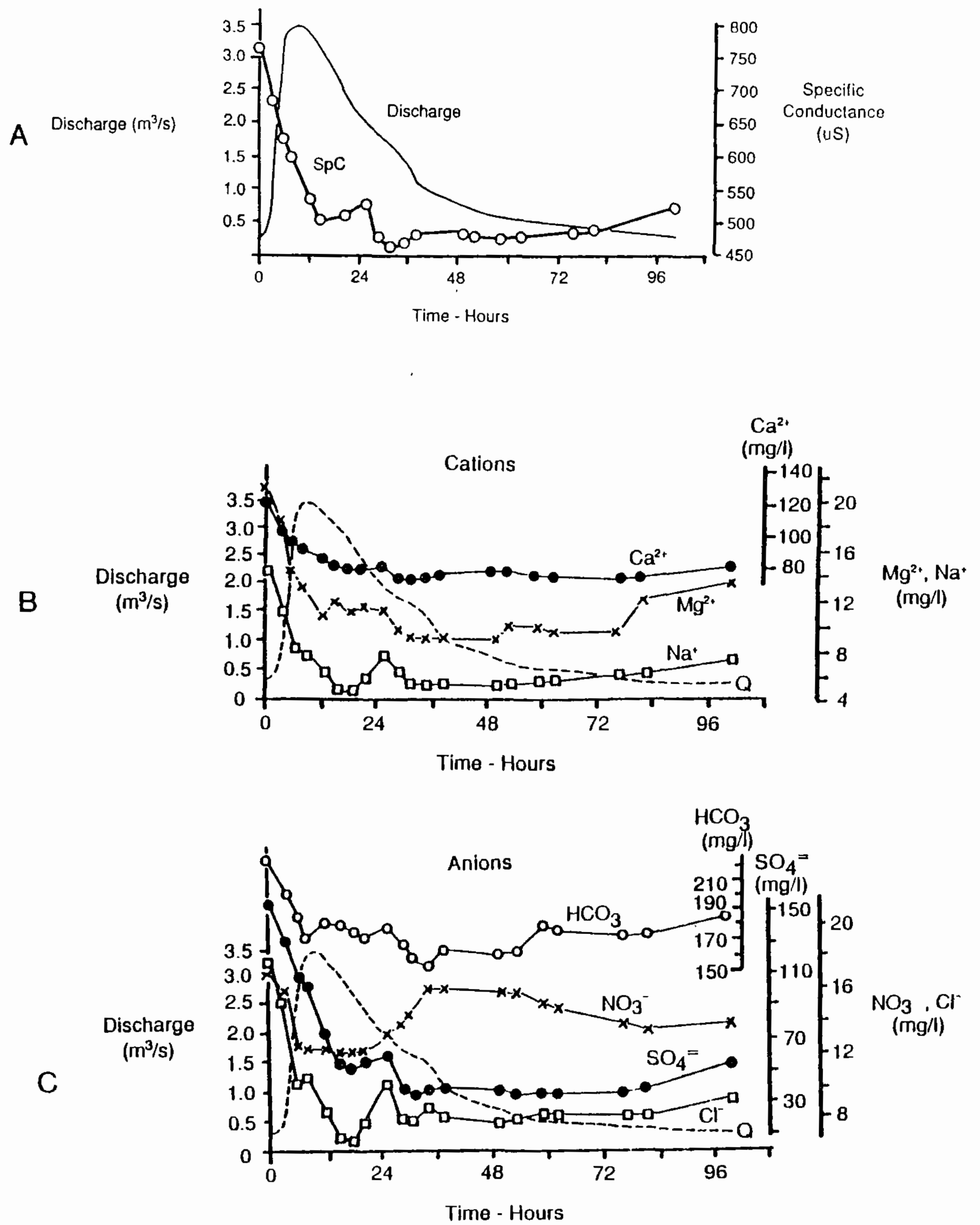

Figure 5. Chemical data from Orangeville Rise, October 4-8, 1990: A) specific conductance, B) cation concentrations, and C) anion concentrations.

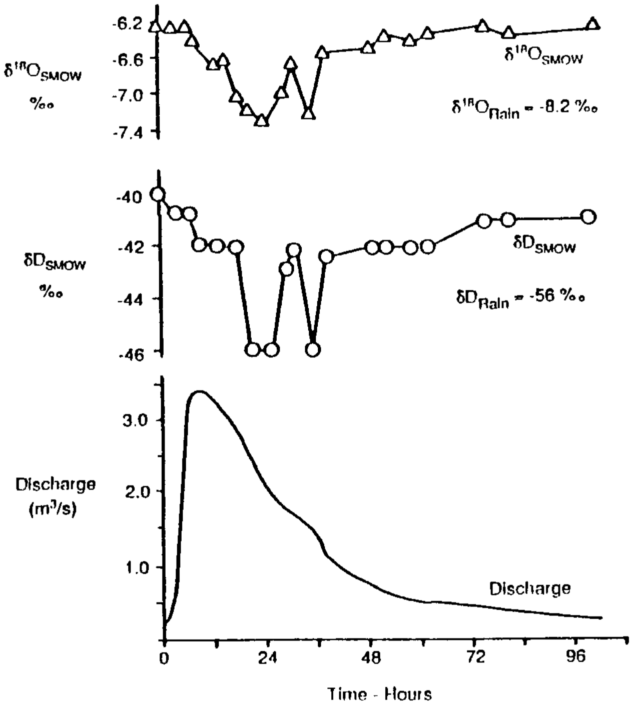

Figure 6. Isotopic composition of discharge at Orangeville Rise, October 4-8, 1990.

discharge and initial discharge recession (Figure 6). Oxygen and deuterium shifts mirrored one another during a period of rapid isotopic fluctuations between 18 and 40 hours, when two pulses of isotopically light water discharged at the Rise. During much of the discharge recession, isotopic composition remained steady, and approached that of baseflow values.

April 1991 Storm

Chemical variability of discharge was monitored during a second storm event in the spring, 1991 (Figure 7). Because of frequent rainfall in the winter and spring, discharge at Orangeville Rise does not achieve baseflow conditions. Therefore, to establish pre-storm conditions for this sampling, Orangeville Rise was monitored for 6 days prior to the storm event. Major-ion concentrations and isotopic composition of discharge remained fairly steady at the values illustrated in Figures 8 and 9 for time equal to zero-hours.

As in the October storm, precipitation was collected at two locations within the basin. Again, the amount and isotopic composition of the two rainwater samples were similar, and were averaged. Rainfall history during this event was much more complicated than the October storm, and is outlined in Table 2. Because of the complex rainfall history, discharge response was significantly different from that observed in October (Figure 7).

Table 2. Isotopic composition of rainfall, April 12-15, 1991, Orangeville Rise basin.

				Volume Averaged	
Date Time	Amount (mm)	δD (‰)	$\delta^{18}O$ (‰)	δD (‰)	$\delta^{18}O$ (‰)
4-12 / 1930	3	-12	-2.6	-12	-2.6
4-12 / 2330	7	-18	-3.4	-16	-3.4
4-13 / 0430	19	-7	-2.3	-12	-2.7
4-14 / 0645	3	-9	-2.7	-11	-2.7
4-15 / 0500	6	-45	-6.8	-16	-3.3

Discharge increased rapidly from 1.6 m^3s^{-1} to 4.2 m^3s^{-1} in 12 hours, in response to 29 mm of precipitation. During the next 24 hours discharge leveled off, then rose again following a short rainfall, to a maximum of 4.9 m^3s^{-1}. After the peak, discharge slowly decreased, following an irregular linear recession.

Behavior of major-ion concentrations during this storm were markedly different from that observed during the October monitoring (Figure 8). Concentration of several ions increased during an increase in discharge. Calcium and magnesium increased by 5 and 10% respectively during the first 6 hours, while sodium fluctuated between 8 and 12 mg/l during the first 25 hours. Bicarbonate remained constant during the first 12 hours of monitoring when discharge increased from 1.6 m^3s^{-1} to 3.2 m^3s^{-1}, and after an initial decrease, sulfate concentration increased to a maximum at 18 hours. There was also a jump in ionic concentrations at 25 hours when Ca^{2+}, Na^+ and $SO_4^=$ increased by 13 to 31% from the previous sample. Minimum specific conductance and ionic concentrations occurred 4 hours after peak discharge. Concentrations increased from this minimum, however the trend was marked by erratic fluctuations.

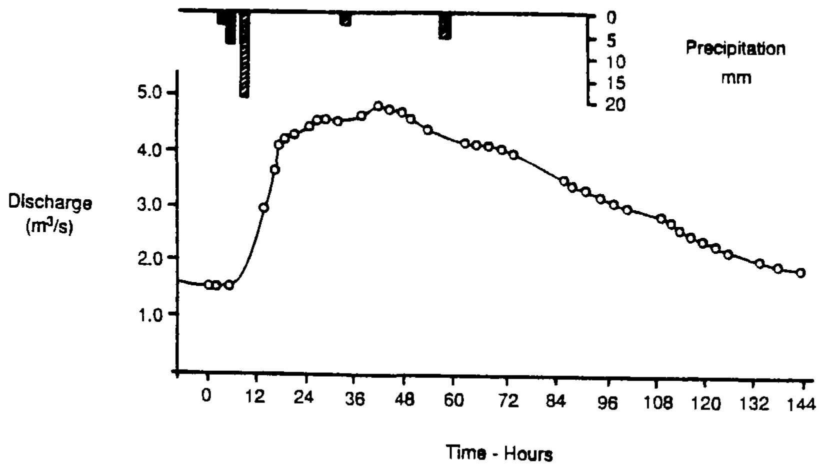

Figure 7. Storm hydrograph and rainfall history for Orangeville Rise, April 12-18, 1991.

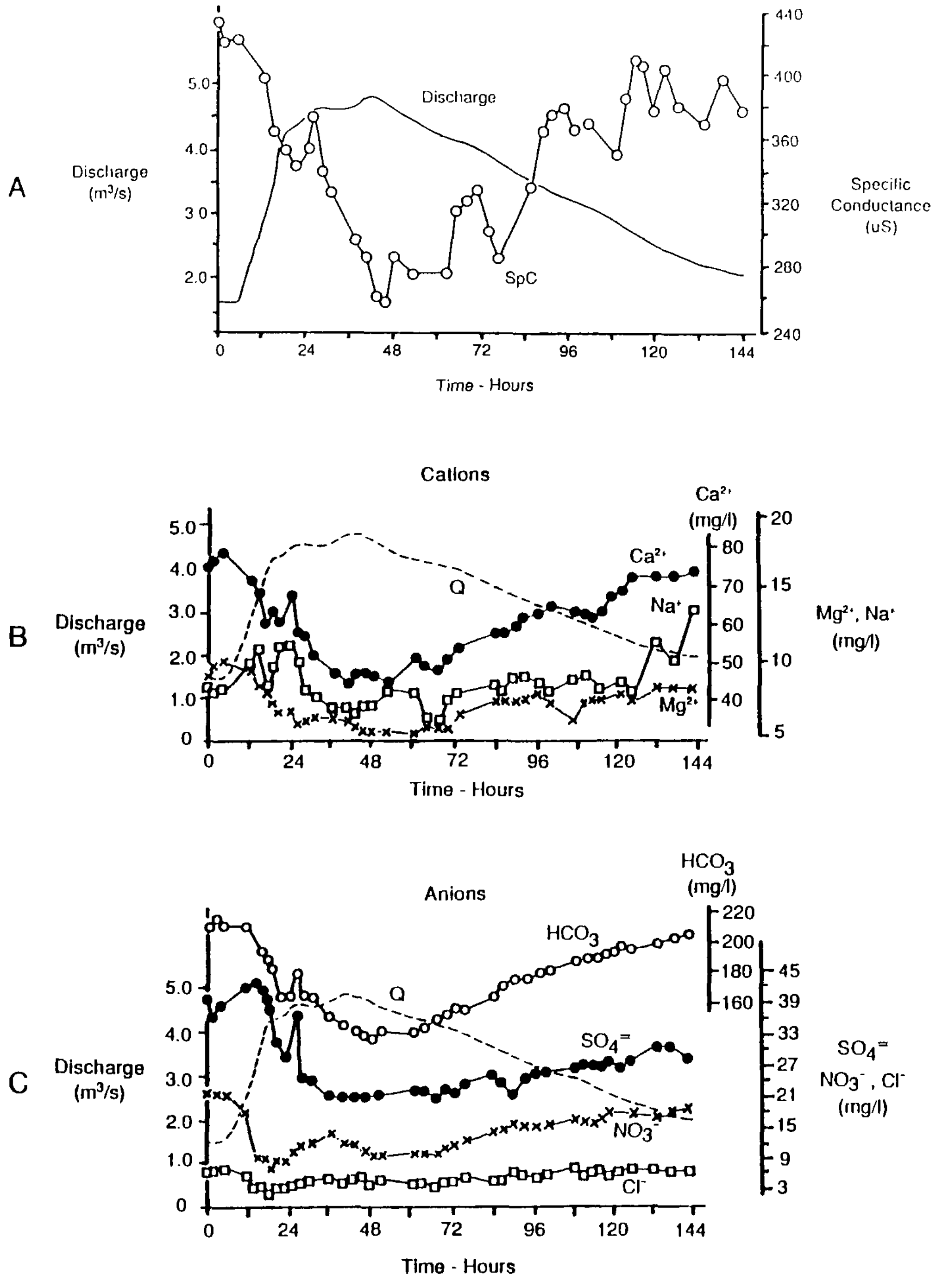

Figure 8. Chemical data from Orangeville Rise, April 12-18, 1991: A) specific conductance, B) cation concentrations, and C) anion concentrations.

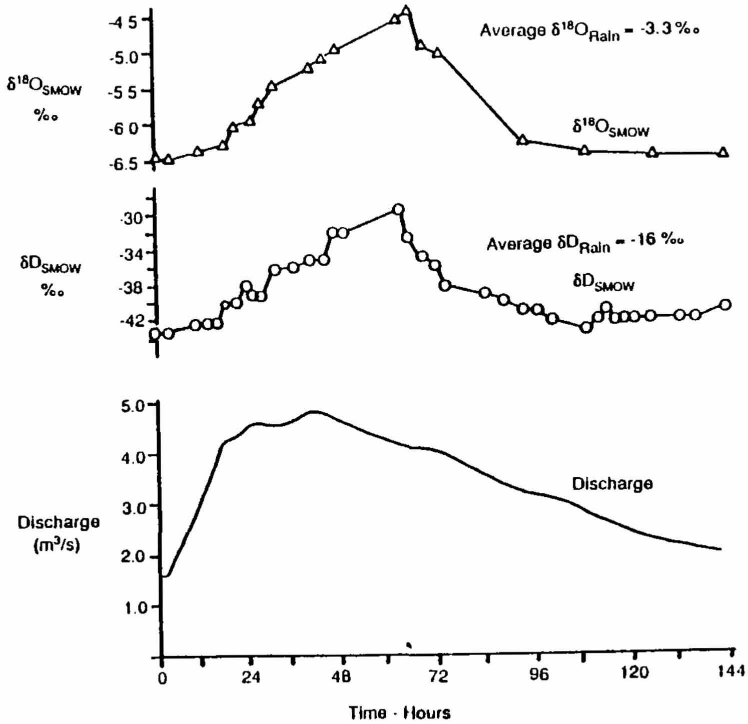

Figure 9. Isotopic composition of discharge at Orangeville Rise, April 12-18, 1991.

Isotopic composition of discharge just prior to the onset of rainfall was -44‰ and -6.5‰ for δD and $\delta^{18}O$ respectively (Figure 9). During the first 12 hours, when discharge increased dramatically, δD increased by only 2‰ and $\delta^{18}O$ increased by only 0.2‰. During the next 24 hours when discharge increased only slightly, the isotopic shift was more rapid. The heaviest discharge occurred from 62 to 65 hours, 24 hours after peak discharge, when δD reached -30‰ and $\delta^{18}O$ climbed to -4.5‰. Isotopic composition of discharge became lighter during the remainder of recession but did not obtain values seen in pre-storm discharge.

Hydrograph Separation

As a means of investigating components of discharge at Orangeville Rise, storm hydrographs were separated into pre-storm and rainwater components. Separations were performed using both deuterium and oxygen isotopic data as a means of checking for consistency in the data.

Hydrograph separation of the October storm is shown in Figure 10 along with the equation used to solve for instantaneous rainwater discharge. At peak discharge, rainwater comprised 12 to 15% of the total discharge. The greatest percentage of rainwater at any time on the hydrograph occurred 18 hours after peak discharge, where it made up 39 to 46% of the total. Separation revealed that a secondary pulse of pre-storm water

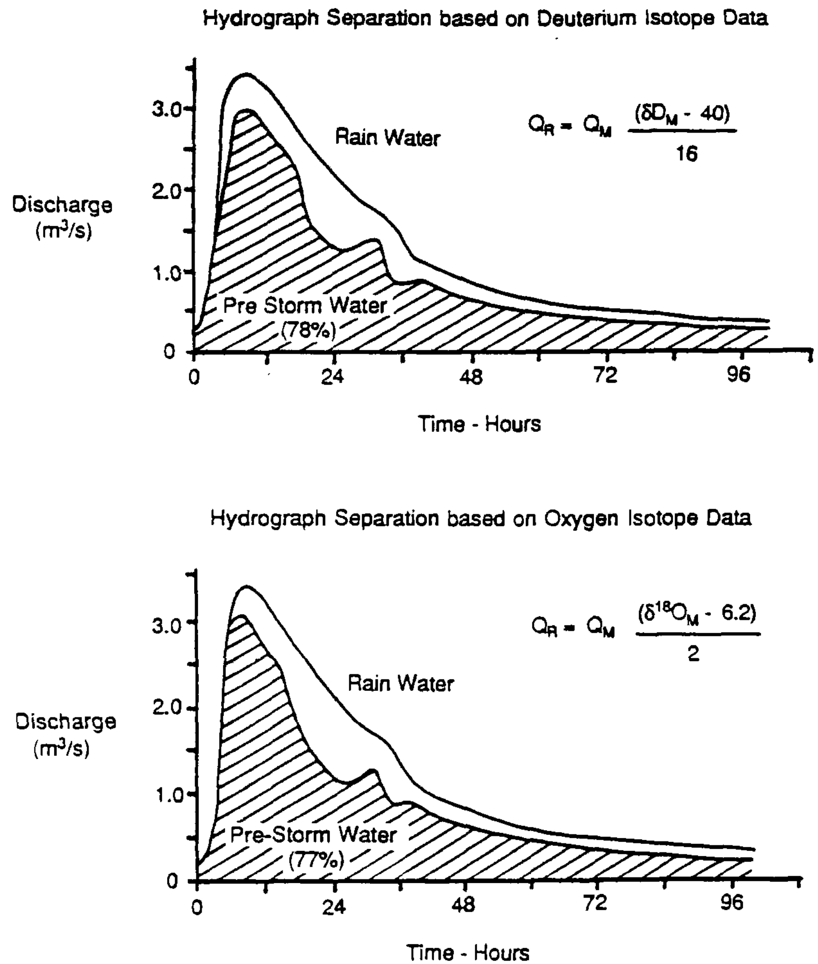

Figure 10. Hydrograph separation of discharge from Orangeville Rise into pre-storm and rainwater components, October 4-8, 1990.

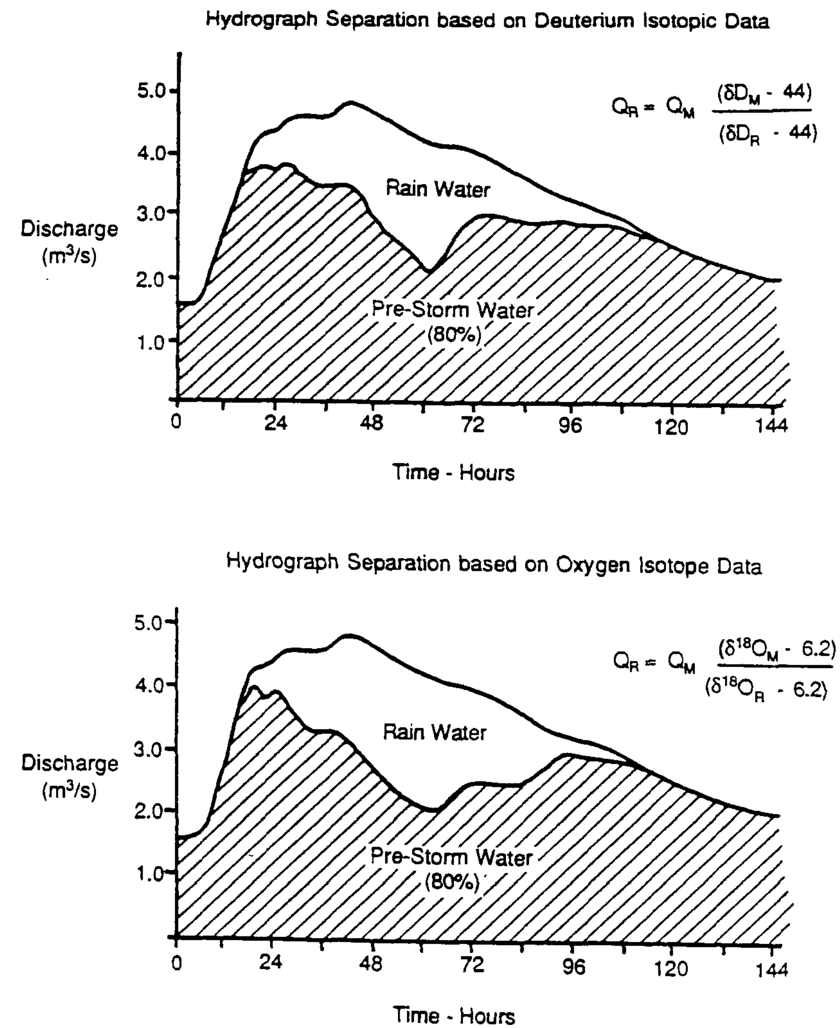

Figure 11. Hydrograph separation of discharge from Orangeville Rise into pre-storm and rainwater components, April 12-18, 1991.

arrived just after the time of maximum rainwater contribution. For the entire five days of monitoring, rainwater made up 20 to 25% of the total discharge at Orangeville Rise.

Separation of the April, 1991 storm hydrograph into pre-storm and rainwater components is shown in Figure 11. The volume-averaged isotopic signatures of rainwater shown in Table 2 were used to calculate the rainwater component of discharge at any one time (McDonnell, et al., 1990).

Hydrograph separation shows that at peak discharge, the rainwater component made up 27 to 33% of the total discharge. Twenty-four hours after peak discharge, the rainwater contribution to total discharge was at a maximum and equaled 47 to 52%. The proportion of rainwater decreased from this point to a very small to non-identifiable component, 110 hours into monitoring. As in the October storm, rainwater made up 20% of the total discharge during the six days of monitoring. This separation also showed pulses of pre-storm water, but the pulses were more subdued than that observed in the October hydrograph.

DISCUSSION and CONCLUSIONS

While discharge increased rapidly in both storms, ionic concentrations decreased only moderately or even increased as in the April storm, indicating that rainwater could not have contributed to a majority of discharge at peak flow. Isotopic data also indicates that there was not a large rainwater pulse arriving at peak discharge. Hydrograph separations reveal that rainwater made up only a small proportion of discharge at peak flow, and the largest proportion of rain water contribution to instantaneous discharge occurred 18 to 24 hours after peak flow.

If the observed isotopic composition of discharge at Orangeville Rise were a product of simple mixing between two end members, namely rainwater, and pre-storm water identified by the isotopic composition of discharge at baseflow, then all data should fall on a line connecting the two extremes. Figure 12 shows the global meteoric water line (Craig, 1961) and the isotopic composition of rain samples collected in the Orangeville Rise basin. The average ground water and baseflow also fall on the meteoric water line, and hence if two component mixing were responsible for shifts in the isotopic composition of discharge, then all data would lie along the line. Data from the October and April storms do not fall on the meteoric water line, but rather define another unique line. These results identify an element of complexity not accounted for in the simple two component mixing model. Since the isotopic composition of all phreatic waters in the basin lies on the meteoric water line, one may be compelled to look to the vadose zone as a source of water which could cause discharge to deviate from the meteoric water line.

The rapidly increasing discharge indicates that there must be substantial concentrated recharge through sinkholes, and rapid transmission of water through soils via macropore flow. Large volumes of water must flow quickly through conduits to discharge at Orangeville Rise, however, this water cannot be rainwater. Rather "old" water must be displaced by new water. The storage

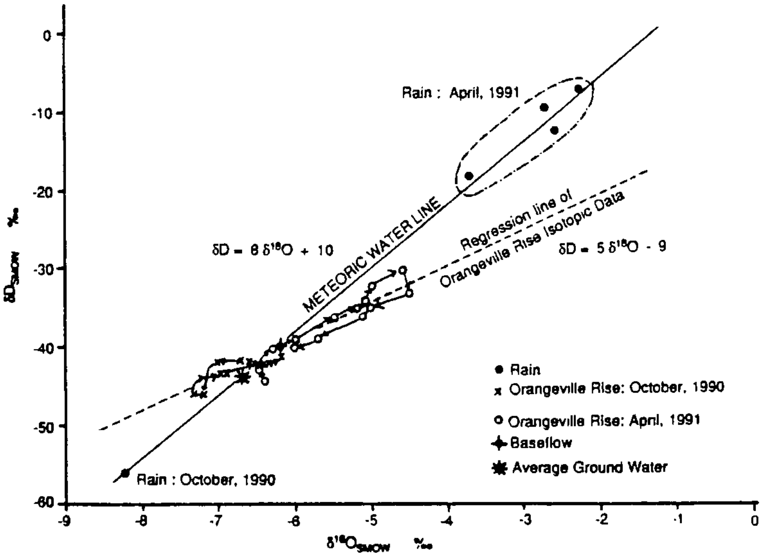

Figure 12. Isotopic data from Orangeville Rise and rainwater collected within the basin, relative to the meteoric water line of Craig, 1961.

compartments which could respond most quickly would be those which contain water stored in conduits, or contain water that is in direct contact with conduits, including epikarst storage, vadose conduit storage and phreatic conduit storage. An influx of rainwater through concentrated recharge could be quickly fed to the epikarst, increasing hydraulic heads there, and causing a rapid discharge of water stored within the epikarst to Orangeville Rise.

The increased hydraulic heads associated with recharge will also result in an increased flow of water through more diffuse compartments of storage. However, transmissivity of the diffuse stores are orders of magnitude lower than that associated with conduit flow. It is difficult, therefore, to see how diffuse water could arrive at the Rise ahead of the water transmitted via conduit flow. As a result, we believe that epikarst and conduit water contributed to the bulk of discharge at peak flow, and diffuse stores of water may be responsible for the pulses of pre-storm water seen during discharge recession.

The rain of October 4 broke drought conditions in the basin. Because it had been so dry, vadose-zone water storage was at a minimum, which can be seen in the rapid rate of discharge recession. The April monitoring occurred at a time of high recharge to the ground water systems. Water storage compartments contained large volumes of water, as can be seen in the slow discharge recession. Ion concentrations remained constant, or increased with initiation of storm flow, and may represent the flushing of waters from epikarst and/or conduit storage.

It is interesting to note that in both storms, the time of minimum specific conductance did not correspond to the time of maximum rain water discharge as identified by the isotopic data. Discharge at any moment of time is a product of all the contributing sources. It is possible that at the time of maximum rainwater arrival, water was also arriving from sources with higher than average total dissolved solids, the resultant discharge having a greater conductivity than might be anticipated for the time of peak rainwater discharge.

Aknowledements

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Gruver - Stable Isotope Separation

1. I want to congratulate the speaker for her particularly thoughtful discussion, and her stimulating suggestions. The stable isotope records presented are suggestive of piston-flow (at least in part). Piston-flow response is difficult to reconcile with the suggestion that the source of the expelled (pre-storm) water deviating from the meteoric water line in the epikarst. Please comment.

2. Can you suggest a method for spiking clean injection water with heavy oxygen or deuterium in order to perform a tracer test with stable isotopes? This could be useful in areas where, for political reasons, it was not practical to use dyes in a karst aquifer. Do you know of such studies? Would light water be more easily obtained? What is the cost per liter (or kg) for heavy water and light water? Having bought same, how much of it is ordinary water? (This is asked in order to get an idea of how much tracer might be needed for an "average" trace and what the cost would be.)

3. The delta ^{18}O - delta D relations of the spring response suggests a third hydrological component, presumably in the vadose zone. However, the end member occupied by the two storms are different, although both lie on the "evaporite line." Why are they different, and why are they displaced to the left and right of the "meteoric water line" for the two storms?

Answer to Question#1:

Water collected during baseflow at Orangeville Rise, and ground waters sampled from domestic water wells lie on the meteoric water line (MWL). As all phreatic and rain water samples lie on or near the MWL, one may need to look to the vadose zone for a source of water that could cause discharge to shift from the MWL.

The first few samples collected during each storm event lie very close to the MWL. I believe these samples represent displaced phreatic conduit water. Six to twelve hours into storm flow, the isotopic signature of discharge begins to deviate from the MWL, as well as in the direction of the rainwater isotopic signature. It seems reasonable to interpret these waters as rain and epikarst water that has been rapidly transmitted via shaft, and conduit flow. Very little work has been done on chemical processes within waters of the epikarst, or what fractionation processes might occur there. This study points to the need for research which addresses these questions.

Answer to Question #2:

I am unfamiliar with any investigation which has used "light" or "heavy" water in place of traditional dyes in karst aquifer tracing. If one feels compelled to use this technique, however, they would be advised to use a water tracer that has the greatest isotopic difference from that of the natural system. The amount of "spiked" water used would be dependent upon the volume of water in the natural system. I could not begin to guess the price of "spiked" water. I can say that a large expense in a stable isotope study is the cost of analysis. Count on at least $35 per sample for deuterium analyses, and $45 for oxygen analyses.

Answer to Question #3:

The delta 18O-delta D relationships suggest at least a third component contributing to total discharge. I would suggest that, while the data seem to lie along a common line with a slope of 5, upon closer examination, the October data lie along a line with slope 4, while the April storm lies along a line with slope 6. I believe that the cause of the shifts for each storm is different. Fractionation processes acting in the dry fall-time, compared with those of the spring may be different and may be responsible for shifts in the isotopic signature of discharge in opposite directions from the meteoric water line.

The Transmission of Light Hydrocarbon Contaminants In Limestone (Karst) Aquifers

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Elizabeth K. Estes M.S., University of Minnesota
Peter J. Idstein M.S., Eastern Kentucky University
Katherine M. Johnson, U. S. Army (USATHAMA)

ABSTRACT

Hydrocarbon contaminants with densities less than water may enter karst aquifers through soil percolation, directly by way of surface runoff into open sinks, or by a combination of these two routes. Once in the aquifer, the movement of these hydrocarbons is governed by four factors: (1) the level of the principal dissolution conduits relative to the zone of saturation, (2) the flow regime (turbulent or non-turbulent) in the conduits which contain the contaminant, (3) the vertical complexity of the framework of dissolution porosity in the aquifer, and (4) the nature of the recharge to the aquifer.

Field investigations indicate that light hydrocarbons can move in these aquifers as gross concentrations of free product at speeds of the order of kilometers per hour. Under other circumstances, the contaminants may be trapped in the aquifer so completely that spring water originating from the point of gross hydrocarbon release shows none of the contaminant.

Trapping of the hydrocarbons occurs so long as two conditions exist. First, the active conduits must remain, to a large extent, in a completely water-filled (phreatic) condition. If a free surface (an air-water interface) develops throughout a large percentage of the active conduits the floating fluid may move as "free product". Second, the groundwater velocity in the vicinity of the floating product must remain slow. Gross turbulence entrains the "free product" in the swift flow of the active conduits.

INTRODUCTION

Contaminant movement in karst aquifers often appears strange and erratic to hydrogeologists more familiar with granular aquifers. Light hydrocarbon contaminants, those with densities lower than water, appear especially capricious. The behavior of these materials in karst aquifers is related to the dissolution porosity, often referred to as conduit porosity, which they possess.

Dissolution porosity in paleozoic carbonate rock normally occurs along partings such as bedding planes, joints, and faults. In areas of flat lying or gently dipping carbonates, conduits formed along bedding planes are commonly most important in forming the master conduits which permit long distance transmission of groundwater. Joint and fault related conduits provide routes for admitting meteoric water to the aquifer. The total length of explorable conduits primarily guided by bedding planes in such areas commonly exceeds the total length of those primarily guided by joints by 10 to 1 (Ewers, 1972). On a worldwide basis, which includes karst aquifers in dipping rocks, the ratio of bedding plane to joint and fault controlled conduits is of the order of 1.3 to 1 (Palmer, 1991) This is the case, not only because bedding planes commonly extend over greater horizontal distances than joints, but also because they can transmit water in any direction, unlike joints and faults.

Unlike the diffuse, dispersive, laminar flow of granular aquifers, groundwater movement in karst aquifers is concentrated, convergent toward the conduits, and at times turbulent (Quinlan and Ewers, 1985). Surface water and contaminants may have direct access to the conduits by way of sinkholes with swallet openings and with sinking streams. The presence of conduits may permit rapid movement of liquid phase hydrocarbons. Because of the turbulent flow regime in these conduits, contaminant movement may be in the form of globules of entrained "free product" as well as a dissolved phase. Groundwater and contaminant velocities in these conduits are commonly as high as several kilometers per day (Quinlan and Ewers, 1983).

FACTORS AFFECTING LIGHT HYDROCARBON MOVEMENT IN KARST

The movement of immiscible floating hydrocarbons in karst aquifers is controlled by factors relating to the characteristics of the framework of dissolution porosity and to the type of flow regime found in these conduits. These factors can be conveniently grouped into the following four categories.

1.- The level of the principal conduits relative to the zone of saturation

Conduits in karst aquifers occur deep within the phreatic zone (the zone of saturation beneath the potentiometric surface) and well above normal levels of groundwater circulation. Their range in a specific area is related to the thickness of soluble carbonate rock, the local geomorphic history, and climatic factors. Light hydrocarbon contaminant transmission is less likely in master conduits located deep in the phreatic zone than in those which are along the upper part of this zone.

Phreatic Conduits

Where the master conduits responsible for horizontal transmission of groundwater lie permanently within the phreatic zone, floating hydrocarbons should not move easily. The entry point for the contaminants should form at least a temporary trap for this free product at the top of the phreatic zone (Fig. 1, point "A"). Likewise, any other portion of the master conduit

Hydrocarbon Traps

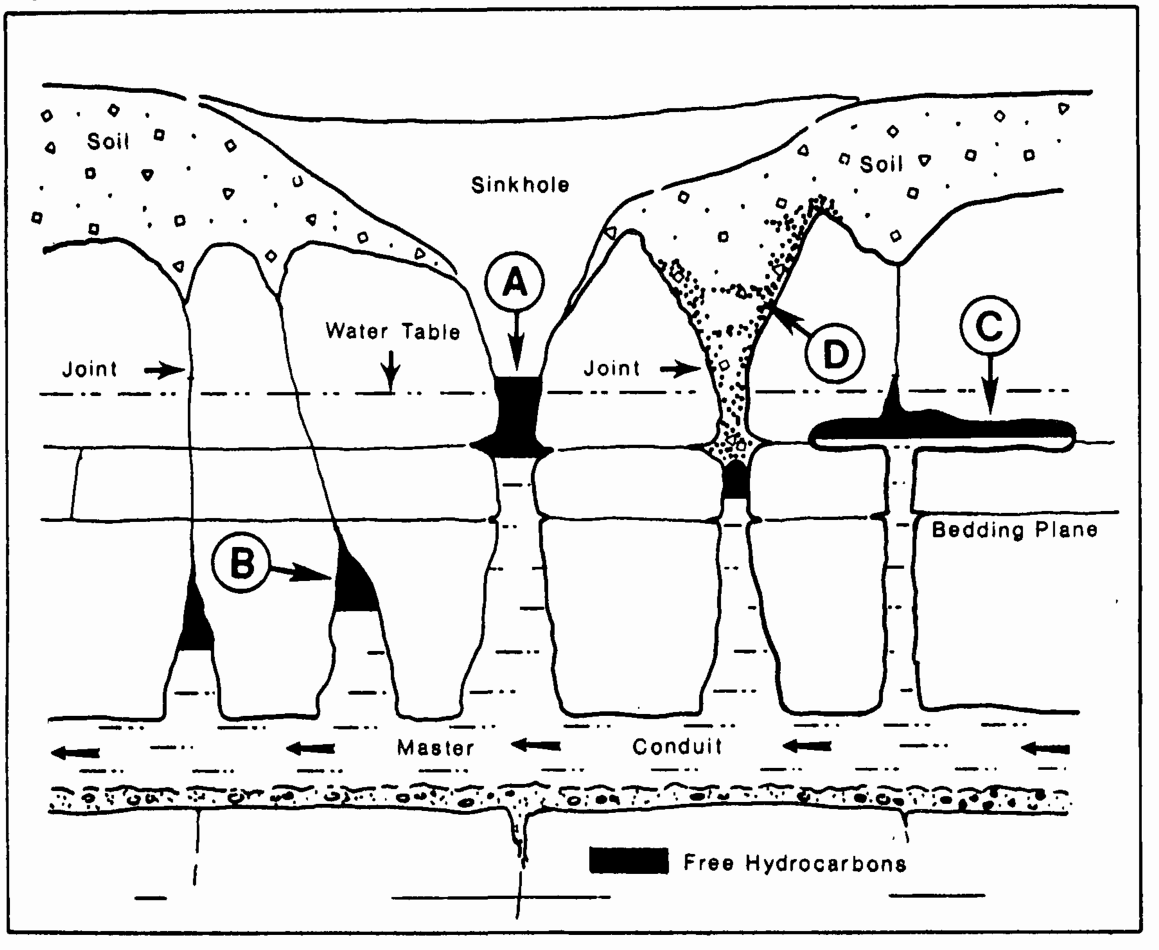

FIGURE 1 Light hydrocarbons can be trapped above phreatic master conduits in ("A") aquifer entry points, ("B") solution widened joints, ("C") horizontal conduits above bedding planes, and ("D") the soil filling of grikes (solution widened joints) and in the soil around nearby clints (bedrock pinnacles).

which extends above the potentiometric surface should form an additional trap for any free product which becomes entrained in the flow of the conduit. Only that portion of the hydrocarbon which dissolves in the groundwater should move easily in phreatic conduits.

Epiphreatic Conduits

Where the master conduits are in the epiphreatic zone (the zone in the immediate vicinity of the potentiometric surface), floating hydrocarbons tend to move more freely. Under base-flow conditions, some master conduits will have an air-water interface along their entire length. This provides a continuously sloped surface for the movement of floating "free product", identical to a surface stream. The hydrocarbons will move at a speed nearly the same as the water in this circumstance. More commonly, epiphreatic conduits have air-water interface conditions spaced intermittently along their length. To the degree that this interface diminishes, floating hydrocarbons will move with increased difficulty, their mobility becoming more and more dependant upon the nature of the flow regime in the conduit.

2.- The flow regime in the conduits which contain the contaminant

Turbulent Flow

During periods of turbulent flow, floating hydrocarbons would be readily entrained in conduit water. Groundwater velocities in typical karst aquifers range from 30 ft/hr at base flow to 1300 ft/hr in periods of high flow (Quinlan and Ewers, 1985). The hydrocarbons, normally excluded from phreatic master conduits, could be mobilized by the turbulence and would travel at nearly the velocity of the water.

Laminar Flow

When laminar flow prevails, floating free hydrocarbons are unlikely to be entrained in the groundwater. The "free product" will separate from the water and collect at the highest points along the conduit ceiling. Diffusion of the hydrocarbons into the water will occur, but the concentrations of contaminants carried in this way may be very low. The large quantities of groundwater moving in the conduits may limit these concentrations as well as the relatively high octanol-water partition coefficients for many of these substances. Both trapping of the floating free product and diffusion of the hydrocarbons into the water are dependant upon the vertical complexity of the conduits.

3.- The vertical complexity of the framework of dissolution porosity in the aquifer

Vertical conduits in carbonate aquifers have two roles in the transmission of light hydrocarbons. They provide entry routes for these contaminants and they also provide traps for the free product. These conduits together with abandoned horizontal conduits are collectively referred to as the epikarst (Mangin, 1974-75). This term is used here in a broad sense to include any solution porosity lying above the level of the active drainage trunks, including that which may be above or below the normal potentiometric surface. Karst aquifers with an extensive system of interconnected vertical and horizontal phreatic conduits above deep phreatic master conduits have an increased ability to trap light hydrocarbons and a reduced likelihood of transmitting these contaminants. Those with simple shallow conduit systems are more prone to contaminant movement.

Hydrocarbon Entry

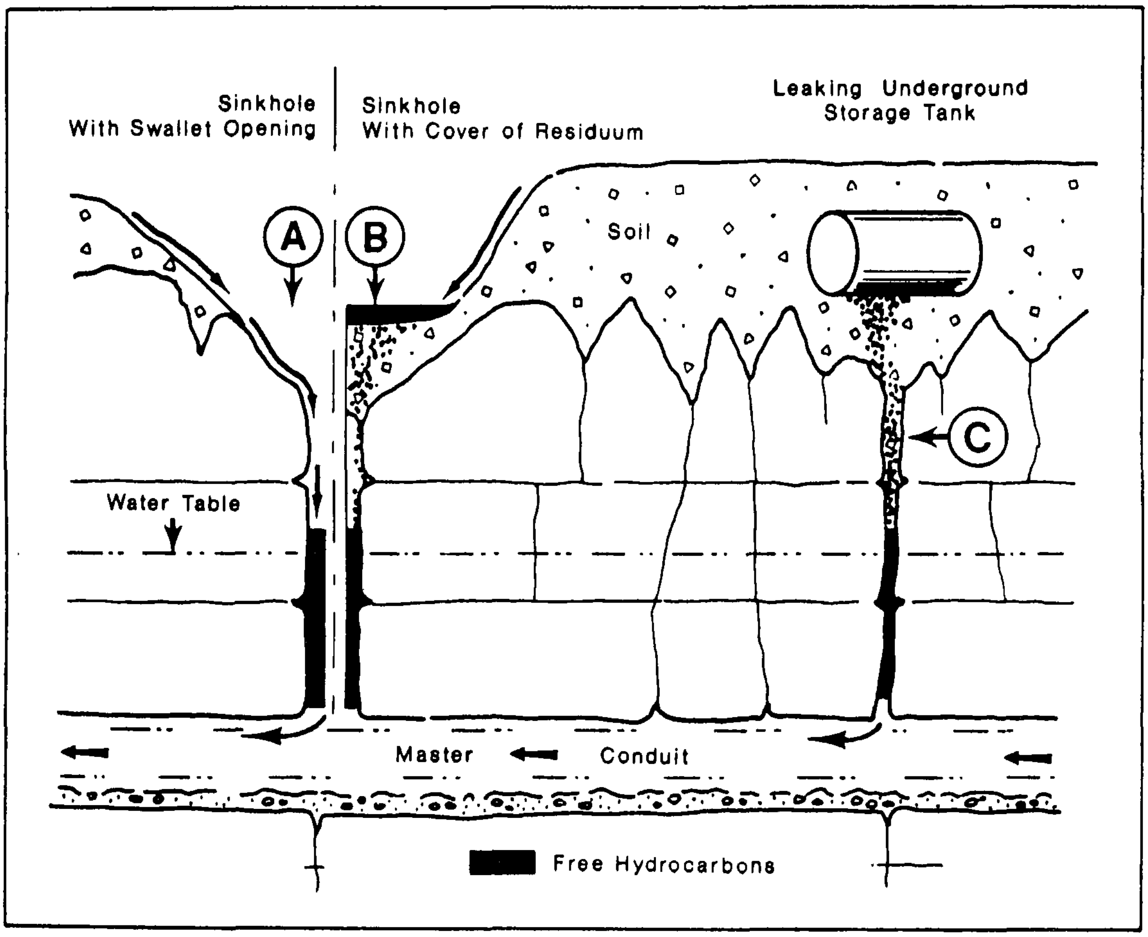

FIGURE 2 Light hydrocarbons can enter karst aquifers in several ways. They may enter directly with concentrated surface recharge entering swallets (point "A"), by seepage through sinkhole bottoms covered by residuum (point "B"), or through the soil cover and minor solution widened joints in the carbonate rock (point "C").

Hydrocarbon Entry

Hydrocarbons may gain entry to a karst aquifer through sinkholes with open swallets which give direct access to the conduit system or by seepage through the residuum in a sinkhole bottom (Fig. 2, points "A" and "B"). In either case, a vertical conduit, typically developed along a joint, provides both the means for the sinkhole to form and the entry route for the contaminants.

A much larger number of vertical epikarst conduits are associated with features called grikes. These are linear depressions in the bedrock surface with narrow bottoms and wide tops which develop along joints. They can conduct hydrocarbon contaminants from the bottom of the soil horizon to master conduits in the phreatic or epiphreatic zone. These conduits are often common in areas where no depressions are visible at the surface of the overlying soil.

Hydrocarbon Trapping

Master conduits are frequently intersected by joints. Where these joints are enlarged by dissolution, they form traps in their phreatic portions for floating hydrocarbons (Fig. 1, point "B"). Vertical conduits of this type intersecting master conduits which are accessible to direct observation frequently have volumes in excess of 1000 gallons. Where the carbonate rock is heavily jointed, these potential trapping situations are seen to occur at intervals of a few meters or less. Thus, they may provide a considerable storage capacity for floating hydrocarbons.

Vertical conduits may connect the master conduit with higher horizontal conduits in the phreatic portion of the epikarst (Fig. 1, point "C"). These conduits are often laterally discontinuous due to truncation by soil filled grikes. Conduits of this type may provide additional storage for light hydrocarbons.

Vertical conduits may extend from the master conduit to the upper bedrock surface and the base of the soil. In such cases the hydrocarbons may collect at the soil bedrock interface if the potentiometric surface reaches above this level either permanently or intermittently (Fig. 1, point "D").

Where the interval between the level of the master conduit and the level of the uppermost phreatic part of the vertical conduit elements is great, the groundwater in the master conduit may be in poor communication with the floating contaminants. Both entrainment of the hydrocarbons by master conduit turbulence and their diffusion into this flow will be very limited.

4.- The nature of the recharge to the aquifer

Two basic types of recharge occur in carbonate aquifers, diffuse recharge and concentrated recharge. The nature of this recharge has a significant effect upon the movement of light hydrocarbon contaminants. Diffuse recharge which passes through a soil cover and enters the aquifer through a large number of joints is less likely to produce gross turbulence in the aquifer conduits than concentrated recharge. This latter includes the concentrated runoff which enters swallets in sinkholes and both allogenic and autogenic sinking streams. Concentrated storm runoff from these sources regularly produce stage changes in conduit systems of 30 feet, and changes of 75 feet are not unusual. Flows of these magnitudes produce gross turbulence in the conduit systems which propel coarse gravel from spring orifices. A modest amount of turbulent recharge entering the top of a free product trap may entrain the collected contaminant. Similarly, hydrocarbons trapped close to the master conduit may be mobilized by master conduit turbulence induced by the storm flow of a sinking stream.

Karst aquifers with thick soil covers, few open sinks, and without sinking streams have a reduced likelihood of experiencing movement of trapped light hydrocarbon contaminants. Those with the opposite characteristics are at higher risk for contaminant movement.

CASE STUDY 1, AN AQUIFER WITHOUT CONTAMINANT MOVEMENT
Quarles Spring Groundwater Basin

Quarles Spring, located in Christian County, Kentucky, is used as a potable and irrigation water supply. Dye tracing by Ewers and his students (Ewers Water Consultants, 1989; Carey, 1990; Greene, 1990) has shown that this spring is the principal discharge point for a groundwater basin which includes Campbell Army Airfield (Fig. 3). Jet fuel spillage at the airfield has appeared in several wells in the karst aquifer at the airfield. As much as 16 feet of "free product" has been measured in monitoring well MCI-2 (Point "C", Fig. 3; Fig. 4) by Ewers (Ewers Water Consultants, 1991). Dames & Moore report free product in four other wells (Dames & Moore, 1991). Investigations by Duda (Dames & Moore, 1991) suggest that the fuel entered the aquifer through a sinkhole which takes drainage from the airfield (Point "B", Fig. 3). Repeated sampling of Quarles Spring by Duda (Dames & Moore, 1991) has failed to show jet fuel, dissolved or free. Thus, it appears that floating hydrocarbons are not mobile in this aquifer. The known and inferred aquifer characteristics fit the criteria for low mobility outlined above.

The principal conduit system-
Quarles Spring possesses a deep rise-pool, suggesting a phreatic condition in the master conduit which is tributary to it. Information from Well MCI-2 (Fig. 4) at the airfield gives further evidence that the conduit system is well within the phreatic zone. This well intersects a solution conduit at about 459 feet msl. This conduit has a minimum vertical extent of 16 feet (MCI, 1987), and is a possible source of the jet fuel contaminant which is seen in the annulus of this well. The well bore entry point into the conduit is approximately 29 feet lower than the normal level of Quarles Spring, well within the phreatic zone (Table 1; Fig. 4).

Quarles Spring Groundwater Basin

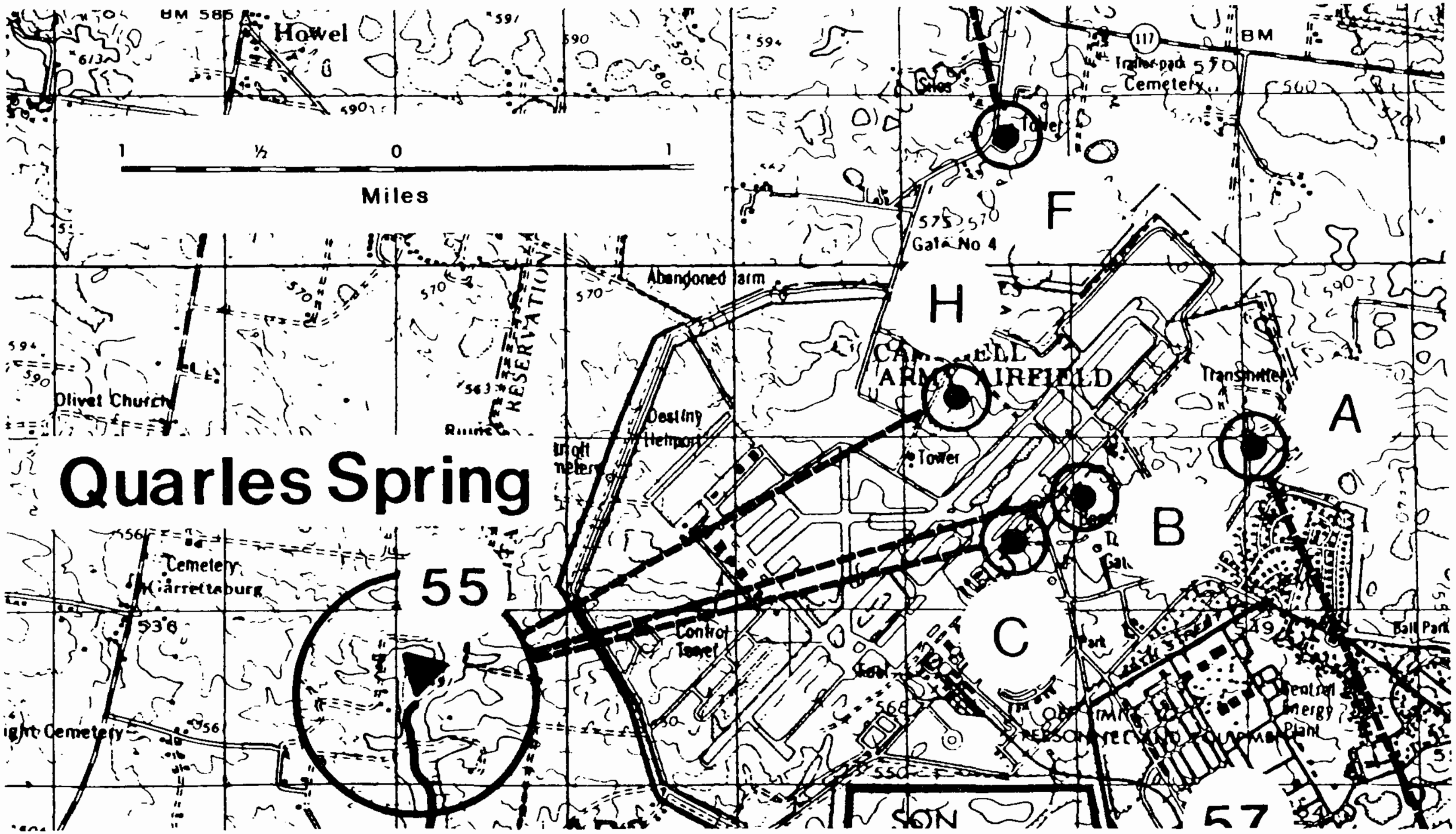

FIGURE 3 Quarles Spring is the principal discharge point for a groundwater basin which includes Campbell Army Airfield. Jet fuel spillage at the airfield has appeared in several wells in the karst aquifer. Point "C" is well MCI-2. Points "B", "C", and "H" are the location of dye inputs. The tracer dyes were recovered at Quarles Spring, number 55. No jet fuel has been recovered from this spring.

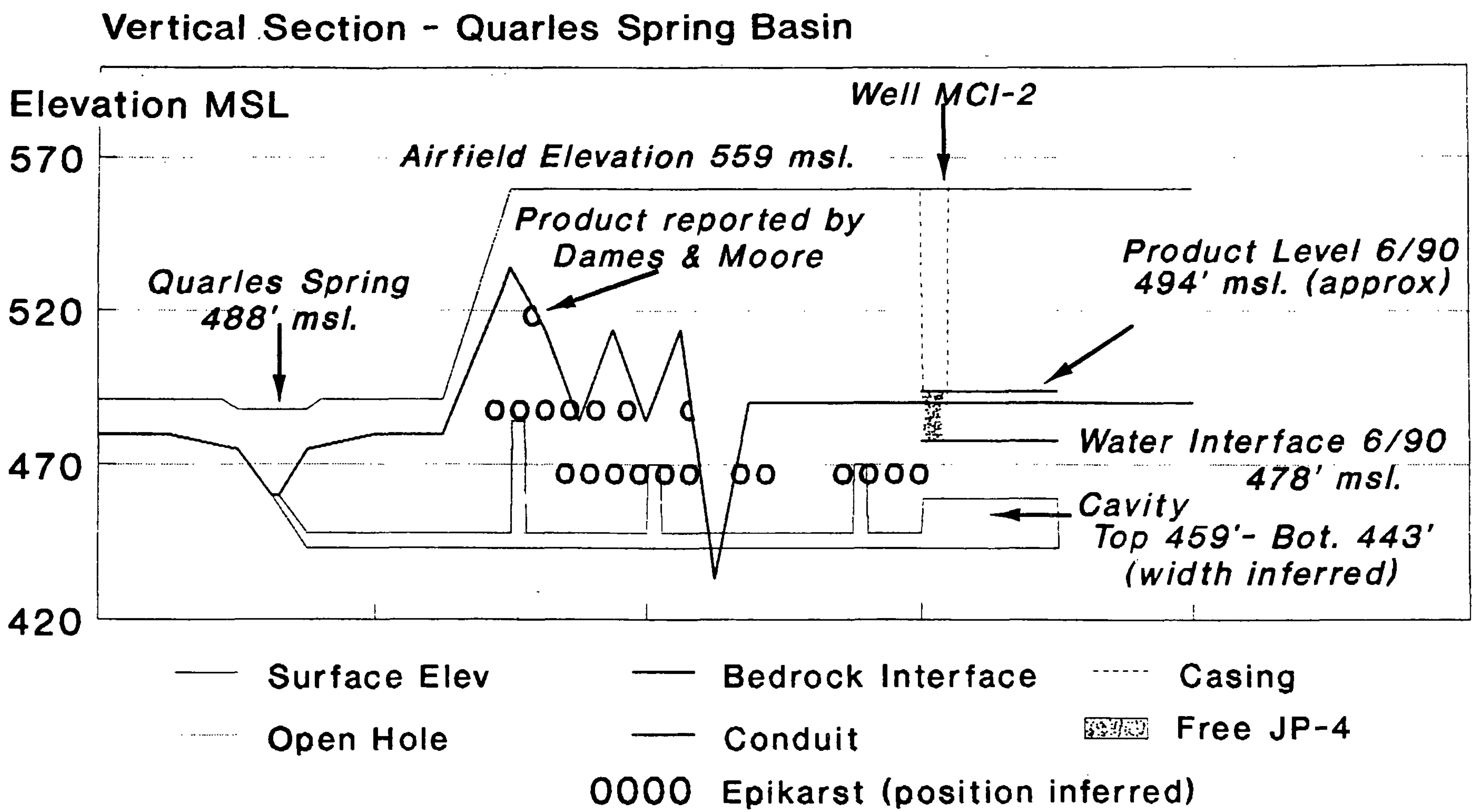

FIGURE 4 This schematic vertical section between well MCI-2 (point "C", Fig. 3) and Quarles Spring is based upon information from 37 wells and numerous soil investigations. The conduits shown are inferred, except for that in the vicinity of well MCI-2. The typical and extreme range of the soil bedrock interface is shown.

Table 1 **ELEVATION OF FEATURES RELATED TO MONITORING WELL MCI-2, THE BEDROCK, AND QUARLES SPRING**

Well Features	Elevation (ft.)	Bedrock & Spring Features
Top of Casing---------	560.16	
Ground Level----------	559.65	
	534 -----	Bedrock/Soil Interface-Shallowest (MCI, 1987)
Product Level 6/90----	494	
	491 ----	Floodplain at Quarles Spr.
Water Level Without-Floating JP-4---	490.25	
Bottom of Casing------	489.74	
	488 ----	Water level in Quarles Spr.
Water Interface 6/90--	478	
Top of Cavity---------	459	
Bottom of Cavity------	443	
	433 -----	Bedrock/Soil Interface-Deepest (MCI, 1987)

The conduit intersected by this well was shown by dye tracing to be connected to Quarles Spring (EWC, 1989). It is reported to have significant amounts of fine sediment moving through it. The driller recorded that a five foot layer of mud was swept into this conduit during a severe rainfall event which occurred during the installation of the well (MCI, 1987). We may assume that this conduit is a part of the active conduit system, tributary to Quarles spring. If this conduit were significantly lower than the active conduit system and if it were not a part of that system, then it would probably be sediment filled. The groundwater gradient to Quarles Spring is very low. The head loss from the well to the spring is approximately 2 feet at base flow (Table 1) over a distance of 11,800 feet, a typical value for a phreatic conduit. These observations suggest that the conduit system beneath Campbell Army Airfield and tributary to Quarles Spring is phreatic in nature, even under extreme low-flow circumstances.

The flow regime and the aquifer recharge-

Flow in the master conduits of the Quarles Spring groundwater basin are almost certainly turbulent much of the time. The spring discharges water even when heavy withdrawal for irrigation occurs during very dry periods. Base flow discharge is estimated at 0.5 ft^3/sec. However, it discharges primarily fine sand-sized sediment and is typically quite clear. It does not have the flashy response and wide range of flow velocities exhibited by many karst aquifers. Flow velocities in the region of most of its concentrated recharge points is probably not turbulent. The reason for the lack of turbulence can be found in the nature of the aquifer recharge.

Recharge to the Quarles Spring groundwater basin does not include sinking streams. All of its recharge is autogenic, it occurs directly over the groundwater basin where only soluble bedrock with chert bands and cherty residuum overlie the bedrock aquifer. Sinkholes are common, but they rarely contain direct openings to the bedrock. A layer of cherty residuum, 25 to 127 feet in thickness, mantles the bedrock and sinkholes restricting most of the recharge to soil percolation (Dames & Moore, 1991).

Low-flow and high-flow time of travel studies utilizing dye tracers were undertaken between the drywell at point "H" and Quarles Spring (Fig. 3)(EWC, 1991). The velocities obtained were 95 feet per hour and 226 feet per hour respectively. These values are quite different from those obtained from the Mammoth cave region where the aquifer is recharged directly through swallets and by sinking streams. Values of 9 meters per hour and 390 meters per hour are quoted by Quinlan & Ewers (1985) for that area.

The vertical complexity of the conduits-
If the conduit encountered in the well at point "C" (Fig. 3) is indicative of the level of the master conduit, 31 feet of space in the vertical dimension are available for the trapping of contaminants (Table 1; Fig, 4). Fracture trace analysis (Dames and Moore, 1991) and the extremely variable elevation of the bedrock soil interface (Dames & Moore, 1991; MCI, 1987) strongly suggest that fractures enlarged by dissolution are common in the bedrock aquifer. Therefore, vertical and horizontal conduits should be common in the epikarstic zone. These should be quite effective at trapping floating contaminants in the Campbell Army Airfield setting. The wide range of the soil bedrock interface, reported by drillers, should insure that there are several horizons of horizontal epikarstic solution porosity which are poorly integrated laterally (Fig. 4, Table 1).

The 16 foot layer of floating jet fuel in the well reported by Ewers is associated with the conduit porosity encountered when this well was drilled. The free product reported by Duda was discovered at the soil bedrock interface, and is probably fuel which has migrated upward through solution widened joints from the master conduit below. These horizons are likely to be poorly connected to sources of concentrated recharge because of the relative lack of open sinkholes. Without this turbulent recharge, accumulated product is unlikely to be flushed from these reservoirs. Because these reservoirs extend well above the active flow system there would be little opportunity for measurable quantities of the floating immiscible contaminants to dissolve in the groundwater circulating in the main conduits.

CASE STUDY 2, AN AQUIFER WITH GROSS CONTAMINANT MOVEMENT
Little Sinking Creek - Big Sinking Creek

This site is located near the Big Sinking Creek oil field in Lee County, Kentucky. Both Little Sinking Creek and Big Sinking Creek are, in part, subsurface streams. Portions of their surface valleys are perennially streamless (Fig. 5). Conduits of explorable dimensions with multiple entrances pass beneath the dry portions of these valleys.

Little Sinking Creek - Big Sinking Creek, KY

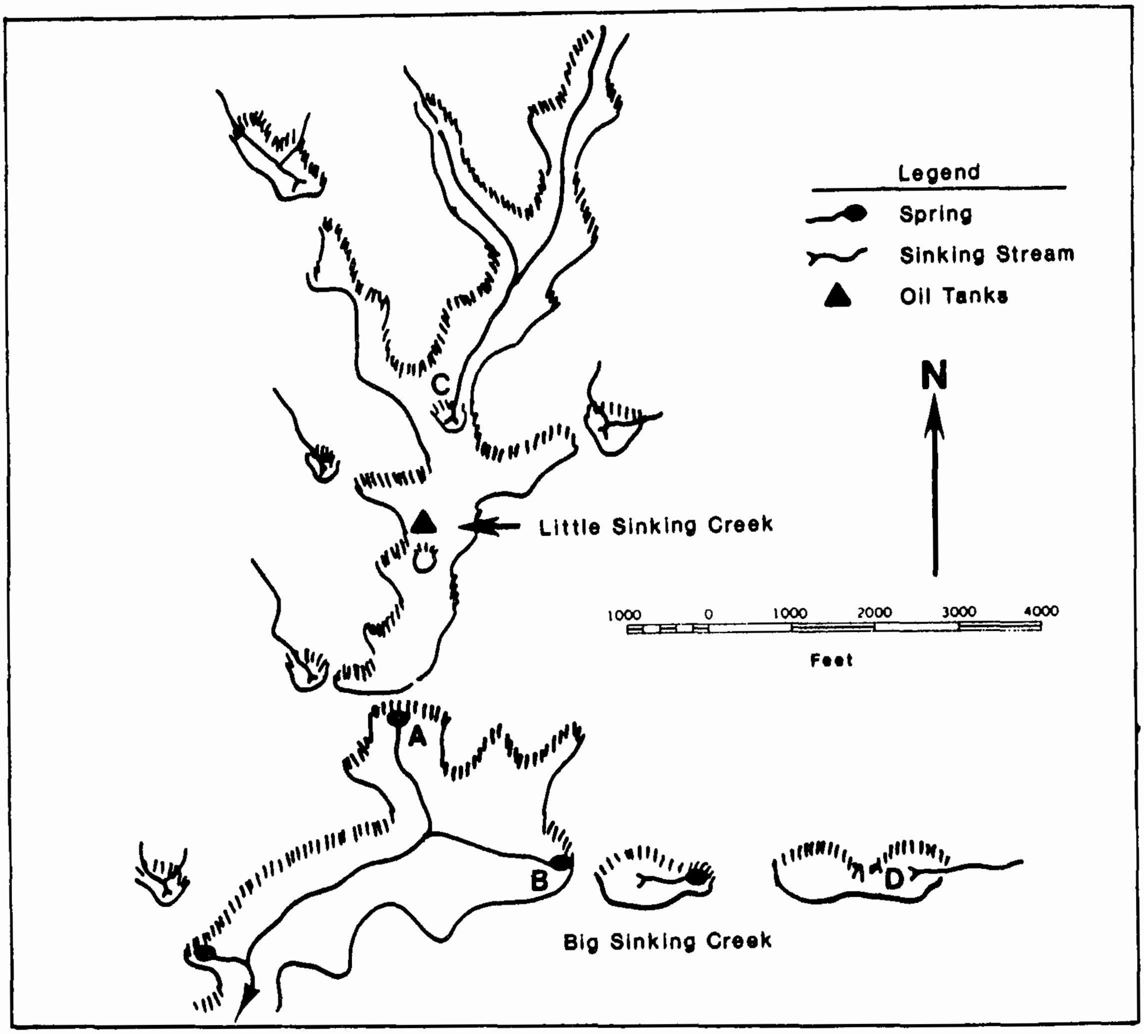

FIGURE 5 Little Sinking Creek and Big Sinking Creek are, in part, subsurface streams. Several smaller sinking streams are tributary to the master conduits beneath these valleys. Oil spills at several points have been transmitted quickly to the springs at points "A" and "B".

This karst aquifer is developed in the Mississippian age Newman Limestone. Oil production is from units 700 feet beneath the karst aquifer. These horizons are isolated from the aquifer by several hundred feet of shale. Pipeline breaks, storage tank cleaning, and brine discharges have introduced chlorides and crude oil into these sinking streams and into the conduit entrances. The interior surfaces of the conduits are coated with petroleum and tar balls are common along the surface and subsurface portions of the streams. A spring, perched upon a shale layer beneath the Newman Limestone discharges from the lower end of each of these valleys near their confluence (Fig. 5). Floating free product is discharged with the groundwater from these spring points after spills have occurred. Thus, it appears that floating hydrocarbons are extremely mobile in this aquifer. The known and inferred aquifer characteristics fit the criteria for high mobility outlined above.

The principal conduit systems-
The master conduits in both valleys are located in the epiphreatic zone. Deeper phreatic circulation is prevented in this case by the presence of insoluble shale beds in the Renfro Member of the Borden Formation. Virtually the entire length of conduit beneath the valley of Little Sinking Creek shown in figure 6 is explorable and occupied by a stream with an air-water interface above. The conduit which forms the spring orifice has been observed to maintain an air-water interface under very high flow conditions. The master conduit beneath the valley of Big Sinking Creek does not possess an air-water interface over its entire length, but the crude oil is transmitted along the total conduit system (Fig. 7).

The flow regime and the aquifer recharge-
The aquifer recharge at this site is almost entirely from allogenic sources, by way of sinking streams. This concentrated recharge is clearly turbulent, and during heavy rainfall events it is grossly turbulent. The conduit flow in the aquifer is visible at several points along each valley and turbulent flow, similar to that in the sinking streams and the spring resurgences, can be directly observed.

The portion of the Big Sinking Creek conduit above mist cave, shown in Figure 7, may not possess an air-water interface even under base flow conditions. Water ponds over the swallet of Big Sinking Creek during high-flow conditions and the crude oil is temporarily trapped at the surface. When the flow subsides, the ponded water lowers and turbulence entrains the floating contaminant, and it is conducted through several spring and sink points to its final resurgence.

The vertical complexity of the conduits-
Only a few feet of limestone overlie the master conduits at this site. Therefore, the opportunities for trapping of hydrocarbons along the few sections of phreatic flow are severely limited.

Vertical Section - Little Sinking Creek

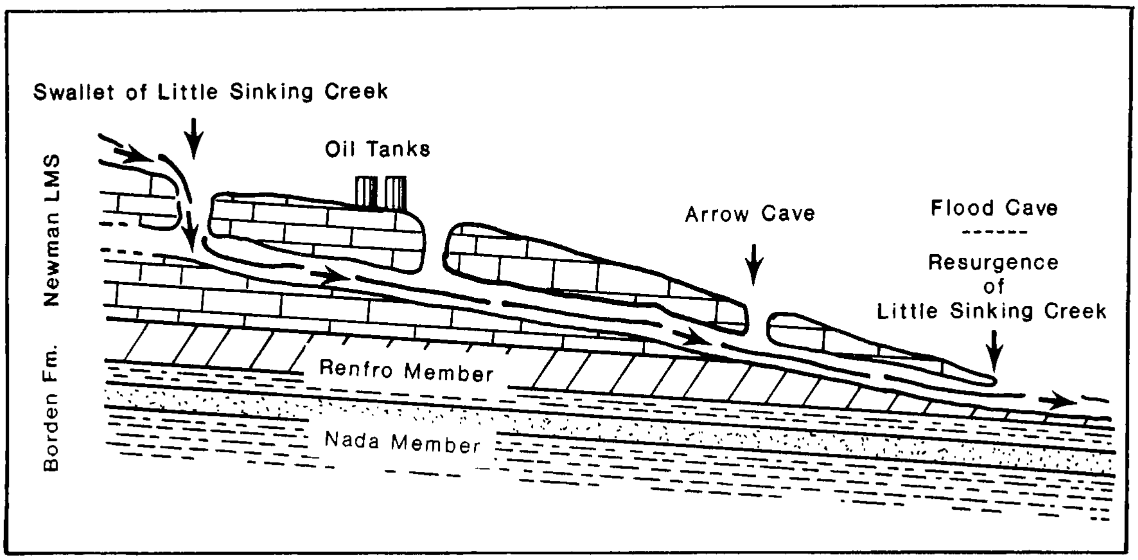

FIGURE 6 Vertical section through Little Sinking Creek
The Swallet of Little Sinking Creek and Flood Cave are points "A" and "C" in Figure 5, respectively.

Vertical Section - Big Sinking Creek

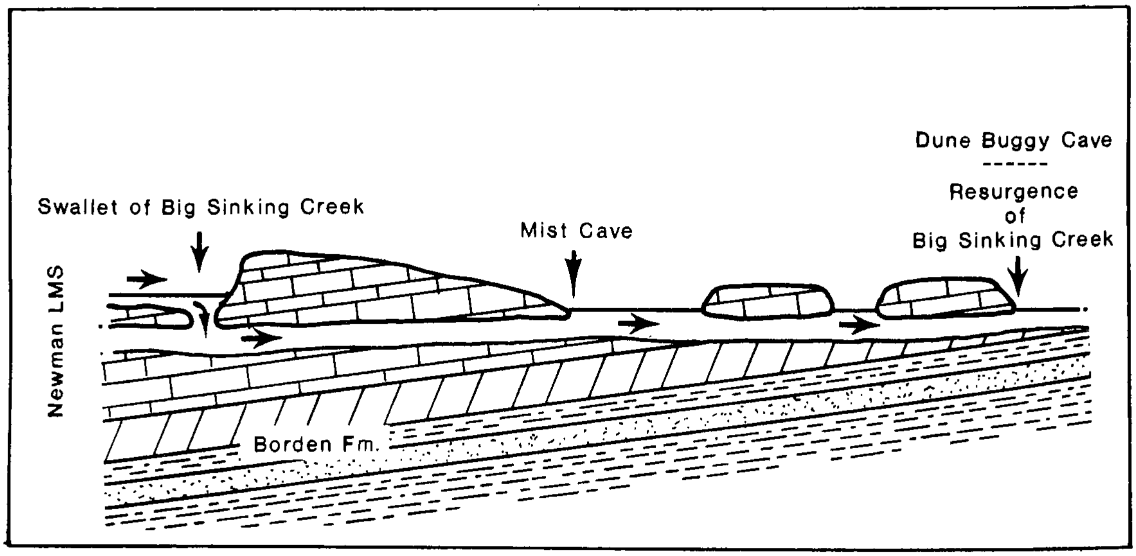

FIGURE 7 Vertical section through Big Sinking Creek
The Swallet of Big Sinking Creek and Dune Buggy Cave are points "B" and "D" in Figure 5, respectively.

CASE STUDY 3, AN AQUIFER WITH MIXED CONTAMINANT MOVEMENT
Lancaster Road Gasoline Station - Richmond Kentucky

An underground storage tank at this site released gasoline into a karst aquifer consisting of Ordovician Limestones. Free product was discharged into Tennis Court Spring, an intermittent seep 150 ft from the leak site (Fig. 8). Fluorescent tracer dye injected into the storage tank pit showed that groundwater flowed to Tennis Court Spring and also to Little Caesar Spring a perennial spring 900 ft distant. Gas chromatograph analysis of samples from these springs showed that the free product discharging from Tennis Court Spring was nearly identical to the tank contents. No gasoline constituents could be discerned in the samples from Little Caesar Spring either in high-flow or low-flow conditions.

There are no data from wells or other sources which can give direct insight into the nature of the conduits which connect these two springs to the leak site. However, the two springs clearly indicate the presence in the aquifer of a natural light non-aqueous phase liquid separation system. This system operates in an environment where water flow clearly occurs in discrete conduits. The tracer introduced into the tank pit leaves no doubt about this assertion. It appeared at both springs overnight. As expected, the level of the spring discharging product was significantly higher than the one which discharged water only (Table 2). This karst aquifer has no sinking streams and few sinkholes, none with swallet openings which could admit water directly to the aquifer.

Table 2 ELEVATION OF FEATURES RELATED TO THE LANCASTER ROAD GASOLINE STATION

Elevation (ft.)	Spring Features
995 ------	Top of gasoline storage tank pit (dye injection point)
985 ------	Tennis Court Spring - Groundwater, gasoline, and dye were discharged at this point.
935 ------	Little Caesar Spring - Only groundwater was discharged at this point.

CONCLUSIONS

The empirical studies suggest that the following conditions decrease the probability of movement of light hydrocarbons in karst aquifers:

1-The limestone formations are thick and conduits are well within the phreatic zone.

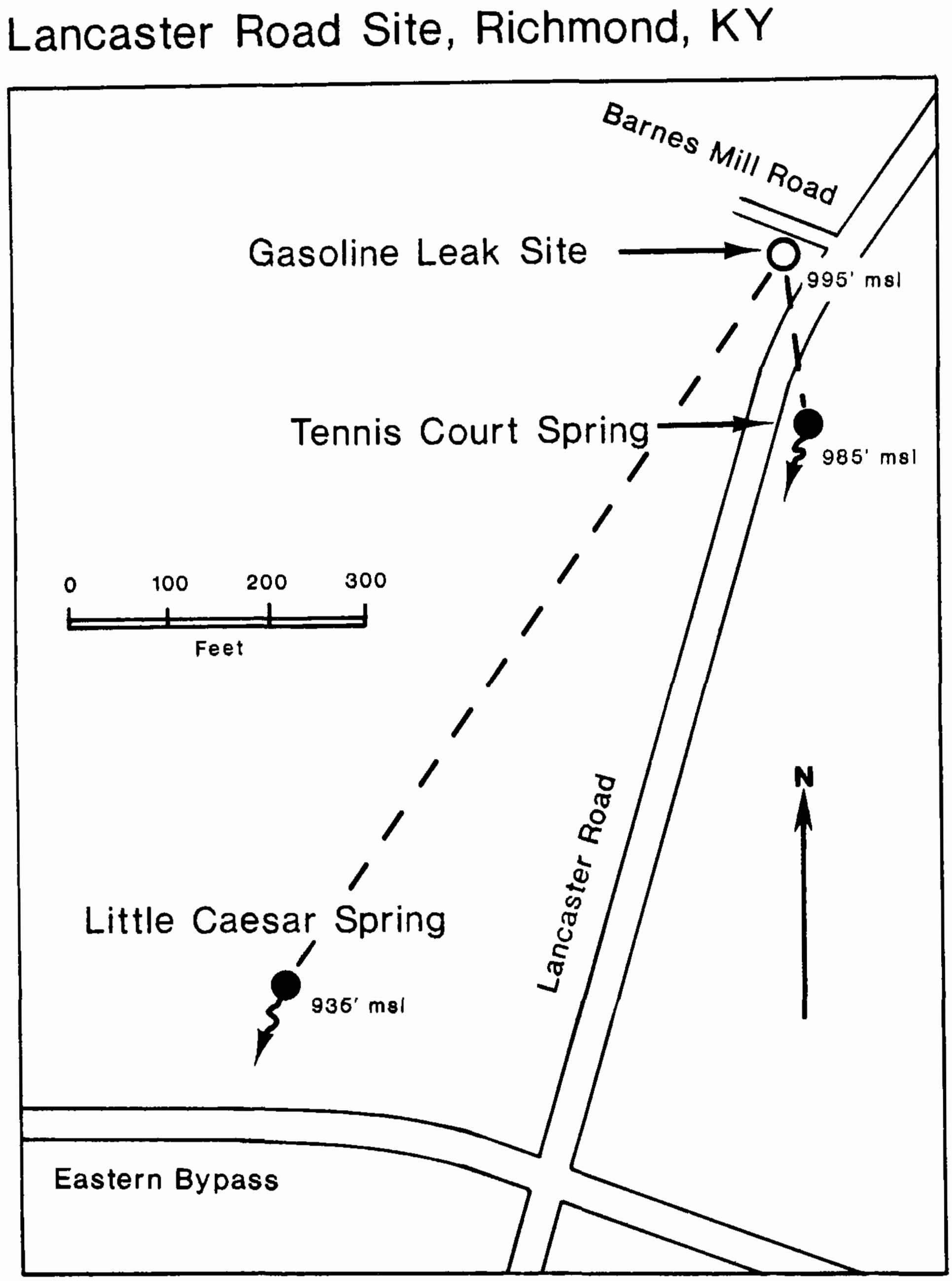

FIGURE 8 An underground storage tank at this site released gasoline into a karst aquifer. Free product was discharged at Tennis Court Spring, an intermittent seep. Fluorescent tracer dye injected into the storage tank pit showed that groundwater flowed to Tennis Court Spring and also to Little Caesar Spring, although no gasoline could be detected at this latter spring. The numbers indicate the elevation of the leak site and the springs.

2-The limestones are covered by a thick mantle of soil causing recharge to the aquifer to be slow and non-turbulent.

3-The aquifer is not recharged by sinking streams so that turbulence in the master conduits is minimized.

4-An extensive system of epikarstic porosity overlies the master conduit providing cavities for trapping of the hydrocarbons.

Given all of these characteristics, hydrocarbon mobilization could occur if drought conditions can lower the potentiometric surface sufficiently to alter the phreatic condition in the master conduit.

Human activities or natural events which would permit storm water to flush the epikarstic zone could possibly mobilize contaminants stored there. These would include intense highly localized rainfall events, developing sinkholes, excavations near bedrock, and dry-wells. Attempts to retrieve contaminants through recovery wells by creating depression cones extending near or below the level of the spring outlets could create conditions for their transport, particularly if this occurred at a time when gross turbulence existed in the trunk conduits.

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BIOGRAPHICAL SKETCHES

Ralph O. Ewers is professor of geology and director of the Groundwater Research Laboratory at Eastern Kentucky University, and a principal in Ewers Water Consultants, a consulting firm specializing in carbonate aquifers. His B.S. and M.S. degrees in geology were earned at the University of Cincinnati and his Ph.D. was earned at McMaster University (1982). Professor Ewers' special interests include the applications of tracer and electronic monitoring techniques to the solution of practical environmental problems in karst groundwaters. He was co-recipient of the 1986 E.B. Burwell Award from the Geological Society of America for a "work of distinction in engineering geology."

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Anthony J. Duda is senior hydrogeologist with Dames & Moore, where he has worked since 1980. He received his B.A. degree in geology from Rutgers University. Mr Duda has been involved in planning and implementing groundwater remediation strategies, soil gas sampling programs, and geophysical surveys. His experience includes investigations at more than 30 governmental and private sites in more than 10 states. He has been lead hydrogeologist at the Ft. Campbell, Campbell Army Airfield site where he was involved in a major UST investigation.

Elizabeth K. Estes is a doctoral candidate at the Department of Geology and Geophysics at the University of Minnesota. She received her M.S. degree from Eastern Kentucky University. Among her many accomplishments is a five year study of the area in the vicinity of Big Sinking Creek in Lee County, Kentucky, and several studies with the Cave Research Foundation at Mammoth Cave Kentucky. Her research interests include, karst geomorphology, speleogenesis, and groundwater geochemistry.

Peter Idstein received his Bachelor of Science degree in Geology at Eastern Illinois University. He is completing his Master of Science degree in Geology at Eastern Kentucky University. Mr. Idstein has spent one year working at the Florida Sinkhole Research Institute conducting studies on conduit dominated groundwater flow. He has also spent a year working for Ewers Water Consultants conducting dye tracing studies and continuous electronic monitoring studies in many karst dominated and non-karst terranes.

Catherine A. Johnson is a project officer for the United States Army Toxic and Hazardous Materials Agency (USATHAMA). She graduated with General Honors from the University of Maryland with a B.S. degree in Chemical Engineering. Her duties at USATHAMA involve managing Preliminary Assessments/Site Inspections, Remedial Investigations/Feasibility Studies, and RCRA Facility Assessments/Investigation studies at Army sites. Ms. Johnson has been the project officer at the Fort Campbell, Campbell Army Airfield site for the last two years. During that time she has managed extensive field work including; geophysical investigations, sediment, Water, and soil sampling, and monitoring well installation.

The Transmission of Light Hydrocarbon Contaminants In Limestone (Karst) Aquifers

Ralph O. Ewers, Anthony J Duda, Elizabeth K. Estes, Peter J Idstein and Katherine M. Johnson.

1. What are the major difference between the physical and chemical properties of the dyes and the light hydrocarbons used in you study? Why not try to use a tracer that has properties similar to the hydrocarbons? Has there been a chemical transformation of the hydrocarbons in the limestone which affects their properties?

ANSWER- The dyes are water soluble substances that are non-toxic. The light hydrocarbons are toxic immiscible organic substances. I know of no non-toxic immiscible hydrocarbons that regulatory agencies would allow me to place in an aquifer in the quantities that would be required. In any case, tracers with the properties of the light hydrocarbons would not give us the answers that we need in the time required. They would be poor tracers.

I suspect that there are many chemical and biochemical transformations which have occurred. However, I do not believe that such changes are the principal reason for their apparent retention in many karst situations. I believe that immiscibility and density are the primary reasons. The demonstrated travel times are too fast for these processes to be effective.

2. What were the dissolved BTEX levels associated with the Little Caesar Spring? Was dilution sufficient to limit benzene level to below 5 micrograms per liter. What were dissolved levels at low flow conditions?

ANSWER- I do not know the BTEX levels because they were below the level of detection at the time.

Demonstrably, the separation system in the aquifer and the dilution were sufficient. Hundreds of gallons of gasoline were probably involved, and the spring flow was relatively small, a 2-3 cubic feet per second at maximum. Separation of the floaters seems to be very important.

VELOCITIES OF PIEZOMETRIC WAVES CAUSED BY PUMPING IN KARSTIC AQUIFERS

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Abstract

The velocities induced by pumping in a karstic area were studied at a test site in 20 boreholes at 5m intervals. The site was 4.6 km from a spring. Pumping of 1 to 2 $m^3 s^{-1}$. was performed at the spring. The time for the effect of pumping at the spring to reach the boreholes varied from 6 to 8 minutes, corresponding to an apparent velocity of 9.6 to 12.8 m s^{-1}. Wave propagation probably occurs in drainage "conduits" (connected to the spring) at a velocity related to physical parameters such as the water (modulus of elasticity), rock (Young's modulus) and the characteristics of the conduits (equivalent diameter) and the type of flow (under head or open surface). The recorded velocities show that flow was not entirely under head since they would have been much higher in this case because of the rigidity of the limestone. At the other boreholes (which boreholes) impact times varied from 26 minutes to 5 hours, corresponding to an apparent velocity of 2.9 to 0.2 m s^{-1}. These observations are interpreted using a conceptual model with double diffusivity taking into account low diffusivity fissured blocks and high diffusivity cracks. The boreholes directly affected by pumping at the spring penetrated a fracture in the cracks with blocks in the model. The boreholes with low velocity are in the blocks themselves and far from the drain. One of the results of this difference in piezometric behaviour is that certain apparent piezometric gradients are reversed between pumping and periods of stoppage.

Introduction

Measurements of the velocity of piezometric waves caused by pumping in a karstic aquifer can contribute to better surveying of the heterogeneity of the medium. This is particularly fruitful when numerous piezometers are installed in an aquiferous zone and the fissure structure can be surveyed.

General considerations

In porous media, pressure transfers can be interpreted by solving the diffusivity equation (Bear, 1972). There are two possible procedures for karstic aquifers:

a) The karst is identified as a continuous medium equivalent, as has been proposed for certain fissured rocks (Maini & Ockins, 1977). The problems are handled like those of porous media. This is possible if the area considered is very large in relation to a basic grid representative of the high conductivity network. However, even more than in fractured rock, heterogeneity is found at all scales in karst (Drogue, 1988) and it is difficult to consider it as a porous medium.
b) The second procedure takes into account the geometrical and hydraulic characteristics of aquiferous karstic networks. However, schematization is necessary because of the extreme complexity. This is the case of the conceptual models proposed by fissured aquifers. In these models, matrix porosity is combined with fracture porosity (Barenblatt *et al.*, 1960; Wilson *et al.*, 1974; Shapiro & Anderson, 1983; Huyakorn *et al.*, 1983; Moench, 1984).
A double porosity fissure model is used for karstic aquifers (Drogue, 1980, 1988). Little-karstified fissure blocks with low hydraulic conductivity ($K' = 10^{-7}$ to 10^{-9} m s^{-1}) are separated by cracks or karstic channels with high hydraulic conductivity ($K' = 10^{-3}$ to 10^{-1} m s^{-1}) (Fig. 1). The model can be used in many karstic aquifers to interpret the phenomena observed: piezometry, hydrochemistry and temperature patterns (Drogue, 1985, Drogue, forthcoming).
The channels between the blocks form the aquifer drainage network which converges on the channel feeding the spring. If the channel is under head it can be considered as a conduit. The velocity of a pressure wave can then be expressed as follows:

$$C = \frac{1}{\sqrt{\rho\left(\frac{1}{K} + \frac{1}{Ee}\right)}}$$

where: ρ = voluminal mass of water, K = modulus of elasticity of the water, D = diameter of the conduit, E = Young's modulus of the conduit material, e = thickness of the wall

In the extreme case of an indeformable conduit, the ratio D/Ee is zero and C = 1425 m s^{-1} (at 15°C), where $\rho = 10^{-2}$ and $K = 2\ 10^{-8}$ kg f m^{-2}.
This velocity that of sound under water - is certainly not attained in karstic aquifers. Indeed, these rocks undergo deformation by the piezometric variations caused by rain water alone. This was demonstrated by extensometric measurements performed in caverns (Crochet *et al.*, 1983). In addition, the channels are often combined with cracks with open surfaces corresponding to the blocks in the model. These cracks contain damping volumes which slow the pressure waves. Aquiferous non-karstic soil cover may also play the same role through the drainage effect.

Experimental conditions

The experimental site consists of 20 boreholes 60 m deep laid out in a 500 m^2 area (Fig. 2). The boreholes are 5 to 7 m apart. They run through much-fractured, karstified Upper Jurassic and Lower Cretaceous limestone forming a free aquifer. All the boreholes reached the saturated zone. Piezometric levels vary from 20 to 50 m according to the season. Various work at the site - test pumping, tracing, logging, piezometric monitoring over a period of several years and hydrochemical investigations have verified that there is hydraulic continuity between all the boreholes. It has also been possible to site each borehole in relation to the drains and blocks of the conceptual model (Drogue & Grillot, 1976).
The spring at which pumping is carried out is 4.6 km from the experimental site. The natural flow ranges from 0.4 m^3 s^{-1} at low water to 8 m^3 s^{-1} during flood periods; average flow is 2 m^3 s^{-1}. It drains a 400 km^2 aquifer within which the experimental site is located. Numerous faults between the spring and the experimental site place karst and non-karstic formations in contact with each other.
Exploitation of the spring with discontinuous pumping started in 1968 at 0.8 m^3 s^{-1}. Pumping discharge has been up to 1.5 m^3 s^{-1} in recent years.

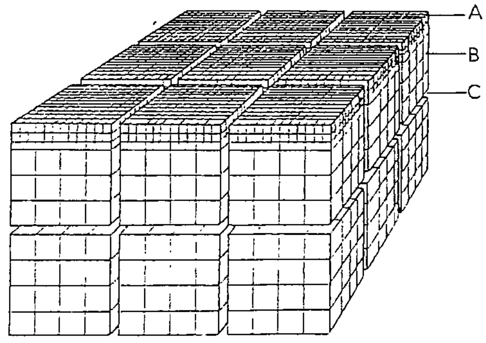

Fig. 1. Conceptual model of a double fissural porosity karst. Blocks with low permeability (B) are bounded by open cracks with high hydraulic conductivity (C). When the karst outcrops, the upper part of the aquifer is extremely fissured (A) because of decompression and surface weathering.

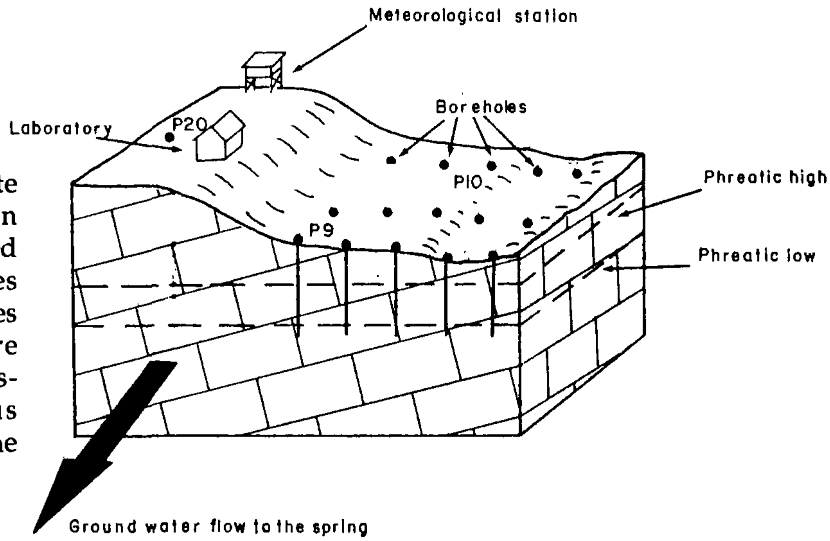

Fig. 2. The experimental site (Terrieu, karst formation in the South of France) equipped with 20 piezometric boreholes 60 m deep. Only the boreholes mentioned in the text are numbered. They are representative of the various piezometers monitored at the site.

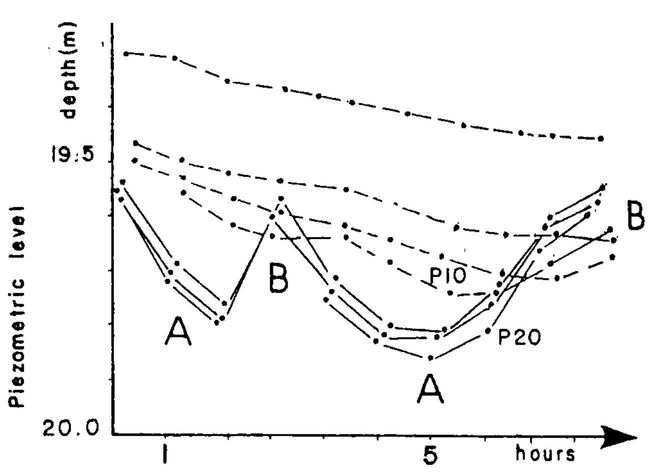

Fig. 3. Piezometric evolution in several boreholes during the first pumping test: A, pumping at the spring; B, pumps stopped.

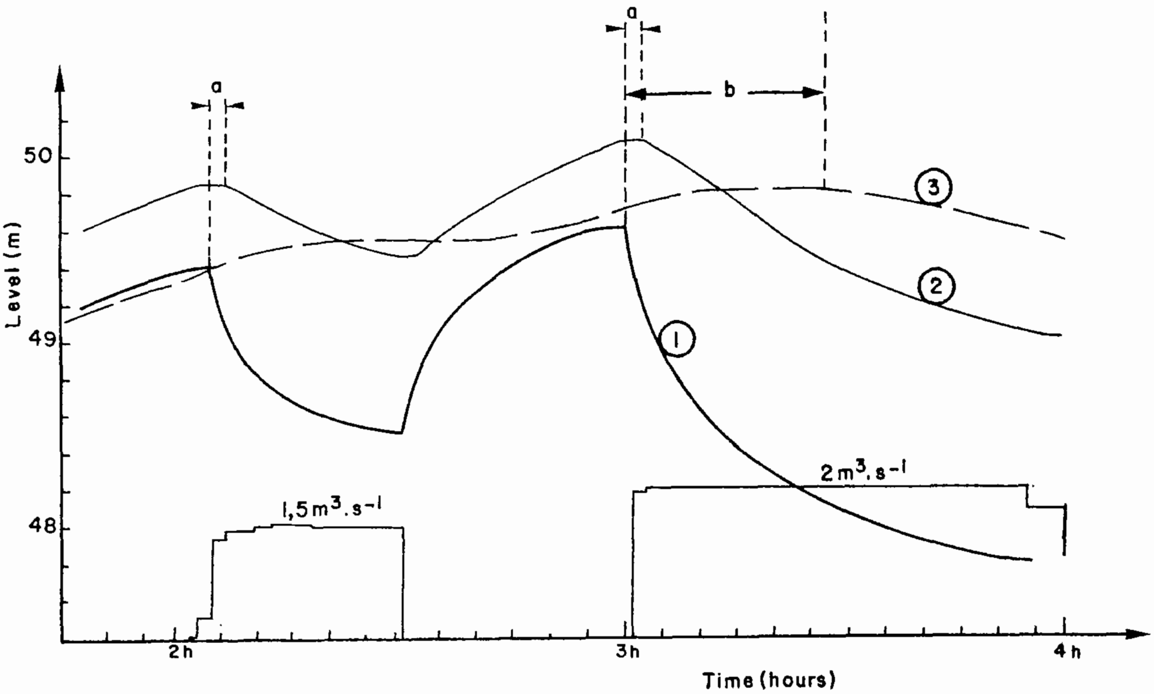

Fig. 4. Piezometric evolution during the second pumping test. 1, piezometry at the spring; 2, piezometry at borehole P20; 3, piezometry at borehole P9; a, 2-minute response time; b, 24-minute response time.

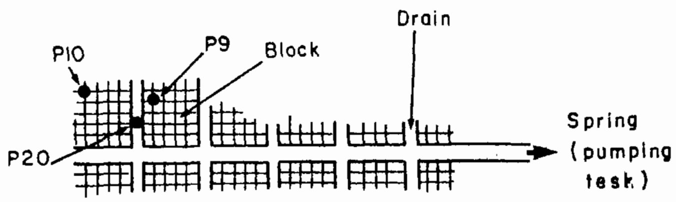

Fig. 5. Diagram (based on piezometer velocity) of the spring and boreholes P20, P9 and P10 in a double porosity fissure model.

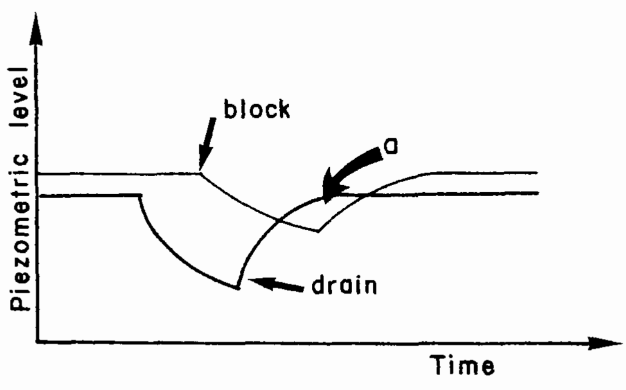

Fig. 6. Piezometric data for a drain and an adjacent block during pumping. Dephasing of the piezometric movement accounts for the inversion of the piezometric gradient (a).

Experimental results

Preliminary experiment

This preliminary experiment was performed with natural flow of 0.5 $m^3\ s^{-1}$ at the spring. Pumping discharge was 0.8 $m^3\ s^{-1}$ with 0.5 m maximum drawdown at the spring.
Piezometric variations at the experimental site displayed the following features (Fig. 3):
- drawdown was considerable at some boreholes: 0.3 to 0.4 m. In contrast, drawdown was slight or non-existent at other boreholes only 5 or 10 m from the former;
- the piezometric levels in the boreholes little-affected by pumping were higher than those of the boreholes in which there was considerable drawdown. When the pumps were stopped, the apparent piezometric gradients between these boreholes were reversed.

In order to confirm and complete these observations, pumping was performed at high discharges to cause greater drawdown at the sight and enable better observation of the phenomena.

High discharge pumping

After pump stoppages limited to 30 min because of water requirements, piezometric variations were triggered by instantaneous pump starting at discharges of 1.5 and then 2 $m^3\ s^{-1}$. Three piezometers representative of the different domains of the double porosity model were observed:
P20: on a drain,
P9: in a block and near a drain,
P10: in a block.
The results are shown in the table below.

	P20	P9	P10
Response time (min)	2	24	approx. 120
Apparent velocity ($m\ s^{-1}$)	38	3.2	approx. 0.6

Drawdown was too slight for accurate measurement at borehole P10.

Discussion

It should first be noted that the hydraulic distance between the spring and experimental site is doubtless greater than the geographical distance of 4.6 km because of twists and turns of the network. Real wave velocities were thus greater than the apparent velocities in the table above.
Borehole P20 responded very rapidly. It cut through a drain which must be connected to the karstic network which feeds the spring. P9, which is on a block but near a drain reacted more slowly. P10, located in a block, reacted much later. These results thus confirm the suitability of the model for the interpretation of the velocities of piezometric waves. The respective positions of the spring and the piezometer boreholes can be schematized in the model (Fig. 5).
Dephasing times are related to reductions in the amplitude of piezometric fluctuations. Thus, stopping pumping between discharges of 1.5 m3 s-1 and 2 m3 s-1 caused piezometric rises of 0.65 m in borehole P20, 0.25 m in P9 and approximately 0.03 m in P10. There were thus apparent piezometric gradients between boreholes. This is explained by the dephasing of the piezometric variations between a drain and an adjacent block during short duration pumping (Fig. 6). This phenomenon might be a characteristic of the karst. It is interesting to observe that these piezometric inversions were similar to those observed in the same boreholes during a flood caused by infiltration (Drogue, 1980). In the latter case, rapid piezometric recharging caused a

temporary rise in level in the drains in relation to the piezometric levels in the blocks. This piezometric inversion phenomenon also occurred between the spring and borehole P10 and was caused by the low apparent piezometric gradient between the site and the spring ($8.7\ 10^{-2}$) when pumping was stopped.

Conclusion

Piezometric wave velocities thus provide extremely accurate information about the aquiferous structure of the karst and its degree of heterogeneity. The representativeness of a piezometer in aquifers of this kind should thus be examined attentively. The hydrodynamic calculations made using data from such a piezometer only apply to a certain domain of the aquifer.

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Solution Mining And Resultant Evaporite Karst Development In Tully Valley, New York

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Abstract

A solution mining operation was conducted in Tully Valley, New York from 1889 to 1988. In excess of 37 million m^3 of halite was removed from 335 to 518 meters below the ground surface. An interbedded sequence of gypsum, shales, limestones, and sandstone overlie the halite beds. This sequence is capped by thick, unconsolidated deposits of till, sand and gravel, and lacustrine clay.

As a result of this mining, large void cavities were created, followed by numerous fractures extending upward to the ground surface. The resulting settlement area is in excess of 550 hectares. Within this area sinkholes formed, gaping fractures developed and streams were pirated into the subsurface. Interformational mixing of groundwater now occurs between formerly separate flow systems, providing substantial recharge to deep formations.

Some 2 kms downvalley of the brine fields, in a smaller settlement area, mud "volcanos" effuse weakly saline groundwater that flows into Onondaga Creek. The clay fraction of the effluent gives Onondaga Creek the appearance of chocolate milk for the ≈26 kms it takes to reach Onondaga Lake. The location of the mud volcanos appears to coincide with an up-valley moving salt front.

The number of mud boils and their areal extent has substantially increased since the onset of brining operations. By characterizing the chemistry of groundwaters in local formations and performing mixing calculations based on mass balance, the volcano effluents were shown to represent a mixture of groundwaters from 3-4 formations. Several working hypotheses are advanced and critically evaluated in an effort to define the dynamics necessary for rapid mud volcano growth in a karst setting.

Location and Geology

Tully Valley is a glacially scoured Finger Lake valley located 24 km south of Syracuse, New York (Fig. 1). The valley floor is blanketed by Pleistocene ice marginal deposits 18 to 134 m thick consisting of gravel/clayey gravel, a middle thick rhythmic clay sequence and a basal sandy gravel (Getchell, 1983; Mullins et al., 1991). Bedrock consists of an interbedded sequence of shales, limestones, and sandstone overlying four evaporite beds ranging up to 21 ± 8 m in thickness (Fig. 2). All beds dip approximately 2° to the south.

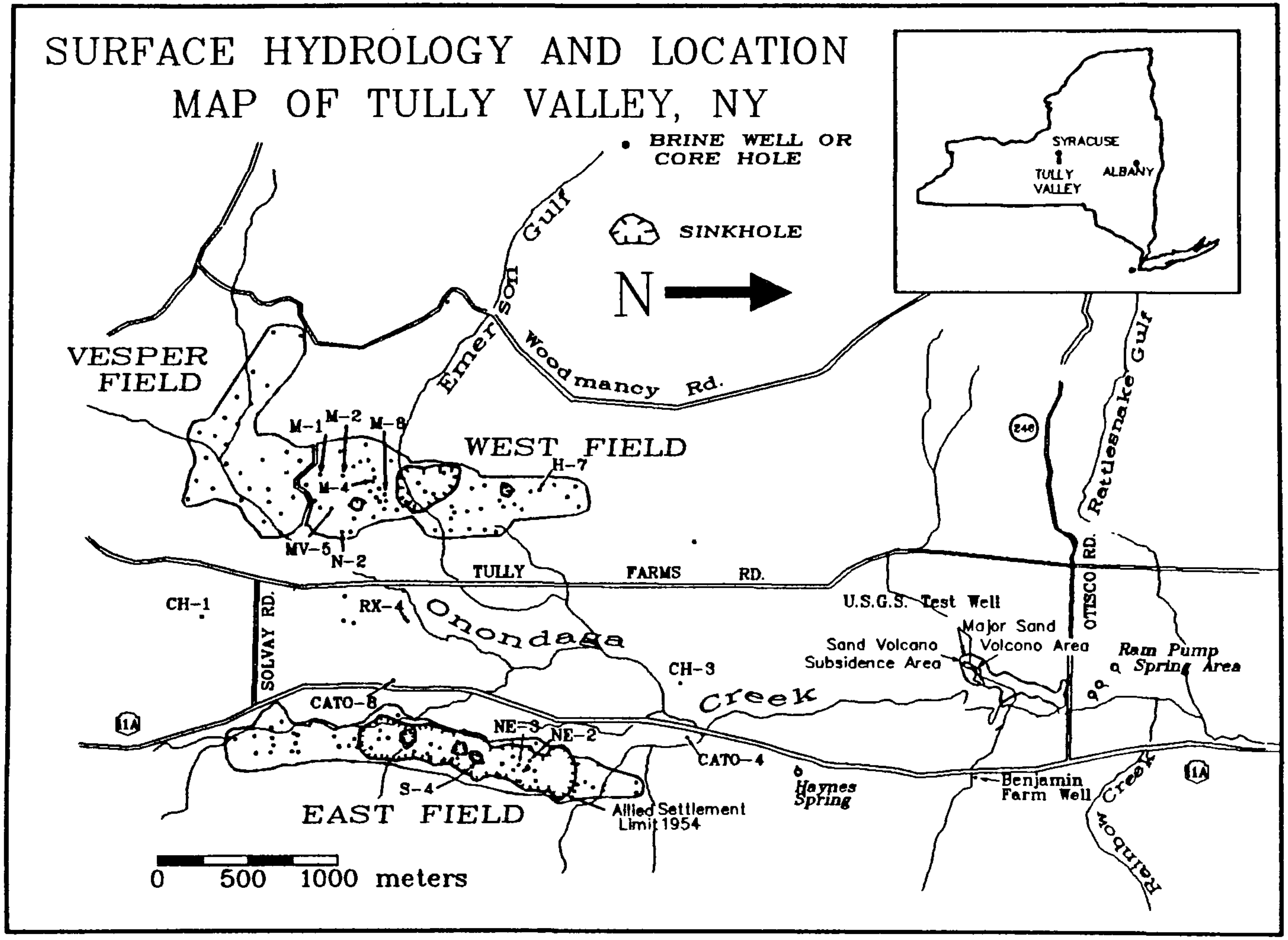

Figure 1:
General location map of the features in Tully Valley, including brining fields and associated collapse.

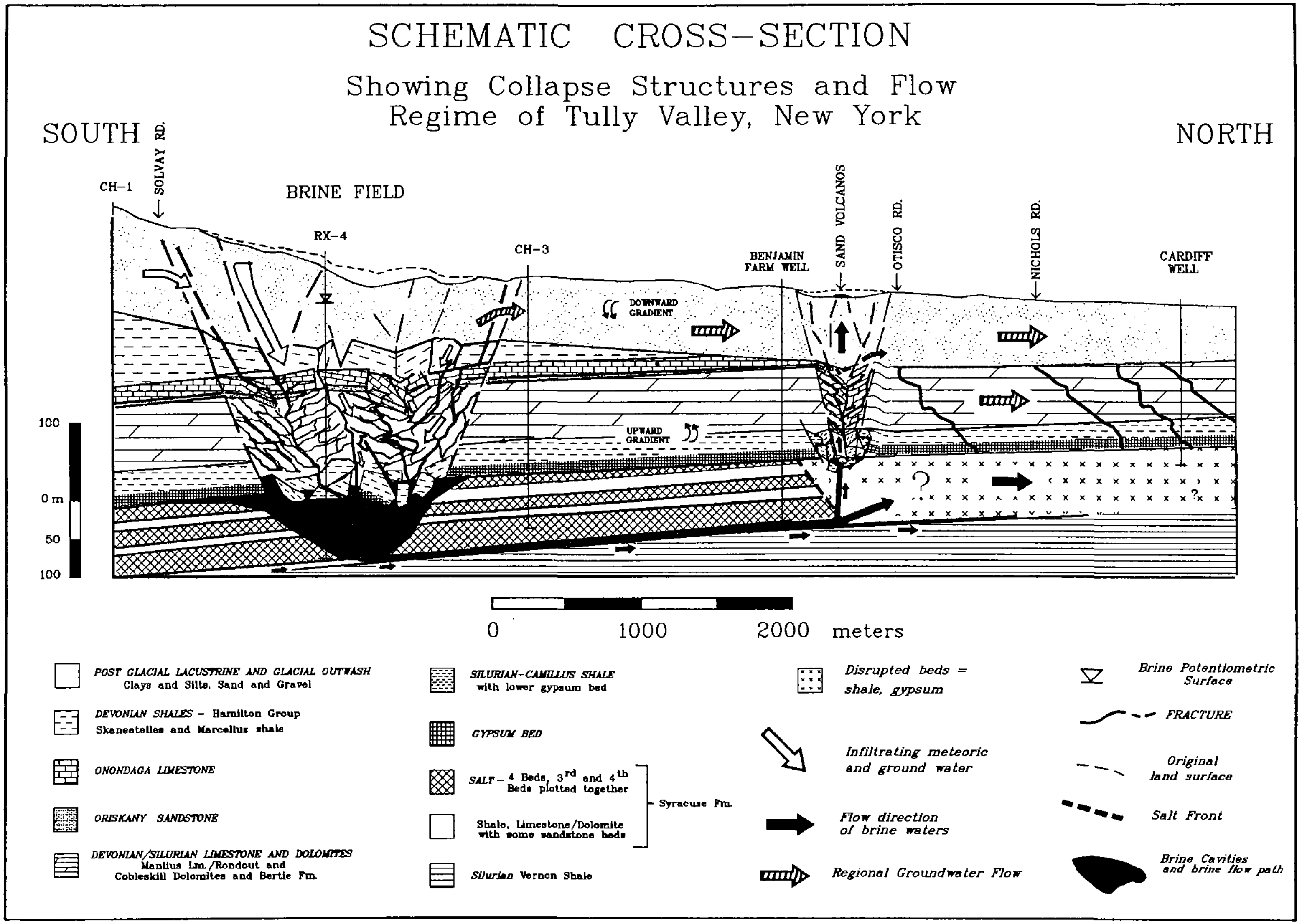

Figure 2: Generalized cross-section of Tully Valley showing stratigraphy, collapse structures and flow regime. See text for further discussion.

Solution Mining Operation and Resulting Subsidence

A solution mining operation was conducted in Tully Valley from 1889 until 1988. Until the late 1950's salt was usually mined by injecting fresh water down one well to the formerly dry salt horizons and then pumping or air lifting brine out another well. Brine was piped downvalley to Syracuse for the production of soda ash. Some 150 wells were drilled, many in close proximity (≈ 40 m) to each other. Continuing brine extraction caused the solution cavities to expand and interconnect. It was not uncommon to drill new wells and find that uncontrolled solutioning had already removed some or all of the upper halite beds (Fig. 3).

Seven large sinkholes formed catastrophically in the Tully brine fields between 1949 and 1980. A monument network was installed and periodically surveyed from 1959 to monitor subsidence. Although the extent of early subsidence will never be known, a steady growth in the subsidence area has been documented. Today, this settlement area is in excess of 550 hectares, has subsided up to 14 meters vertically, and is progressively expanding outward from the cavities. A comparison between the 37 million m^3 of halite extracted and the total settlement volume of 5 million m^3 calculated as of 1982 (Tully, 1983) indicates that only 14 percent of the total possible settlement volume has occurred.

Such extensive settlement resulted from fracturing and subsequent collapse of beds overlying the brine cavities. Evidence supporting this includes the continued growth of the large surficial settlement area, individual sinkhole formation, the presence of large gaping fractures open to the surface, sheared well casings, and the piracy of streams (e.g., Emerson Gulf) into the subsurface. In essence, extensive brine cavity roof collapse and bedrock disruption has resulted in a fracture network of high permeability.

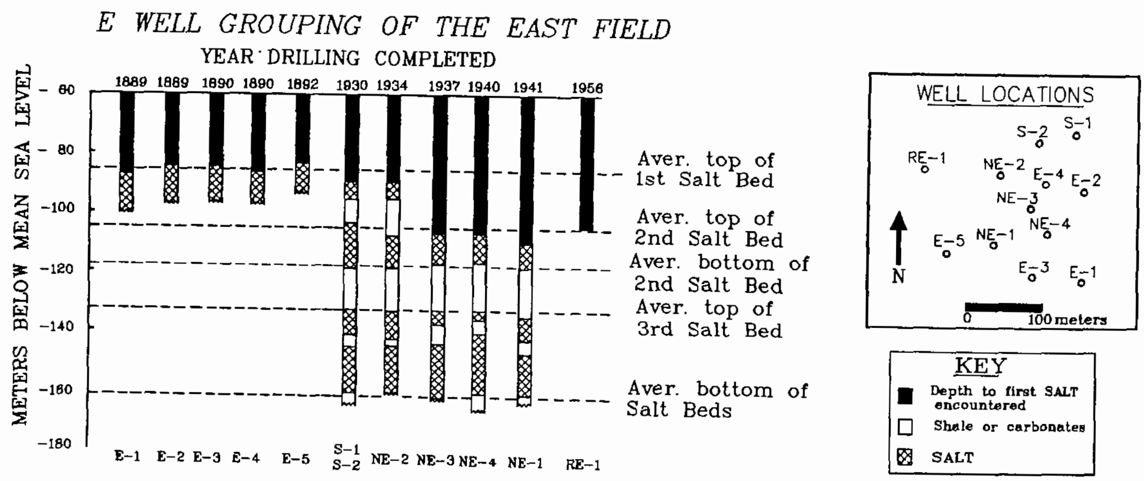

Figure 3:

Time versus depth plot of the E well grouping. By 1937 when NE-3 was drilled, the first salt bed was already uncontrollably dissolutioned by brining activities.

Hydrologic evidence supports the presence of at least one major confining bed near the top of the Manlius Limestone (Fig. 2). Large artesian pressures have been encountered when penetrating this horizon during drilling. Recent subsidence and fracturing has led to interformational mixing of formerly separate hydrologic flow systems and greatly increased the natural recharge to all subsurface formations. Increased recharge meant that by the late 1950's it was no longer necessary to pump fresh water down into the salt horizons to obtain brine during mining operations.

Further evidence of the extent of subsidence-induced fracturing and interformational mixing is found by comparing potentiometric surfaces between pumping brine wells and glacial aquifer wells. For example, brine wells H-7 and MV-5 are 432 and 428 meters deep respectively, while CATO-4 penetrates deep glacial deposits some 1.8 kilometers from well H-7. Measurement of the potentiometric surfaces of these wells over a period of four years by Allied-Signal and the U.S.G.S. reveals similar mimicking responses to brine pumping, although they are some 1.2 kilometers apart and differ in depth by approximately 300 meters. During a period of high brine field pumping (August 1985) the water table in CATO-4 was drawn down 15.7 meters below its "normal" flowing elevation. At the same time the potentiometric surface in H-7 was drawn down approximately 38 meters. Continued observations revealed that the magnitude of fluid level fluctuations in these wells also significantly decreased when pumping rates decreased.

The similar hydrologic response of bedrock and soil aquifers to pumping suggests that the ≈300 m between them is now hydraulically connected by the increased permeability resulting from subsidence-induced fracturing. The damage to the structural integrity of beds beneath and adjacent to the brine fields and the resultant interformational mixing may now be so extensive that they are irreparable. Similarly, the piracy of surface water from Emerson Gulf and the increased recharge through formerly competent lacustrine clays and bedrock must increase the flow through subsurface formations.

Appearance of Mud Boils and Sand Volcanos

Some 2 kms downvalley of the brine fields, in a smaller settlement area, sand "volcanos" effuse weakly saline groundwater that flows into the Onondaga Creek. Projection of geologic units described in distant wells into this area and observation of the growth pattern (N to S, and E and W) of these mud boils suggest that they may mark the present location of the subsurface salt front. Figure 4, time sections 1-3, depict the natural southward migration of the salt front.

An early newspaper documents the presence of a sand volcano at Otisco Road (Fig. 1) in 1899. These early sand volcanos were areally limited. Perhaps they were a reflection of the earliest down valley effects of the solution mining operation. The number of mud boils and their areal extent have substantially increased since the onset of brining operations. Individual effusion features vary in size from small mud boils to broad, flat cones (sand volcanos) measuring 12 m in diameter and 1 m in height.

Historic accounts and the interpretation of historic aerial photography reveal that the mud boil areas expanded rapidly during solution mining operations. Sometime between July 27, 1936 and October 15, 1951, the clay fraction of the mud boil effluent caused Onondaga Creek to change from clear to turbid. Onondaga Creek now has the appearance of chocolate milk for the ≈26 kms it takes to reach Onondaga Lake.

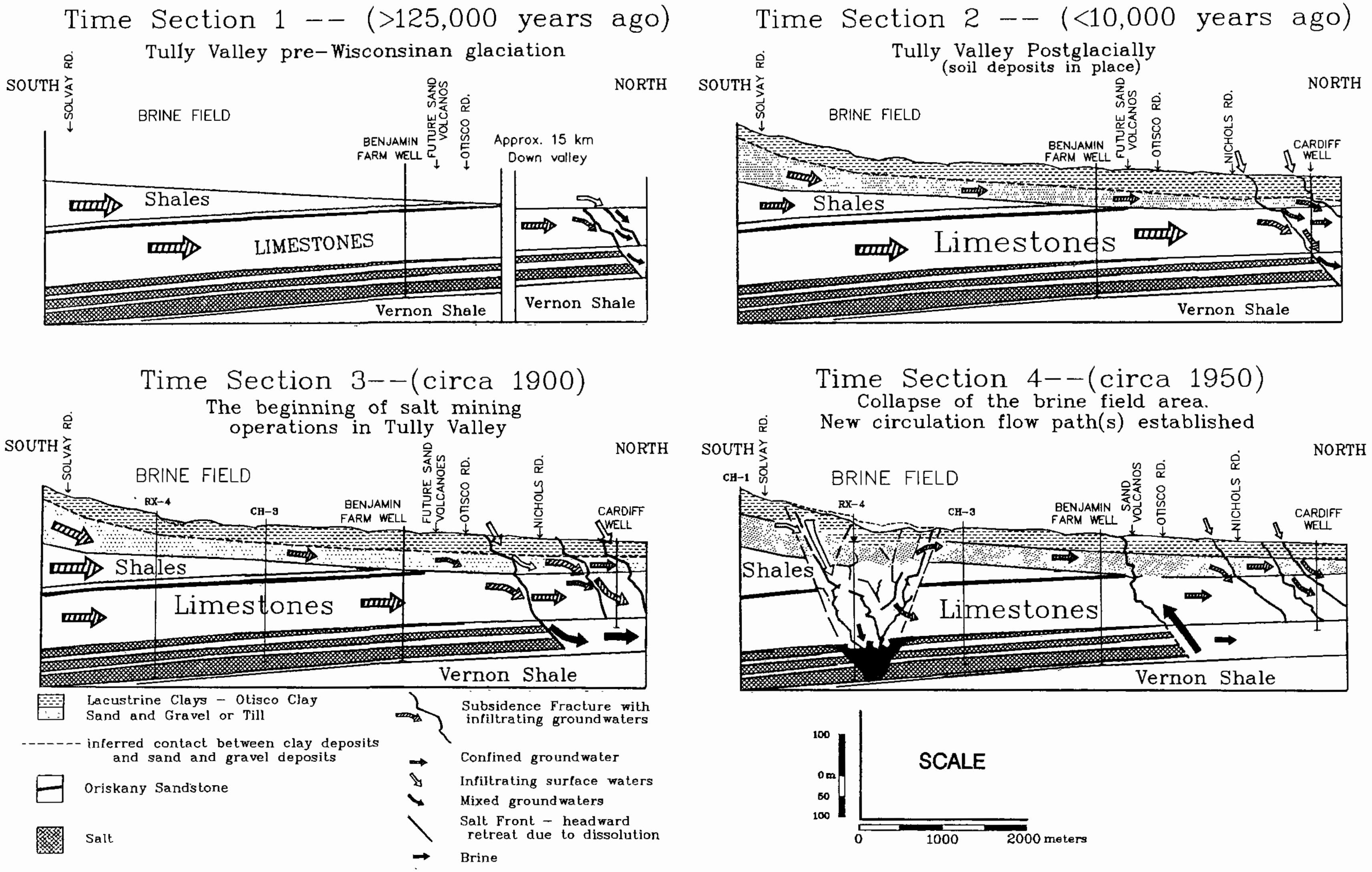

Figure 4: Time sequence diagrams showing the flow regimes and sequence of events that led to the present day. See text for further discussion.

Massive subsidence in the recharge area coincides in time with the greatest sand volcano activity. The largest and most prolifically effusing sand volcano field, approximately 400 meters south of Otisco Road, was not present as of April 28, 1967. By April 29, 1972 it measured an irregular ≈ 60 m by 25 m; by March 28, 1981, ≈ 160 m by 73 m; and on November 29, 1989, 191 by 175 m. Continued settlement, to a depth of 6 m, has further enlarged this area in the last two years. Much of the growth of the mud boil areas appears to develop along the strike of the inferred location of the salt front.

The physical setting of the mud boils is inconsistent with natural springs. Springs typically occur in one or a few closely spaced discharge points and are stable in size and discharge location over time. The artesian Tully Valley mud boils have migrated rapidly southward along Onondaga Creek, and southwestward along a tributary of Onondaga Creek in the last 80 years. This is not in keeping with known short term geologic phenomenon.

Chemistry of Groundwaters and Sand Volcano Effluents

The chemical relationships among uncontaminated groundwaters in Tully Valley and sand volcano effluents (SVE) can be used to identify the source of SVE. Stiff diagrams were found to have limitations when trying to define the chemical relationships between groundwaters and SVE. Because there are many groundwaters that are potential sources, and most have distinctive chemistries, multivariate models were necessary to differentiate among them. The multivariate models give quantitative results in the form of significance levels of chemical discrimination and calculated mixing proportions of groundwaters.

The chemical analyses were taken from three main sources; the New York State Department of Law; the New York State Department of Environmental Conservation; and a thesis by Noble (1990). Many samples from Noble (1990) were taken just outside Tully Valley, but from the same geological formations. Chemical analyses with > 5% error in ionic balance and > 10% error in mass balance were deleted from the sample database.

Water samples were classified into hydrostratigraphic units based on geologic interpretation and well logs. The five units present in Tully Valley in order of increasing depth (Fig. 2) are Glacial Till, Devonian Shale, Silurian-Devonian (SD) Carbonate (Onondaga-Devonian/Silurian carbonates), Evaporite (Silurian shale-Syracuse Fm.), and Vernon Shale (Noble, 1990). The compositions of dilute groundwaters in the Glacial Till and Devonian Shale units are similar, therefore samples from these units are grouped together with the name Near-Surface waters. Average compositions of water from these four units and the SVE are given in Table 1. The SVE contain significant halite and gypsum components.

Statistical Tests

If SVE derive from a single hydrostratigraphic unit, their compositions must be the same within error as samples from that unit. Because samples were analyzed for ≥ 8 chemical species, a multivariate technique called multiple discriminant analysis was used to test for statistical differences in composition between each unit and SVE. Multivariate statistics give at least as good a discrimination as any univariate technique. Although some variables (species concentrations) have non-normal distributions for some hydrostratigraphic units (log transformations do little to correct this), this should not affect the results of the discriminant analysis, although the significance levels obtained may be questionable (Le Maitre, 1982).

TABLE 1

Average Chemical Compositions of Waters in Tully Valley‡

Element	Near-Surface	SD Carbonate	Evaporite	Vernon Shale	SVE	Model #1§	Model #2†
	n=48	n=5	n=5	n=6	n=6		
K	1.05 (0.66)	0.94 (0.46)	232 (20.2)	11.1 (12.4)	6.47 (2.00)	6.47	6.48
Na	37.4 (64.2)	18.2 (12.2)	110,000 (10,500)	51.0 (51.6)	1800 (963)	1900	2650
Ca	73.5 (34.3)	207 (180)	1880 (198)	392 (144)	241 (64.1)	241	239
Cl	54.4 (80.9)	51.6 (30.2)	171,000 (19,900)	117 (130)	3500 (1610)	2980	4160
SO_4	35.1 (35.7)	320 (405)	2300 (1150)	829 (390)	329 (136)	329	350
Sr*	0.69 (0.96)	3.0 (1.3)	76 (11)	5.4 (2.5)	84 (16)	4.0	4.6
Mg*	17.3 (9.15)	32.6 (17.2)	187 (106)	32.3 (16.0)	126 (29.5)	37.1	35.4
HCO_3*	271 (85.3)	314 (35.1)	75.7 (50.9)	274 (73.4)	90.0 (51.4)	401	306

‡ Concentrations in mg/l; standard deviations in parentheses.
* Element not included in model calculations (not conserved).
§ Model #1 proportions of end-members (calculated using non-linear least squares): SD Carbonate = 0.91, Near-Surface = 0.062, Evaporite = 0.024, SSR (sum of squares of residuals) = 2.3 E6.
† Model #2 proportions of end-members (calculated using linear programming and multiple linear regression): SD Carbonate = 0.38, Near-Surface = 0.53, Evaporite = 0.017, Vernon Shale = 0.11, SSR = 3.9 E5.

The objective of discriminant analysis is to find functions that maximize the differences between chemical groups. Specifically, the functions are linear combinations of variables that maximize the ratio of between-group to within-group variances (Le Maitre, 1982). Here the groups are the four hydrostratigraphic units and SVE and the variables are the concentrations in mg/l of the ions Na, K, Ca, Mg, Cl, and SO_4. Use of element ratios did not improve the discrimination between groups. Calculations were made using the program DISCRIM in the SAS software package (SAS Institute, © 1988) using equal weighting for each group.

Over 99% of the variability in the sample data set is accounted for by the two discriminant functions CAN1 and CAN2:

$$\text{CAN 1} = 0.0011*\text{Na} + 0.045*\text{K} - 0.00033*\text{Ca} - 0.0063*\text{Mg} - 0.00043*\text{Cl} - 0.0016*\text{SO}_4$$

$$\text{CAN 2} = 0.000025*\text{Na} - 0.0075*\text{K} + 0.011*\text{Ca} + 0.0083*\text{Mg} - 0.00014*\text{Cl} + 0.00070*\text{SO}_4$$

Species concentrations are in mg/l. The angle between these two vectors is 90°, which allows plotting of the values of CAN1 and CAN2 for each sample on an x-y diagram (Fig. 5). Waters from Near-Surface and SD Carbonate are not significantly different. However, compositions of SVE and waters from the four hydrostratigraphic units are distinctive. The F-statistic shows that SVE are significantly different from all units at the 99% significance level. The SVE therefore cannot form by direct sampling of a single unit.

If waters in SD Carbonate are more saline than our analyses indicate, SVE may be derived directly from this unit. Samples taken from the SD Carbonate unit during drilling of the M-series wells (Phalen, 1928) were in fact more saline than modern samples taken from wells penetrating SD Carbonate. However, multiple discriminant analysis shows that the SVE and M-series samples are compositionally different at the 99% significance level.

Since SVE cannot be derived from a single hydrostratigraphic unit, they must represent a mixture of waters from the various units (chemical end-members). In Fig. 5, lines connecting the end-members define a mixing polygon. Any mixtures formed from the end-members must lie within the polygon. The location of average SVE suggests mixing of Vernon Shale + SD Carbonate + Near-Surface ± Evaporite. The next section discusses quantitative tests of these general conclusions.

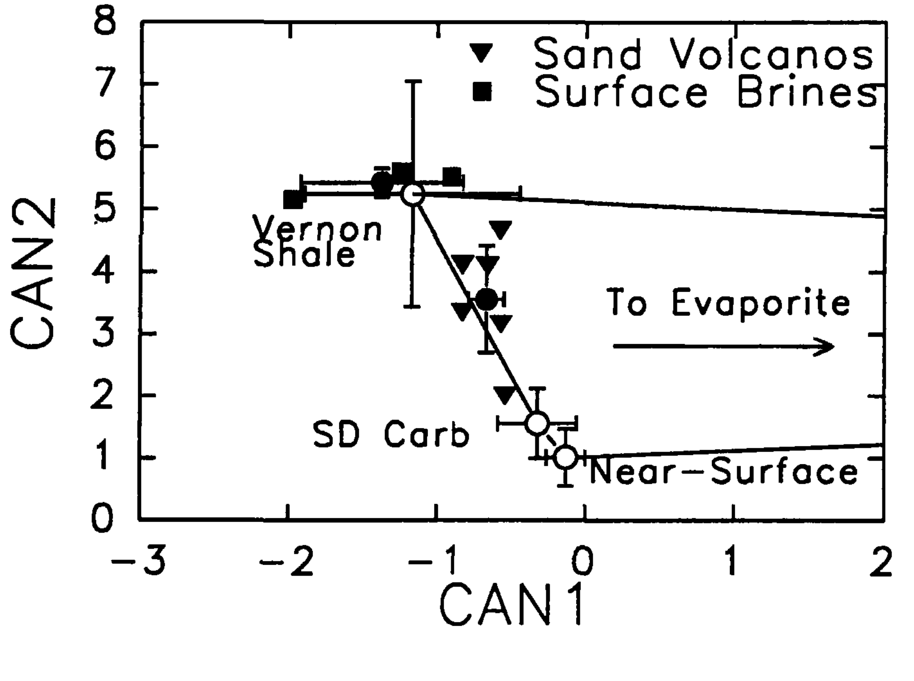

Figure 5: Chemical variation diagram relating compositions of the sand volcano effluents and surface brines to the end-member groundwaters in Tully Valley (Evaporite is off-scale). CAN1 and CAN2 are the functions obtained from the multiple discriminant analysis (see text). Mean values for SVE and surface brines (filled circles) and end-members (open circles) are plotted with ± 1-sigma error bars. Lines connecting the end-members define the mixing polygon.

Mass-Balance Calculations

The process of groundwater mixing can be modelled by mass-balance if species behave conservatively. In the mass balance model, the average composition of SVE represents the mixture and the average compositions of the four units represent the end-members (Table 1). Species are conserved if no dissolution or precipitation of minerals occurs between mixing and sampling. On short time scales precipitation is more likely to occur, but only if the mixture is saturated in a mineral, i.e., the saturation index (SI) is ≥ 1. Calculation of SI for various minerals (Truesdell and Jones, 1974) shows that SVE are saturated only in calcite, dolomite, and strontianite. Species other than Ca, Mg, Sr and HCO_3 should behave conservatively. Because there must be as many species as end-members in the mixing calculations, we were forced to use Ca as a variable, but the fact that the Ca data fit the model suggests that Ca behaves conservatively. Magnesium and Sr have anomalously high concentrations in SVE, and comparison of analyses of filtered and unfiltered samples suggest this results from clay particles that remain suspended even after filtering to 0.45 um.

Two different models were used to calculate the mixing proportions, X_j, of the end-members. The required constraints are that all $X_j > 0$ and $\Sigma X_j = 1$. Model #1 is a non-linear model based on the Marquardt-Levinson algorithm (Press et al., 1985). The objective function minimized was $\Sigma(M_i - \Sigma X_j C_{ij})^2$, where Mi is the concentration of element i in the mixture and C_{ij} is the concentration of element i in end-member j. Model #2 is a linear model that uses multiple linear regression to minimize the same objective function. However, since there is no way to constrain all $X_j > 0$ using multiple linear regression, the first step is to choose those end-members that can mix in positive proportions to form the mixture (Wright and Doherty, 1970). This is accomplished using linear programming based on the simplex algorithm (Press et al., 1985). The optimal proportions of the chosen end-members are then calculated using multiple linear regression. The disadvantage of this approach is that the objective function for the linear programming step (minimize $\Sigma(M_i - \Sigma X_j C_{ij})$) is different from that for the multiple linear regression step. However, this discrepancy is unlikely to have any effect when choosing the best mixing model.

Table 1 shows the results from Models #1 and #2. Although the calculated proportions of end-members are different, both models fit the data within error. Model #1 gives the proportions 0.91 SD Carbonate, 0.062 Near-Surface, and 0.024 Evaporite with SSR (sum of squares of residuals) = 2.3 E6. Model #2 gives the proportions 0.38 SD Carbonate, 0.53 Near-Surface, 0.017 Evaporite, and 0.11 Vernon Shale with SSR = 3.9 E5. Note that the relative proportions of SD Carbonate and Near-Surface have no meaning, since the compositions of these two end-members are not significantly different (Fig.5). Also, the stated proportions of end-members were calculated for average SVE, but the same calculations for each sand volcano effluent show highly variable proportions of end-members. This is to be expected in such a complex mixing scenario.

The fact that the solution to the mass balance problem is non-unique (i.e., averaged SVE chemistry can be explained by several different models) is not surprising, since there are too few high quality constraints on the mixing problem. This results from having too many end-members and not enough elements. Also, the variance of each element is too large, distributions of some variable are non-normal, some elements are correlated, and some end-members have similar chemistries. More data are required before a choice can be made between mixing models. However, we can conclude that SVE waters form from a mixture of groundwaters that are present beneath the sand volcanos, apparently to a depth of at least 335 m. This figure represents the depth to the Vernon Shale, as projected into this area from distant wells.

Many wells down valley of the sand volcanos at Otisco Road have encountered dilute brine waters after penetrating the confining Otisco Clay. Similar waters effuse from a large spring called the Sulfur Hole (≈ 3500 m north of Otisco Rd. and ≈ 500 m west of Tully Farms Rd.), which has been present near the valley flank for at least 130 years. We assume that its waters probably rise along the clay/valley flank contact or through fractures in the clay where it thins out. Discriminant analysis of these surface brines reveals that they are chemically similar to the Vernon Shale (Fig. 5), yet distinct from SVE at the 99% significance level. The average chemistry of these waters is modelled as a mixture of 97.4% Vernon Shale and 2.6% Evaporite. Thus, the natural brines appear to rise undiluted from the Vernon Shale and up-valley Evaporite units because these units are closer to the surface than at Otisco Road and because the uppermost confining beds have been erosionally removed.

Valley Axial Groundwater Flow

Discharge from the sand volcanos responds dynamically with significant infiltration in Tully Valley. This may result from rapid flow through highly transmissive bedrock conduits or by exertion of a large pressure head on confined porous media.

A more complete geologic column to the south limits natural recharge to brine-rich bedrock formations. Similarly, the thick lacustrine clays near the surface of the valley act as a confining bed, severely limiting natural surface infiltration. Several possibilities for the saline discharge flow routes exist; all of which require substantial system recharge to highly transmissive formations (Fig. 2).

Groundwater in deep regional flow systems moves slowly (Fetter, 1988). Rapid flow through deep-seated formations is rare, unless there is substantial fracturing at both the recharge and discharge ends of the flow system. Naturally-occurring fractures are present at the up-dip end of the Tully system, but the mining has provided access routes at the down-dip (brine field) end (Fig. 2). To generate rapid flow through intervening tight bedrock formations requires pathways such as fractures and/or the dissolution of soluble geologic units (e.g., carbonates or evaporites).

A surcharged pressure head acting on highly fractured limestones is one flow route option (e.g., brine field to volcanos). However, the rapid increase in volcano discharge following system recharge would, if transmitted from the brine field to the volcanos, require solution conduits. The natural transmissivity of fractured limestone would not be expected to communicate a hydraulic pressure wave over any appreciable distance due to pulse damping under laminar/non-open conduit flow conditions. Neither should the increased discharge from the sand volcanos originate from valley flank sources, because recharge waters would have to breach density stratified waters and one or more confining beds (e.g., clay and/or bedrock). Fracturing of the thick Otisco clay along the valley flank contact (e.g., Sulfur Hole area) would, if present, probably serve to discharge artesian valley flank and axial pressures rather than serve as a recharge site.

In evaluating whether the limestones would be preferred flow routes one would have to assess the likelihood that solutional conduits were present pre-glacially between the brine field and the volcanos. Solutional conduits develop in limestone only where a pre-existing network of integrated openings connect the recharge and discharge areas (Palmer, 1991). Furthermore, carbonate dissolution requires that undersaturated water remain in contact with soluble fracture or conduit walls. A minimum of 10,000 years is required to widen joints and partings enough to produce significant permeability (Palmer, 1984; Dreybrodt, 1990; Palmer, 1991). A typical water sample obtained from the Devonian shales contained 51.9 mg/l calcium, or 129.6 mg/l $CaCO_3$ equivalents. For this concentration, the rate of wall retreat (e.g., joint enlargement) in limestone is calculated to be 0.05 cm^3/liter in 10,000 years (Palmer, pers. comm.). Based on an evaluation of these considerations, it is our opinion that geologic conditions have not been favorable pre- or post-glacially for the development of significant solution conduits in the limestones.

The authors evaluated several other lines of reasoning in assessing the likelihood that the main flowpath between the brine fields and the sand volcanos occurs in the limestones. This included an evaluation of natural infiltration pathways for calcite undersaturated waters. Geologic logs reveal that little or no exposed limestone was present up valley of the sand volcanos pre-glacially. Furthermore, much of the mud boil subsidence has occurred along bedrock strike, which is inconsistent with typical downdip vadose cavern development in this type of geologic setting (Palmer, 1991). Not only are the effusion areas migratory in nature, they exceed the largest natural sinkhole size in New York State by well over an order of magnitude. Conditions often attendant to sinkhole formation, such as aquifer dewatering, were also not present. The variable mud boil discharge in recent time would not be expected from a naturally deep-fractured karst aquifer system if it was physically separated from large scale recharge.

The hypothesis might be advanced that the artesian pressures and volcano effluents may derive from infiltration entering alluvial fans at the outlets of either Rattlesnake Gulf or Rainbow Creek (Fig. 1). The chemistry of the SVE would require this water to breach density stratified brines and discharge up-valley against the regional hydraulic gradient. In addition, significant recharge would have to completely penetrate the thick confining Otisco clays. It is the combination of these unlikely events and the areally wide and migrating distribution of the mud boils that virtually removes all potential sources of system recharge other than the brine field settlement area. It is more likely that any infiltration into the Rattlesnake Gulf alluvial fan flows a short distance below the ground surface, daylighting in the Ram Pump Spring area (Fig. 1).

The variable discharge rate and the chemistry of SVE are critical to our understanding of likely source areas. The presence of gypsum and halite components in the effluent is a clear signature telling us that at least some volcano effluent must derive from the deep evaporite unit. The salt content of SVE is diluted by shallow groundwater in sediments and bedrock in the valley. Geochemically, the SVE represent mixtures of waters from all of the hydrostratigraphic units beneath the sand volcanos to a depth of 335 m. SVE have moderately variable salinities, but groundwaters from the hydrostratigraphic units show much less variation. The chemistry of SVE suggests fluctuating amounts of dilution of deep saline and gypsiferous waters by near-surface fresh waters. The brine must originate at a location having a higher hydrostatic head than at the sand volcanos. Mixing occurs during ascent along vertical pipes (joint or fracture flow paths) with discordant hydraulic efficiencies.

Two separate flow regimes are probably responsible for the response of volcano discharge following recharge events; deep evaporite and shallow confined sediments. The high salinity of SVE observed during storm events may indicate the development of a subsurface flow route, first through fractures in the Vernon shale and then through salt dissolution and conduit formation along the shale/salt contact (Fig. 2). Density-driven circulation initiating at the brine fields may now rise along the fractured U-shaped (areally) salt front beneath the volcanos. Jenyon (1986), discusses the natural circulation systems attendant to salt fronts. Formation of fractures at the salt front (Fig. 4, sections 1-3 and Fig. 2) normally results in infiltration and downvalley flow. Instead, water rises at the mud boils, almost as if the flow system was artificially short-circuited (Fig. 4, section 4). The large sub-vertical arrow in Time Section 4 of Figure 4 indicates upward flow (not against the regional flow gradients) along salt front fractures resulting from this short-circuited circulation.

Alternately, the rise of saline waters south of Otisco Road may directly relate to the erosional removal of critical confining beds near the sand volcanos. Whatever the flow path of the saline volcano component, the storm discharge of the volcanos requires conduits or a confined porous media capable of rapidly transmitting a recharge pressure pulse.

Several flow routes have been hypothesized to explain the rapid increase in volcano effluent following recharge events, but the actual pathway is unimportant. Visual observation of increased volcano discharge during periods of high infiltration indicate that the source area responds in tandem with surface recharge, operating through and beneath the thick Otisco clay confining bed. It is unlikely that significant storm infiltration through this clay can naturally occur to cause the increased discharge observed at the mud boils. It is more likely that the sudden surcharge on the hydrologic system from extensive infiltration in the brine field settlement basin acts as a pressure pulse. Rubin has proposed several means of verifying these relationships.

The sudden increase in pressure head, exerted through one or more saturated formations, is partially released at the mud boils. While some of the volcano discharge originates in the evaporite units, much of the observed discharge may reflect a pressure wave being transmitted through the sandy gravel aquifer confined beneath the Otisco clay. As sand is entrained and removed from below the clay, a void develops until the mechanical strength of the roof is exceeded. Ring fractures develop, the subsidence area grows, and the efficiency of the exit pathways continues to increase. The SVE are piped upward to an elevation well in excess of the shallow water table common to valley centers. The only viable explanation for the storm discharge observed at the sand volcanos (Fig. 6) is from recharge through severely fractured brine field clay and/or bedrock. Here, interformational mixing of groundwaters from massively fractured clay and bedrock formations has resulted in a vast pressure head on a saturated flow network.

Fluids from the deep saline beds are successfully diluted at this time. It is the quantity of discharge that is of concern. This turbid water degrades Onondaga Creek, Onondaga Lake, the aquatic ecosystem, and the aesthetic quality of the valley.

Conclusion

Solution mining operations resulted in large-scale subsidence. Subsidence-induced fractures serve to significantly recharge formerly confined flow systems. This increased recharge has resulted in the formation of sand volcanos downgradient. The bulk of the sand volcanos appear to have formed by a sequential combination of: 1) increased recharge to formations above solutioned out evaporite beds that has unnaturally surcharged the flux through geologic units below the brine fields. This flow through has created an unnatural circulation pattern that vents upward at the sand volcanos. Pumping of brine increased recharge through formerly competent and confining beds leading to the 2) solution of salt and the transmission of a recharge pressure pulse through one or more geologic horizons. This water 3) rises under artesian pressure through pre-existing fractures underneath the sand volcanos, 4) mixing with other groundwaters during ascent. Rising waters 5) suspend unconsolidated sediments (e.g., clay, silt, and sand) and discharge to Onondaga Creek via the sand volcanos.

Figure 6: Typical Tully Valley sand volcano.
Diameter is approximately 2 meters.

Acknowledgement

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Solution Mining And Resultant Evaporite Karst Development In Tully Valley, New York

Paul A. Rubin, John C. Ayers, and Kristin A. Grady

1. What is the likelihood that the sand boils are a result of the expulsion of water due to progressive collapse of cavities created by ongoing solution along the upper portion of the salt by water entering at cracks?

The process of cavity collapse, exemplified by sinkhole formation in limestone karst settings, occurs at either an extremely slow rate as soil is sapped downward and removed or as a catastrophic event. The downward sapping of sediments and rock debris from above the salt horizons would occur so slowly that it could not be expected to continuously displace quantities of water. If catastrophic cavity collapse were the driving force behind sand volcano effusion, then sand boils would form instantaneously, resulting in a high effusion rate immediately following cavity collapse. This effusion rate would rapidly drop to zero. Without a large and continuous fresh supply of rock or sediment to displace groundwater, the sand boils would soon be removed by erosion. Therefore, the likelihood that the sand boils are a result of the expulsion of water due to progressive collapse of cavities is unlikely. Instead, we see a slow, steady growth of the sand boils, and although the effusion rate may be variable, there is always significant flow.

2. There is an incredibly vast, international literature on the effects of both natural and anthropogenic solution of salt in the subsurface. I know of at least 100 references and could probably find 1000. Some of the most useful that come to mind are:

Calvert, A.F., 1915. Salt in Cheshire, Spon, London, 1206 p. (includes many photos).

Evans, W.B. et. al., 1988. Geology of the country around Macclesfield ... Geological Survey of Great Britain, Memoir (plus a more recent memoir on Cheshire published 1987±).

Reuter, F. and Tolmanacer, V.V. 1990. Bauen and Bergbau in Senkungs und Erdfallgebieten. Schriftenreihe fur Geogische Wissenschaften, Akademie Verlag, Berlin, No. 28., 176 p.

Weber, H. 1952. Pliozan und Auslaugung im Gebiet der Oberen Werra, Geologica Akademie Verlag, Berlin, v.8, 136 p.

Session V:

Case Histories

Hydrogeology and Ground-Water Monitoring of an Ash-Disposal Site at a Coal-Fired Power Plant in a Karst System

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Abstract

A coal-fired power plant located along a lake in south-central Kentucky is situated on karstified Mississippian-age carbonates. Ground-water flow and chemistry are evaluated to assess impacts of current and past coal-ash-disposal practices. Ash disposal in an unlined, active slurry pond is allowing ash and pond water to enter the ground-water system through underlying sinkholes and fractures. Water infiltrating through 20-year-old ash in a closed-out slurry pond enters the ground-water system similarly. Ground water discharges at springs near lake pool elevation. Sinkholes, swallets, and springs were field mapped, and a dye-tracing plan was implemented. Dye traces indicate separate flow systems for each disposal site. Dry fly ash was leached to determine constituents present in fresh ash. Water-quality analyses were obtained from upgradient and downgradient springs and surface-water sources. Wells installed in the closed-out slurry pond were sampled. Monitoring data indicate that major ion composition in the active slurry pond is more similar to the lake water used for the ash slurry than natural ground water. Active-pond water is elevated in sulfate, barium, boron, copper, iron, manganese, and zinc compared to background waters. Spring discharge associated with the active slurry pond shows major ion composition intermediate between pond water and natural ground water. Boron, iron, manganese, and zinc are elevated in these springs. Spring discharge associated with the closed-out slurry pond is similar to background springs except for increased concentrations of boron and iron. Water samples collected from wells drilled into the ash are similar in major ion composition to background springs but show elevated concentrations of barium, boron, iron, manganese, and vanadium.

Introduction

The purpose of this research is to study the ground-water flow system and chemistry in a karst aquifer underlying a coal-fired power plant. The results of this research will be used by the power company to develop a ground-water-monitoring program which meets the requirements of the state regulatory authority.

Coal ash is an aluminosilicate glass which forms from the burning of coal at temperatures of approximately 1500 degrees celsius (Elseewi, 1980). Ash is composed primarily of silicon, aluminum, iron, and calcium oxides. Lesser amounts of sodium, magnesium, potassium, and sulfur are typically present (Cherkauer, 1980; Elseewi, 1980). The major constituents are derived from thermal breakdown of minerals, primarily illite and kaolinite. As a result, the composition of coal ash is highly dependent on the composition of the pre-combusted coal.

Coal ash consists of two products. Fly ash is fine grained and often forms concentrically layered spheres. This fraction accumulates in electrostatic precipitators and comprises approximately 20 percent of the total ash fraction at the study site (Rai and Zachara, 1990, personal commumication). Bottom ash is coarser grained material which falls to the bottom of the combustion chamber.

Elements associated with ash may be incorporated into the aluminosilicate matrix of individual particles or may accumulate on the outer surface of the spheres. Many elements volatilize during combustion and subsequently condense on the outer surface of the cooling particles. Antimony, boron, fluoride, selenium, arsenic, cadmium, lead, zinc, barium, chromium, copper, chlorine, sulfur, and vanadium are often preferentially enriched on ash particles in this manner (Ainsworth and Rai, 1987; Rai, 1987). Aluminum, cobalt, iron, manganese, nickel, strontium, and vanadium are also associated with coal ash, though not generally enriched on the surface of the particles (Wu and Chen, 1987; Rai, 1987).

Setting

The power plant was constructed in 1964 and houses two generating units which produce a total of 320 megawatts of electricity. Maximum coal usage is 5,000 tons per 24-hour period.

The site has two ash-disposal facilities (Figure 1). From 1964 to 1978, coal ash was sluiced to a 40-acre slurry pond. This disposal facility was retired in 1978 when its storage capacity was reached. The maximum thickness of ash in this facility is 35 feet.

Since 1978, ash has been disposed of in an unlined pond that has an area of approximately 50 acres. Fly ash and bottom ash are transported to this facility as an ash-water slurry via a 3,500-foot-long pipeline.

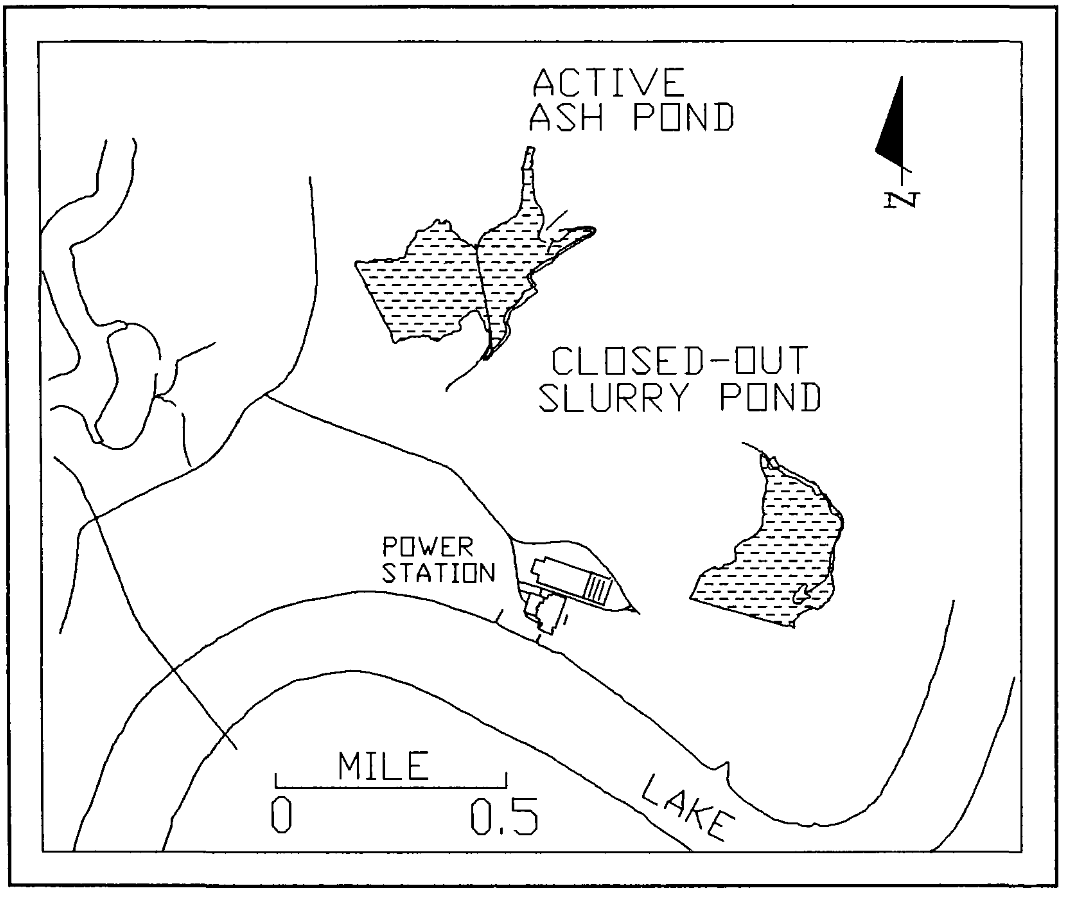

Figure 1. Map showing location of power station and ash-disposal facilities.

The plant is located in the Mississippian Plateau carbonate region of Kentucky, adjacent to a major impounded river. The topography is a rolling, upland, karst plain which exhibits subsurface drainage. In the immediate vicinity of the plant, the land surface is dissected by small v-shaped valleys. Surface flow in these channels is ephemeral and often disappears into swallets in the stream bed. The land surface drops steeply, often vertically, from the upland karst area on which the plant is situated to the surface of the lake.

Rocks underlying the site are primarily limestone and dolomite (Figure 2). Structures associated with the plant are constructed on the Ste. Genevieve Limestone and St. Louis Limestone. The Salem and Warsaw Formations (undifferentiated) crop out a few feet above permanent lake pool elevation.

The Salem and Warsaw Formations, which are considered together because of their lithologic similarity, are the stratigraphically lowest formations that crop out near the plant. Total thickness ranges from 30 feet to 75 feet (Lewis, 1974; Taylor, 1975). The two formations represent an intertonguing sequence of dolomite, limestone, shale, siltstone, and sandstone, which shows significant lateral variation. The exposed section of Salem and Warsaw near the

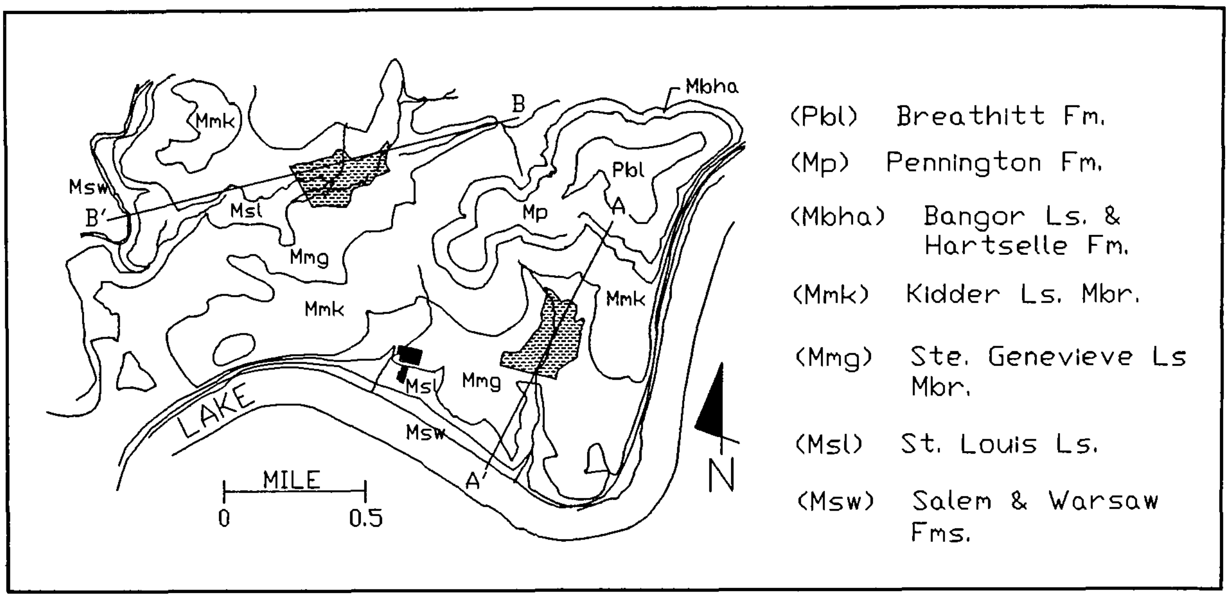

Figure 2. Geologic map of the study area.

plant is predominantly crossbedded limestone underlain by a thin-bedded, argillaceous dolomite.

The St. Louis Limestone lies directly above the Salem and Warsaw Formations. It is 55 to 120 feet thick (Lewis,1974; Taylor, 1975) and is predominantly cherty, argillaceous, fossiliferous thin- to thick-bedded limestone with interbeds of claystone and siltstone.

The Ste. Genevieve Limestone, the stratigraphically highest formation associated directly with plant facilities, is 30 to 80 feet thick (Lewis, 1974; Taylor, 1975). This unit is predominantly micrograined to medium-grained limestone.

The study area is situated in a fluviokarst terrain. Sinkholes occur in clusters or as isolated features throughout the area. The headwaters of a well-defined surface drainage system originate in knobs capped by Pennsylvanian sandstone northeast of the site. As these streams descend into the Mississippian-age limestones, swallets and surface fractures direct the drainage underground where it later emerges as springs along the lake. Downward movement of ground water appears to be inhibited by argillaceous zones interbedded with limestone in the Salem and Warsaw Formations.

Methods

Extensive field mapping of the site and immediate vicinity was completed in 1990. Field personnel verified the existance of sinkholes shown on topographic maps and mapped additional sinkholes and swallets. One hundred thirty-two sinkholes were field mapped. Sinkhole locations are shown on Figure 3.

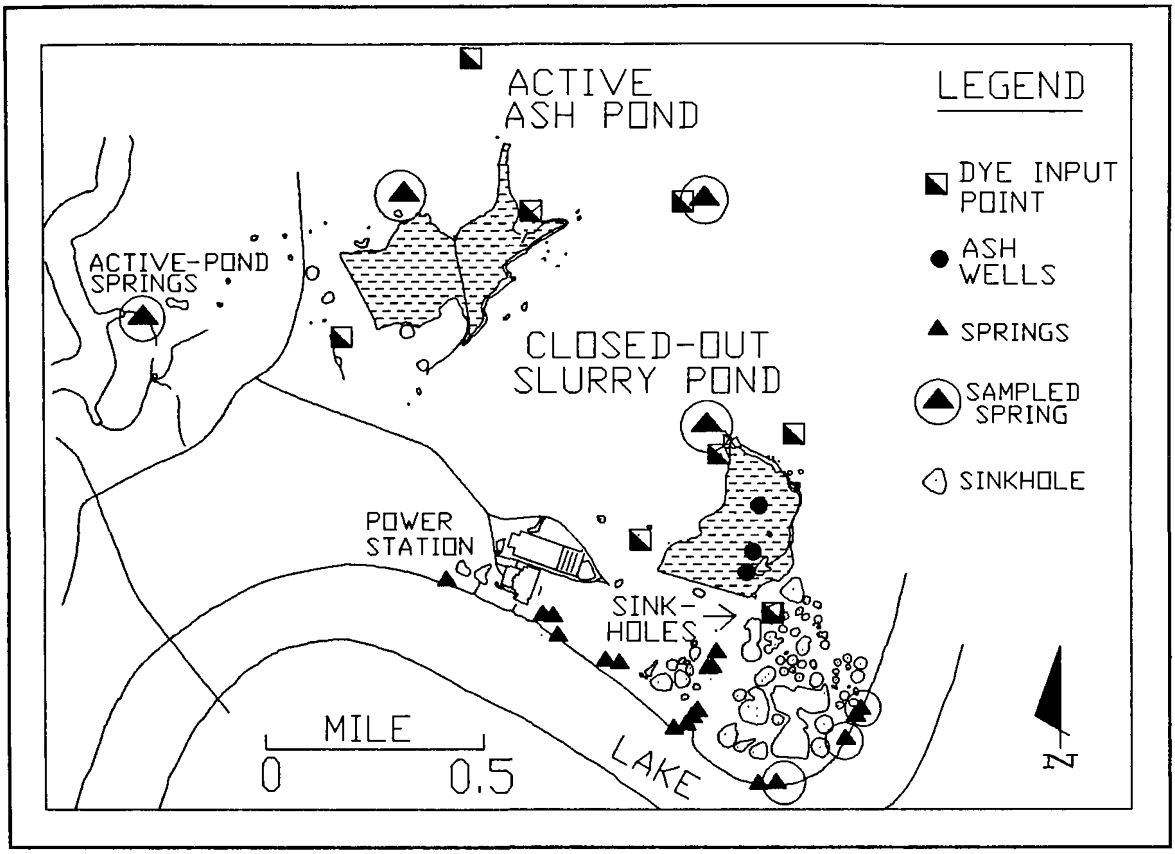

Figure 3. Hydrogeologic features map.

A survey was conducted to locate springs potentially related to subsurface drainage from the site. The survey began in late winter when lake levels were highest. As summer approached, lake levels declined and additional springs became visible. Mapped springs are shown on Figure 3.

A submerged spring situated about 22 feet below permanent pool elevation was noted by plant employees during the time the closed-out slurry pond was an active facility. It was thought that this spring may represent a major discharge point from the conduit system associated with the closed-out slurry pond and that stratigraphically higher springs may be wet-weather overflows. During early spring of 1990, the spring was documented when a conductivity plume was delineated relative to ambient lake conductivity at the same depth. In July 1990, an attempt was made to locate this spring using SCUBA but the attempt was not successful.

By October 1990, the lake level had fallen and exposed a small seep at the top of the agillaceous dolomite zone in the Salem and Warsaw Formations. It was concluded that this spring maintains a very low flow in the summer and fall and is not a major discharge point.

A dye-tracing program was initiated in the spring of 1990 to determine hydraulic connections between swallets and springs located around the periphery of the site. The program was developed with the assistance of Dr. John Thrailkill, Professor of Geology at the University of Kentucky, and James Currens, a hydrogeologist at the Kentucky Geological Survey.

The fluorescent tracers Fluorescein LT (Acid yellow 73, CI 45350) and optical brightener (Tinopal CBS-X) were chosen for this study. Fluorescein imparts a fluorescent green color to water, and optical brightener fluoresces blue-white under ultraviolet light. These dyes have been used in the karst regions of Kentucky by both Thrailkill and Currens. Fluorescein and optical brightener do not interfere with one another and can be used concurrently.

Passive detectors were used to adsorb the tracers from the water. Laboratory-grade activated charcoal in nylon screen bags was used for fluorescein adsorption. Cotton broadcloth test fabric stretched over a wire frame was used for adsorbing optical brightener.

Prior to the addition of dye into the conduit system, a background test of fluorescence was conducted using both types of detectors. No evidence of potential interference with either dye was detected.

Several measures were taken to prevent accidental contamination of detectors with dye. Dye and detector materials were not stored or handled in the same room, nor were dye and detectors handled on the same day. To minimize handling of dye during the course of the project, all bulk dye products were prepackaged into 50, 100, and 200 gram bottles before any dye tracing was initiated. As an extra precaution, bottles of dye were sealed inside plastic bags. Once the amount of dye needed for a trace was calculated, the appropriate amount was chosen from the prepackaged selection. Detector placement and dye injection were always performed on different days.

The amount of fluorescein introduced into each point was calculated from the following formula (Aley and Fletcher, 1976):

$$W_d = 1.478\ D \times (Q/V) \quad \text{where:}$$

W_d = Weight of dye in kg
D = Distance travelled in km
Q = Approximate discharge in m^3/sec
V = Approximate velocity in m/hr

The calculated amount was rounded to the next even 50 grams.

For optical brightener, Quinlan (1986) suggests that doubling the amount calculated from the above formula works well for the average Kentucky spring. This rule of thumb was used for determining the amount of optical brightener to inject.

Eight sites were selected for dye injection, four near each disposal facility. One detector site was located at the submerged spring. Detectors were attached to a rope anchored to a 30-pound weight on the lake bottom and could be raised and lowered using small pulleys. A float was tied to the anchor in order to mark the site. Use of this method allowed easy access from a small boat.

Seven of the eight dye traces were successful. One fluorescein dye trace was not detected at any of the detectors. Since laboratory tests showed that fluorescein is strongly adsorbed onto coal ash, it is possible that this trace was lost as a result of adsorption onto ash which may be present in the conduit system. It is also possible that not enough water was used to flush the dye through the system.

Three ground-water-monitoring wells were installed in the closed-out slurry pond. Well locations are shown in Figure 3. These wells were used to monitor water levels and to collect water-quality data in the dewatered, closed-out facility. Each well was completed in ash using 10-foot-long screens, the bottoms of which were placed at the top of bedrock.

Water samples were obtained from each monitoring well during March and May to analyze for non-metals. One set of field-filtered samples was collected for dissolved-metals analyses. Each well was purged prior to sampling. When possible, three well volumes were removed. Low-yield wells were purged until dry, then allowed to recover enough to permit sampling. A bladder pump was used for sampling when there was sufficient water volume. Wells with little water were sampled using a stainless steel bailer with a bottom-emptying device. Springs were sampled by filling a sample bottle with water obtained from near the spring mouth.

Water samples for dissolved-metals analyses were preserved immediately upon collection using nitric acid. All samples were placed on ice immediately after collection and delivered to the laboratory the next working day. Water-quality analyses were performed by the Kentucky Geological Survey laboratory. Analyses were done according to EPA procedures described in "Methods for Chemical Analyses of Water and Wastes." Metals analyses were performed using the Inductively Coupled Argon Plasma method (ICAP).

Fresh, dry ash samples were collected directly from the hoppers within the plant. Numerous leaching tests of varying duration were conducted by Battelle-Pacific Northwest Laboratory on fly ash and bottom ash through a range of solid/solution ratios. Data were obtained on elements associated with the ash fractions and their leachability.

Results

Dye-trace results show that the active disposal pond and the closed-out pond are located in different ground-water-flow systems. Figure 4 shows the dye input points and the resulting positive traces.

The lower boundary of karst development in the closed-out slurry pond flow system is located approximately 30 feet below the top of the Salem and Warsaw Formations. The lower boundary is marked by an argillaceous dolomite layer (Figure 5). Spring

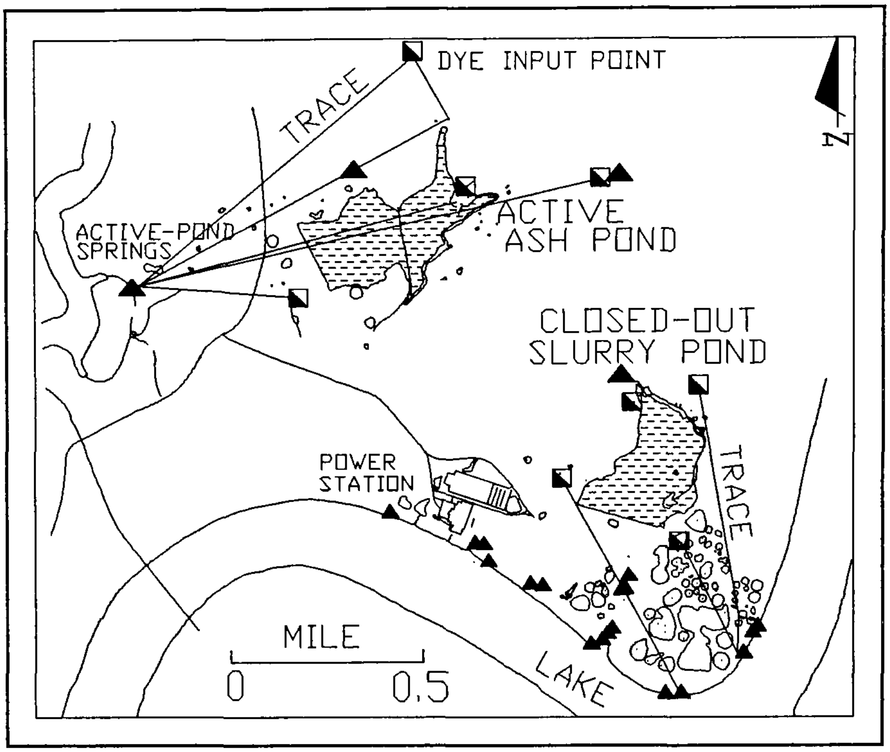

Figure 4. Locations of positive dye traces.

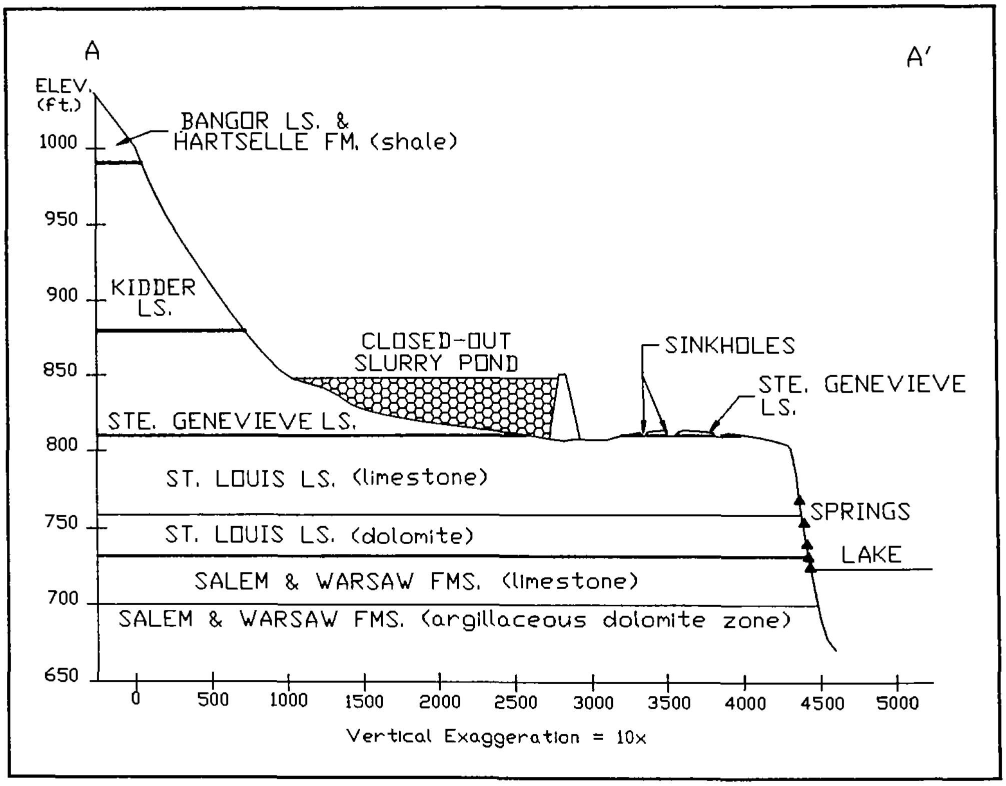

Figure 5. Cross section through closed-out slurry pond. Location of cross section is shown on Figure 2.

flow in the closed-out pond system varies seasonally. The greatest flows generally occur in the winter and spring and the lowest flows are in the summer and fall.

The lower boundary of karst development in the active-ash pond flow system appears to be argillaceous zones at the top of the Salem and Warsaw Formations. All springs associated with the active pond occur in the lower section of the St. Louis Limestone (Figure 6). Spring discharge from this system remains constant throughout the year.

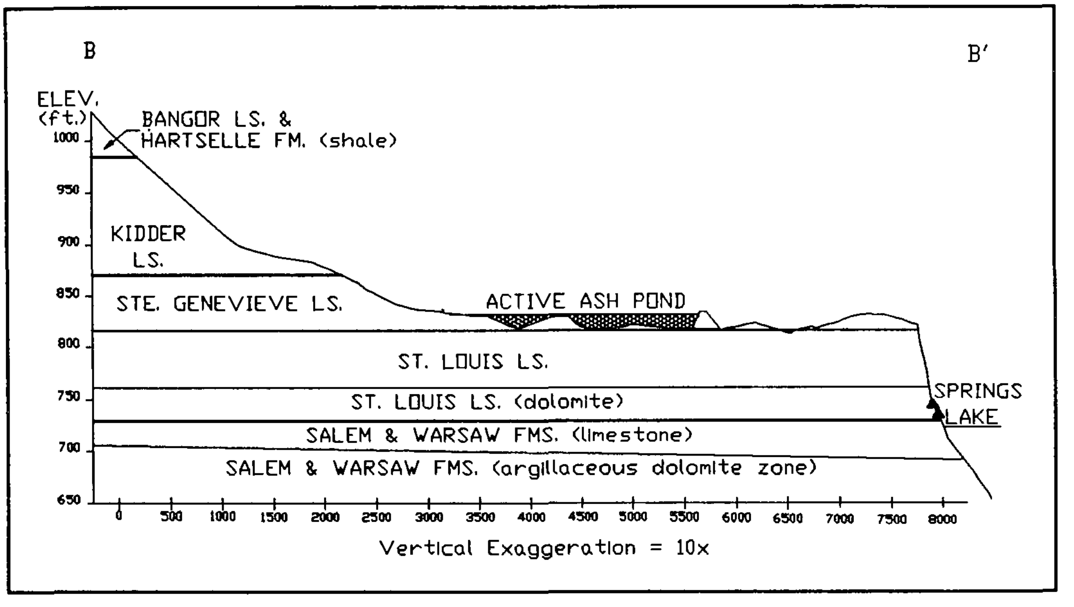

Figure 6. Cross section through active ash pond. Location of cross section is shown on Figure 2.

Water-quality analyses were grouped into the following six categories for comparison among groups:

1. background spring quality (three sites for all analyses),
2. monitoring wells in the closed-out disposal pond (three sites for all analyses),
3. springs identified by dye traces as being in the active-pond flow system (seven sites for non-metals analyses and two sites for dissolved-metals analyses),
4. springs identified by dye traces as being in the closed-out pond flow system (three sites for non-metals analyses and two sites for dissolved-metals analyses),
5. surface-water samples collected from various points in the active pond (five sites for all analyses), and
6. lake water used for sluice water (one site for all analyses).

Tables 1 and 2 summarize the maximum and minimum values for constituents tested.

Table 1. Maximum and minimum concentrations for dissolved metals (mg/L).

Constituent	Background springs[1]		Wells in closed-out ash pond[2]		Springs down-gradient of closed-out ash pond[2]		Springs down-gradient of active pond[2]		Surface water samples of active pond[3]		Lake water
	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	1 value
aluminum	.139	<.027	.057	.030	.118	.101	.117	.099	.047	<.027	.086
antimony	<.017	<.017	<.017	<.017	<.017	<.017	.020	<.017	<.017	<.017	<.017
arsenic	<.028	<.028	.071	<.028	<.028	<.028	.029	<.028	<.028	<.028	<.028
barium	.032	.028	.125	.082	.044	<.001	<.001	<.001	<.001	<.001	<.001
beryllium	<.001	<.001	.001	<.001	<.001	<.001	<.001	<.001	<.001	<.001	<.001
boron	.020	<.006	1.06	.657	.175	.175	.081	.061	.231	.125	.011
cadmium	<.003	<.003	<.003	<.003	.003	<.003	<.003	<.003	.006	<.003	<.003
calcium	123	85.1	70.2	53.3	73.9	71.7	82.9	82.2	28.7	26.1	16.9
chromium	.010	<.006	.006	<.006	.009	.008	.010	.010	<.006	<.006	<.006
cobalt	<.004	<.004	.004	<.004	<.004	<.004	<.004	<.004	.005	<.004	.004
copper	.008	.005	.005	<.004	.011	.008	.010	<.004	.095	.009	<.004
iron	.116	<.004	.050	.028	.157	.142	.198	.154	.730	.006	.008
gold	<.004	<.004	.007	.004	<.004	<.004	<.004	<.004	<.004	<.004	<.004
lead	<.038	<.038	<.038	<.038	<.038	<.038	<.038	<.038	<.038	<.038	<.038
lithium	<.006	<.006	.118	.110	.028	.024	.013	.011	.138	.119	<.006
magnesium	8.62	5.55	8.35	6.95	7.88	7.74	6.12	6.02	8.42	7.20	7.40
manganese	.010	<.001	.654	.160	.040	.015	.005	.001	.100	.043	.006
nickel	<.012	<.012	<.012	<.012	<.012	<.012	.013	<.012	.035	.012	<.012
phosphorus	<.068	<.068	.463	<.060	<.068	<.006	<.068	<.068	<.068	<.068	<.060
potassium	<1.32	.400	5.79	5.67	2.80	2.71	2.14	1.70	4.64	4.20	<1.32
selenium	<.052	<.052	<.052	<.052	<.052	<.052	<.052	<.052	<.052	<.052	<.052
silicon	3.01	.074	10.5	4.62	1.44	1.27	1.38	.992	2.55	.581	.073
silver	<.003	<.003	<.003	<.003	<.003	<.003	<.003	<.003	<.003	<.003	<.003
sodium	6.67	1.77	13.2	8.55	11.84	10.1	5.63	2.98	14.08	6.94	12.2
strontium	.028	.252	2.29	1.71	.399	.398	.404	.391	.246	.233	.082
sulfur	14.7	11.0	27.6	23.5	20.9	20.1	11.5	9.30	25.6	25.3	15.1
thallium	<.026	<.026	<.026	<.026	<.026	<.026	<.026	<.026	.029	<.026	<.026
tin	<.026	<.026	<.026	<.026	.066	<.026	<.026	<.026	.116	<.026	.062
vanadium	.006	<.003	.031	<.003	<.003	<.003	<.003	<.003	.005	<.003	<.003
zinc	.009	.006	.010	.008	.088	.011	.014	.010	.038	.007	.008

1 Three analyses except for Sb, Be, Cd, Co, Au, Pb, Se, Ag, S, and Sn which have two analyses.
2. Two analyses.
3. Three analyses except for Sb, Be, Cd, Co, Au, Pb, Se, Ag, S, and Sn which have two analyses.

Table 2. Maximum and minimum concentrations for non-metals (mg/L).

Constituent	Background springs[1]		Wells in closed-out ash pond[2]		Springs down-gradient of closed-out ash pond[3]		Springs down-gradient of active pond[4]		Surface water samples of active pond[5]		Lake water
	Max	Min	Max	Min	Max	Min	Max	Min	Max	Min	1 value
pH	8.08	7.47	7.18	6.55	7.61	7.34	7.42	6.93	7.79	5.92	6.94
alkalinity as $CaCO_3$	249	225	208	122	216	122	221	210	10	8	39
bicarbonate	304	274	254	149	263	149	270	256	12	10	48
total dissolved solids	314	258	294	146	310	224	290	274	202	128	128
chloride	2.85	2.31	2.10	1.43	4.03	3.03	2.84	2.18	3.99	2.76	2.21
fluoride	.15	.10	.37	.16	.20	.17	.17	.14	.20	.10	.06
sulfate	32.6	21.6	22.8	15.4	95.6	19.8	33.7	20.9	116	80.7	45.4

1 Five analyses except for alkalinity, bicarbonate, and TDS which have four analyses.
2 Five analyses.
3 Nine analyses.
4 Five analyses.
5 Six analyses except for alkalinity, bicarbonate, and TDS which have three analyses.

Table 3 contains a partial summary of the ash-leaching results using deionized water. After 15 minutes of contact time, high concentrations of some elements were present in the leachate. Leaching continued for most elements after one week of contact time. In general, the most enriched leachate resulted from the fine-grained fly-ash fraction.

Table 3. Concentrations (mg/L) of dry ash leachate leached with deionized water at different solid solution ratios for different time increments.

Constituent (mg/L)	1:1 Fly ash; 15 min.	1:1 Bottom ash; 15 min.	1:10 Fly ash; 15 min.	1:10 Bottom ash;15 min.	1:1 Fly ash; 1 week	1:1 Bottom ash; 1 week	1:10 Fly ash; 1 week	1:10 Bottom ash; 1 week
aluminum	700	239	65	20.3	619	287	81	69.2
arsenic	40	7.9	8.0	.86	19	<.050	2.7	<.050
barium	.84	.240	.262	.454	.08	.15	.072	.033
boron	62	14.1	5.5	1.38	66	21	7.82	2.34
cadmium	.18	<.010	<.010	<.010	.24	.12	.023	.014
chromium	14.2	2.86	1.4	.22	1.2	.3	.139	.020
chloride	1.9	8.07	.89	1.53	2.8	22.37	.10	3.50
cobalt	1.6	.44	.16	.04	1.7	.6	.206	.088

Table 3 continued.

Constituent (mg/L)	1:1 Fly ash; 15 min.	1:1 Bottom ash; 15 min.	1:10 Fly ash; 15 min.	1:10 Bottom ash;15 min.	1:1 Fly ash; 1 week	1:1 Bottom ash; 1 week	1:10 Fly ash; 1 week	1:10 Bottom ash; 1 week
copper	29.1	6.44	2.81	.718	23.5	4.54	3.02	.602
fluoride	12.61	4.61	2.33	.54	8.80	6.39	3.89	1.18
iron	19.6	25.5	2.53	2.32	5.69	6.48	1.38	3.01
lithium	30.5	7.1	2.6	1.04	24.2	10.2	3.0	1.11
manganese	4.06	4.11	.379	.308	4.77	----	.540	2.18
nickel	4.65	1.27	.434	.115	4.9	2.10	.597	.253
selenium	.78	<.010	.39	.018	----	----	----	----
strontium	20.4	7.69	2.77	.913	16.4	7.19	4.40	1.78
vanadium	33.6	1.10	3.34	.15	.72	----	.219	----
zinc	7.86	2.15	.761	.186	7.95	2.90	1.00	.324

Summary

Dry-ash samples collected from the site indicate that coal ash generated at the plant is enriched in most elements tested. Dry-ash leach tests show that significant leaching of the ash particles occurs after a 15-minute contact time in deionized water and that continued leaching occurs. The system of ash disposal at the plant which incorporates large volumes of water with a 9 to 15 minute transit time, effectively leaches many trace elements from the surface of the ash particles during sluicing.

Dye traces in the vicinity of the closed-out slurry pond define a flow system which is approximately 2,000 feet wide and 4,000 feet long. The closed-out pond lies within the boundary of the flow system. Anecdotal evidence suggests that spring flow in this system was much greater when it was an active slurry pond. Numerous sinkholes reportedly opened while water was impounded, allowing water to enter the ground-water system. Ash was reportedly observed at some spring outlets when the pond was active.

Major-ion chemistry of the springs connected to the closed-out pond is very similar to background spring chemistry (Table 4). Comparison of mean values for background and down-gradient springs for trace elements associated with coal ash shows that leaching of the ash has already occurred (Table 5). On the average, constituent values for springs below the closed-out pond are similar to background values. Only boron and iron are more than two times greater than background for these springs.

Wells completed directly in the ash show that boron and manganese exceed background by more than one order of magnitude. Barium and fluoride are two to three times

Table 4. Mean concentrations of major ions (mg/L).

Constituent	Back-ground springs	Lake water	Wells in closed-out ash pond	Springs down-gradient of closed-out ash pond	Springs down-gradient of active pond	Surface water samples of active pond
bicarbonate	286	48	196	264	173	11
sulfate	27.8	45.4	18.4	28.5	80.14	95.9
sodium	4.00	12.2	10.8	4.31	10.93	9.49
potassium	<1.32	<1.32	5.73	1.92	2.76	4.38
magnesium	6.97	7.40	7.65	6.07	7.81	8.13
calcium	99.4	16.9	61.8	82.6	72.8	27.4

Table 5. Mean concentrations of trace constituents (mg/L).

Constituent	Back-ground springs	Lake water 1 value	Wells in closed-out ash pond	Springs down-gradient of closed-out ash pond	Springs down-gradient of active pond	Surface water samples of active pond
aluminum	.079	.086	.044	.108	.110	.035
antimony	<.017	<.017	<.017	.019	<.017	<.017
arsenic	<.028	<.028	.049	.029	<.028	<.028
barium	.030	.021	.104	.036	.044	.203
boron	.015	.011	.859	.071	.175	.168
cadmium	<.003	<.003	<.003	<.003	.003	.005
chromium	.008	<.006	.006	.010	.009	<.006
chloride	2.60	2.21	1.66	2.44	3.53	3.43
cobalt	<.004	.004	.004	<.004	<.004	.005
copper	.006	<.004	.005	.007	.010	.031
fluoride	.13	.06	.28	.16	.19	.15
iron	.057	.008	.039	.176	.150	.204
lead	<.038	<.038	<.038	<.038	<.038	<.038
lithium	<.006	<.006	.114	.012	.026	.128
manganese	.005	.006	.407	.003	.028	.071
nickel	<.012	<.012	<.012	.013	<.012	.024
selenium	<.052	<.052	<.052	<.052	<.052	<.052
strontium	.276	.082	2.00	.398	.399	.237
sulfur	12.9	15.1	25.6	10.4	20.5	25.5
vanadium	<.003	<.003	.017	<.003	<.003	.003
zinc	.008	.008	.009	.012	.050	.024

background levels. Vanadium and strontium values are approximately six times greater than background concentrations. Other constituents are similar to background measurements.

Dye traces define a wedge-shaped flow system which encompasses the active pond and is at least 2,900 feet wide and 6,200 feet long. Total ground-water spring flow in this system is approximately two orders of magnitude greater than spring flow observed in the closed-out pond system. The source of much of this flow is apparently the active disposal pond. Water from the pond enters the ground-water system through fractures and sinkholes.

Comparison of water quality from springs and the disposal pond with background quality shows that leaching has occurred in this system. Most trace constituents are similar to background spring and lake water quality for the elements listed (Table 5).

Barium, boron, iron, and manganese are more than 10 times greater than raw lake water in the surface-water samples taken from the active pond. Copper and zinc are eight and three times greater, respectively. The remaining constituents are similar to raw lake water concentrations. Water quality in the active pond reflects the major-ion composition of the raw lake water used for sluicing (Table 4). The pH, alkalinity, bicarbonate, total dissolved solids, and calcium are depressed compared to background springs.

Boron values in the springs below the active pond are approximately 10 times greater than background spring measurements. Manganese and zinc concentrations are approximately six times the background values in springs associated with the active pond. Iron is three to four times greater than background spring values. Major-ion composition for these springs is intermediate between background and active pond water.

Conclusions

1. The ground-water flow system in which the disposal sites occur have been defined relatively well. Even though deep wells have not been installed at the site, the base of the flow system can be placed near the contact between the St. Louis Limestone and Salem and Warsaw Formations.

2. Total dissolved solids are relatively low with the maximum value occurring in the background springs.

3. Mean values for all constituents analyzed are within U.S. EPA standards, even though some elements are 10 times background spring concentrations.

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Biographical Sketch

Shelley A. Minns obtained the B.S. degree in geology from West Virginia University in 1977. She worked for 10 years throughout the southern Appalachian coal field region, primarily for coal-related consulting firms. She obtained the M.S. degree in environmental studies from Ohio University in 1989 and is currently a Ph.D. candidate in geology, specializing in hydrogeology, at the University of Kentucky, Lexington, Kentucky. Her research interests are coal-ash disposal and coal hydrology. She is currently employed with the Kentucky Geological Survey as a research assistant.

Hydrogeology and Ground-Water Monitoring of an Ash-Disposal Site at a Coal-Fired Power Plant in a Karst System

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Question 1: Did you define two flow systems or two drainage basins?

The two disposal ponds are located in different surface water drainage basins. Dye traces were conducted on all available sinkholes considered upgradient of each pond. The dye traces defined the flow systems and as it turned out, they also reflected the surface water basins.

Question 2: What was the rationale for selecting 5 times background concentration as the criterion for determining whether the levels were elevated above background?

Concentrations of metals for water associated with the two disposal sites range from nearly identical to background spring concentrations to about two orders of magnitude greater than background concentrations. The ranges of five times greater than background and ten times greater than background were selected to use during the presentation of the paper at the conference as a way to get a feel for how metals were distributed when compared to background spring concentrations. These numbers to do not imply any significance relative to background concentrations and are not used in the paper. The paper contains detailed data of constituents in the water associated with the two disposal sites.

Question 3: What leaching tests were found to be most representative of actual processes presumably operative within the ponds? Was there a significant difference in the trace element nature of the coals burned at the site between the old and the active ponds? Were there other significant waste streams contributing to the ponds?

Leaching tests were not conducted as part of our research effort. Battelle-Pacific Northwest Laboratory conducted the leaching tests as part of a separate investigation. We have not received the complete results of these tests. We do know that the solid/solution ratio, which is generally representative for the sluicing system used at the plant, is ten parts ash to one part lake water.

Trace element analyses of coal burned at the plant when the closed-out pond was an active facility are not reported.

No other waste streams impact the disposal ponds. The contributing watershed is mainly undisturbed forest. The coal stockpile area is located downgradient of both ponds and does not affect the ash disposal site.

DEVELOPMENT OF A MONITORING PROGRAM AT A SUPERFUND SITE IN A KARST TERRANE NEAR BLOOMINGTON, INDIANA

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Noel C. Krothe, Indiana University
Bloomington, Indiana

Abstract

Lemon Lane, a former municipal dump from 1933 to 1964, is located entirely within two sinkholes on a ridgetop over Mississippian limestones. Materials containing PCBs were discarded there from 1957 to 1964.

Twenty-two monitoring wells have been installed in stages into two separate ground-water zones. Results from three tracer tests conducted from 1987 through 1990, and interpretations of limited potentiometric data, indicate that the site is drained by a convergent conduit system that, during low to moderate flow, drains through a main conduit which exits at a series of springs. A quantitative tracer test showed that 97% of the tracer recovered flowed to one spring. During high flows, when the conduit is flooded, some water is temporarily stored in the epikarst and the fractured bedrock adjacent to the conduit. Small amounts of water in storage may spillover into adjacent spring drainage basins, suggesting the site is near the ground-water drainage divides of several subterranean basins.

Fluorescein injected as a tracer in an on-site well, in 1989, was still detectable at the main spring outlets 19 months later. Monitoring at selected on-site wells has shown tracers at levels near the detection limit in water (0.005 ppb) of a scanning spectrofluorophotometer.

The springs, indicated by tracing to be directly connected to the site, will provide the most time-sensitive and relevant monitoring locations for potential off-site migration of contaminants. Existing monitoring wells should effectively monitor for contaminants in temporary storage. As a result, the installation of additional monitoring wells is not recommended because the probability of any of them intersecting conduits draining the site is minimal.

Description of Lemon Lane Landfill

Lemon Lane Landfill is located on the west-central edge of Bloomington in Monroe County, Indiana. Bloomington is located in the Mitchell Plain physiographic unit. The Mitchell Plain is a low plateau underlain with carbonates of Mississippian age on which a characteristic karst terrane has developed. The Lemon Lane landfill, comprising about 9 acres, was located entirely within two sinkholes on a ridgetop at or near several surface water or topographic divides. However, all of the natural surface drainage at the site is into several sinkholes. The 1908 topographic map for the area (Figure 1), shows the two sinkholes that were used for the disposal of solid waste, which are actually part of a large compound sink extending to the northwest and southwest of Lemon Lane.

Lemon Lane was operated as a municipal dump from 1933 to 1964. A nearby plant manufactured capacitors containing PCBs as an insulating fluid beginning in 1957. Materials containing PCBs were discarded at Lemon Lane from 1957 to 1964 by local contract waste haulers.

Geology of Lemon Lane Landfill

The site is underlain by Mississippian age limestones, dolostones, and shales of the St. Louis Formation. The Salem Limestone, which has strata similar to the St. Louis in its upper part underlies the St. Louis Formation. The lower part of the Salem is a thick, cross-bedded calcarenite which is famous as a building stone known as the "Indiana Limestone" (Shaver et al., 1986). The contact between the St. Louis and the Salem is not well defined lithologically in south-central Indiana.

The strata generally dip westward from their outcrop on the flank of the Cincinnati Arch into the Illinois Basin. The dip averages about 30 feet to the mile in the Bloomington area (Gates, 1962).

Monitoring Well Installation

Twenty-two monitoring wells were installed in two phases at Lemon Lane. Seventeen wells were installed, from 1982 to 1983, in the first phase and five additional wells were installed in 1987 (Figure 2). Seventeen of the wells were installed to monitor ground water in shallow bedrock and five of the wells monitor a deeper zone. Figure 3 shows the stratigraphy encountered. The shallow zone occurs between elevations 798 ft. and 820 ft. amsl and the deeper zone occurs between elevations 760 ft. and 770 ft. amsl. Monitoring wells MW-4D, MW-5, MW-8D, MW-12, and MW-14 are installed in the deeper zone and the rest are installed in the shallow zone.

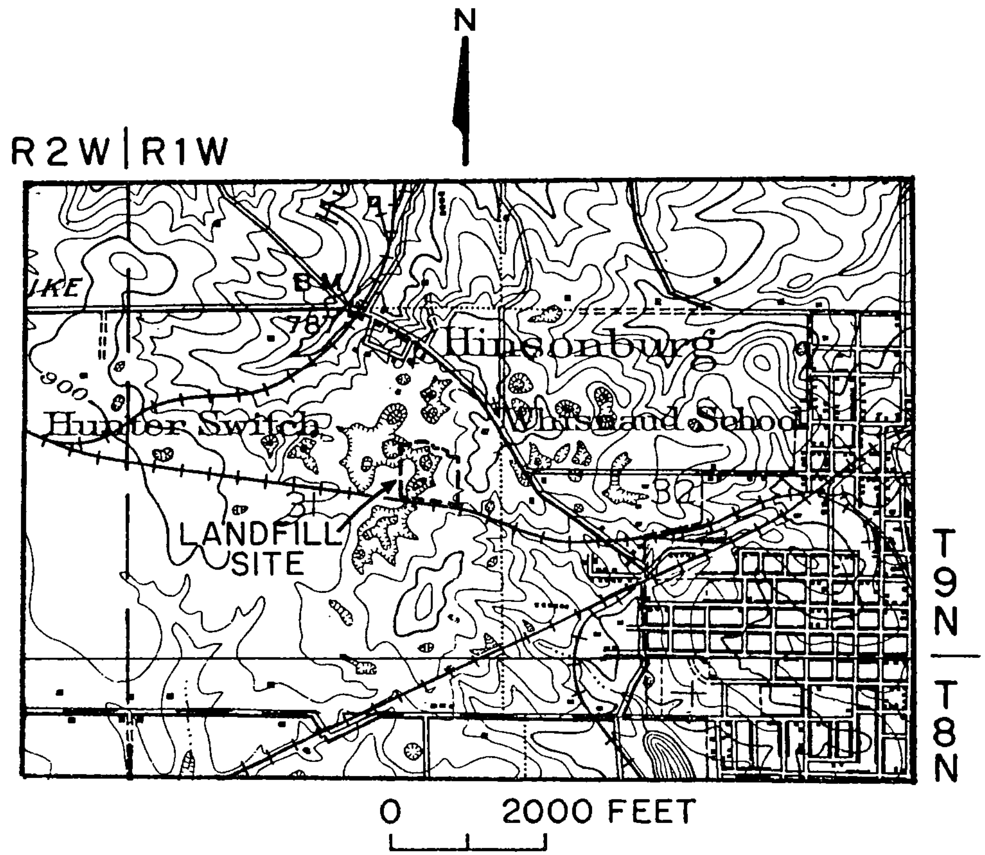

Figure 1. Location of Lemon Lane Landfill relative to sinkholes prior to use as a landfill. (Bloomington, Indiana, Quadrangle 15 minute series, 1908)

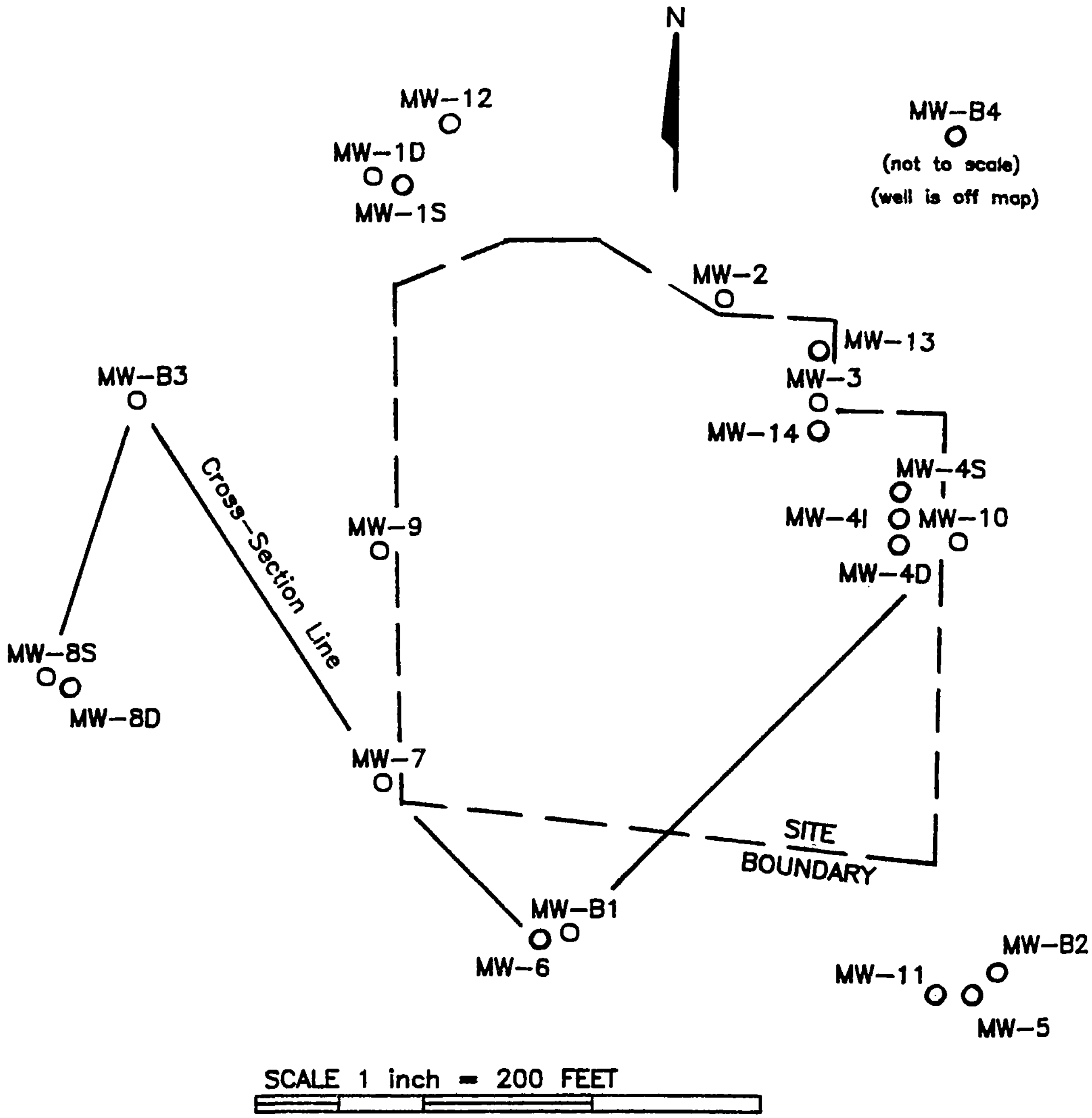

Figure 2 - Monitoring Well and Cross-Section Location

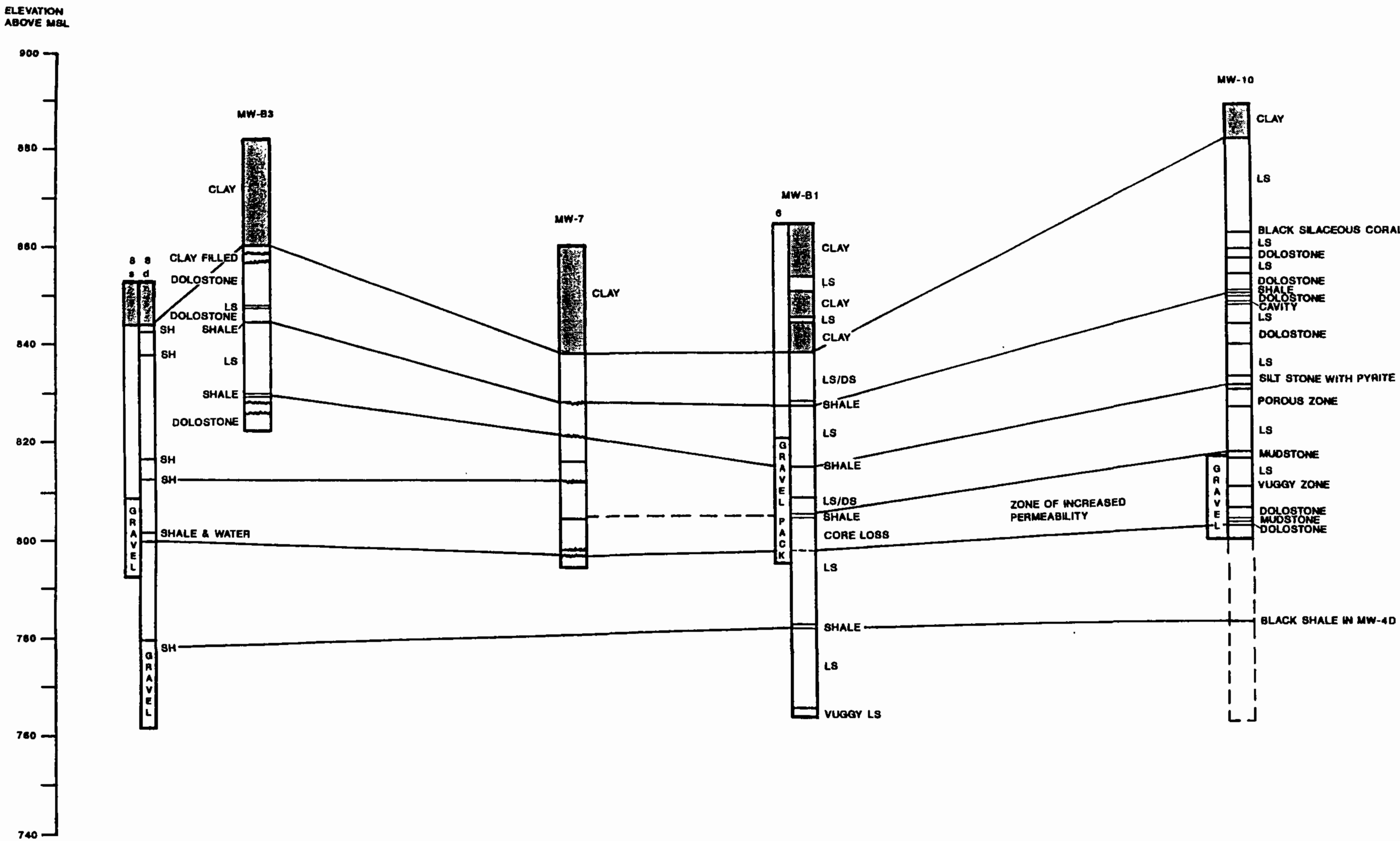

Figure 3 - Geologic Cross-Section

The two zones appear to be separated by a shale unit that is at elevation 782 ft. to the east and 778 ft. to the west (see Cross-section, Figure 3).

Packer tests and slug tests conducted during installation indicated the wells in the deeper zone encountered fractures much less transmissive than those encountered by the wells in the shallow zone.

Water Level and Water Quality Measurements

Potentiometric surface maps for water levels taken on 2/2/88 and 6/14/88 are shown for the upper and lower zones in Figures 4,5,6,and 7. The interpretation of ground water flow directions in the lower zone appears to be relatively simple. During the higher flow period (2/2/88) recharge is from the west and flow directions are to the east. During the lower flow period (6/14/88) recharge is from the west and east and flows are generally to the north and northwest. Interpretations of flow directions in the upper zone, however, are problematic. During the higher-flow period, ground water appears to be mounded in the vicinity of wells MW-7, MW-B3, MW-4I, MW-8S, and MW-11. Water levels are depressed in the vicinity of MW-9, giving the appearance of a sink. During the low-flow period, mounding is apparent near MW-B3, MW-3, and MW-11. In neither flow regime is there an unambiguous flow direction indicating the probable direction of ground-water movement.

Table 1 shows the analytical results of PCB sampling at the 17 monitoring wells installed in 1982 and some springs in the vicinity of Lemon Lane. Wells MW-10, MW-11, MW-12, MW-13, and MW-14 were installed in 1987 and have not yet been sampled. The results are difficult to interpret. For example, although PCBs were found in wells MW-8S and MW-8D, the potentiometric maps for both flow regimes seem to indicate these wells are upgradient. Illinois Central and Quarry springs, located about 2000 feet southeast of Lemon Lane, have concentrations of PCBs one to two orders of magnitude higher than the on-site wells.

The original monitoring plan for Lemon Lane required on-site monitoring wells to be selected on the basis of ground-water flow directions and the results of initial PCB sampling. Locations for off-site wells were to be selected on the basis of preferred flow directions as interpreted from on-site wells. The original monitoring plan was based on the assumption that an aquifer was present that behaved as a porous-media-equivalent and that ground-water flow direction could be predicted by idealized flow nets constructed from water level measurements. Contaminant movement could thus be predicted from aquifer properties deduced from pumping tests which were analyzed by standard equations of ground-water flow. In an attempt to resolve the ambiguities, tracer tests were conducted.

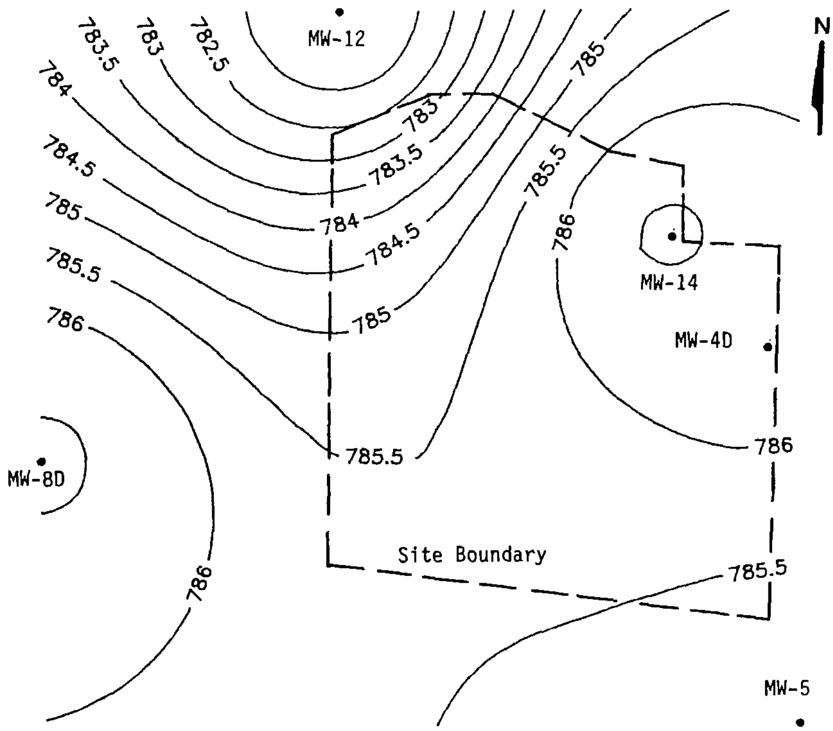

Figure 4 - Lower Zone Potentiometric Map 6/14/88

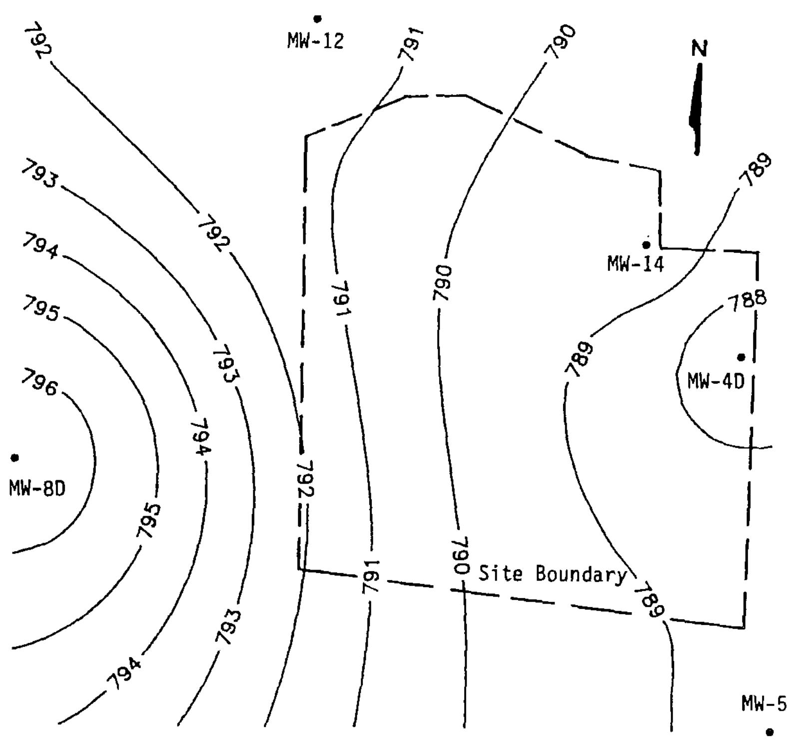

Figure 5 - Lower Zone Potentiometric Map 2/2/88

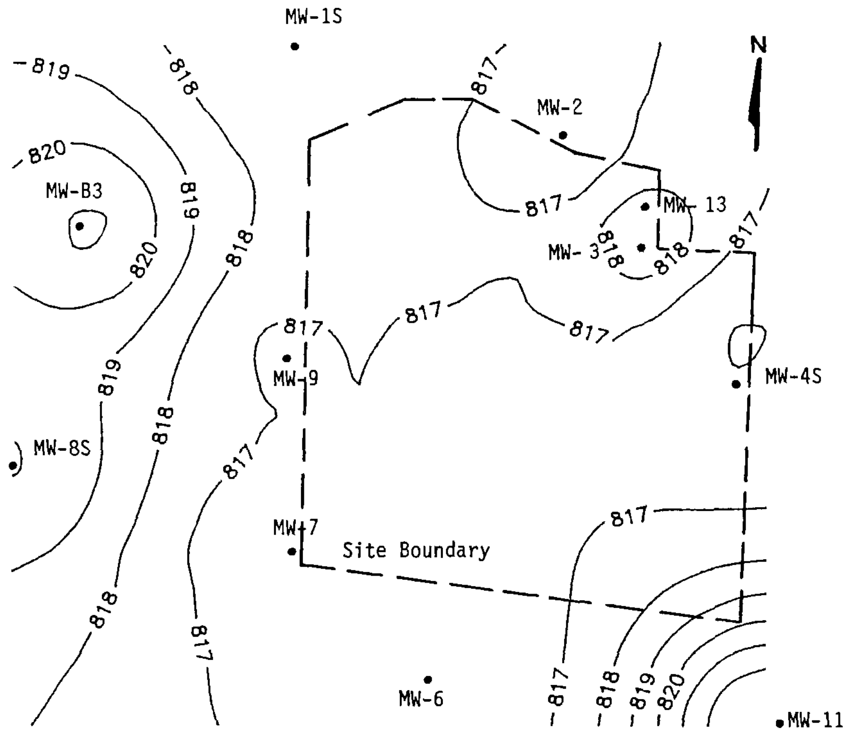

Figure 6 - Upper Zone Potentiometric Map 6/14/88

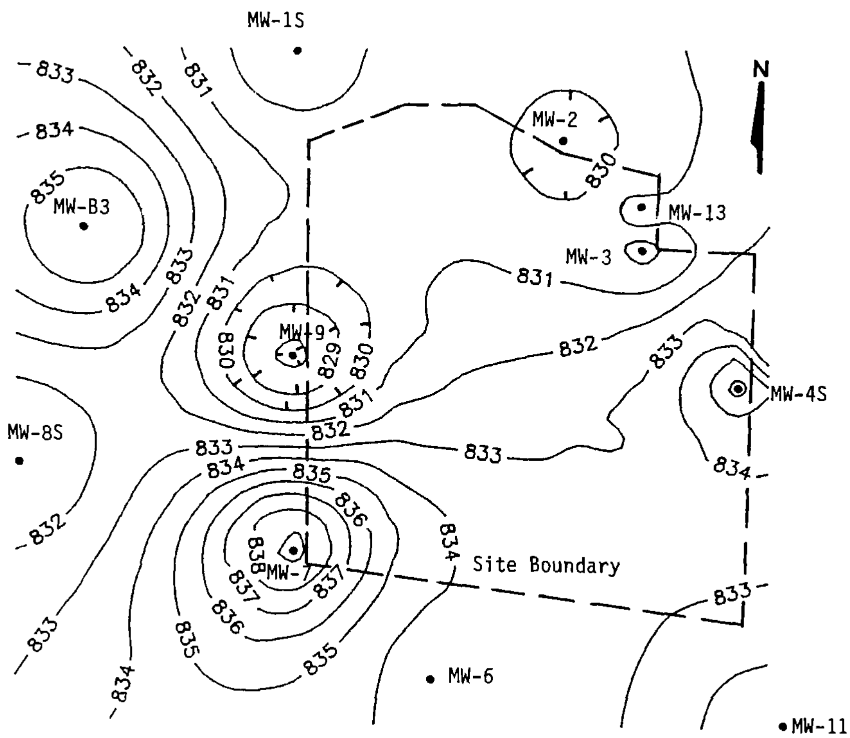

Figure 7 - Upper Zone Potentiometric Map 2/2/88

Table 1

PCB ANALYTICAL DATA FOR GROUND WATER AND SURFACE WATER
LEMON LANE LANDFILL
BLOOMINGTON, INDIANA

Location		6/25-29/81	7/1-2/81	8/17/81	10/7-8/81	7/28-29/82	10/19/82	12/9,15/82
Springs/Surface Water								
Sargent Pond #1	ND	0.9*						
Sargent Pond #2	ND						ND	
Stout Creek		ND	ND					
Robertson Spring				ND				ND
Snoddy Spring				ND				ND
Stoney Spring-East				ND	ND		1.85**	ND
Stoney Spring-West			ND	ND	ND		ND	ND
P.H. Road Spring				ND			ND	
Detmer Spring				ND			ND	
Packing Plant Spring							ND(ND)	
Ill. Central Spring							12.2**	10.**
Quarry Spring			5.7***	6.8*			2.7***	5.5***
Slaughter House Spring			ND			ND		
Hinkle Wet Weather Rise							ND	

Location	10/19/82	12/17/82-1/5/83	2/2/83-2/14/83	6/8/83
Monitoring Wells				
MW-B1	ND			ND
MW-B2	ND			ND
MW-B3	ND			ND
MW-B4	ND			ND
MW-1S		0.4	0.11	
MW-1D		0.3	0.13(.05 dup)	
MW-2		<0.01	0.02*	
MW-3		0.5	0.04*	
MW-4S		0.8	0.07	
MW-4I			1.5*	
MW-4D		2.4	0.09	
MW-5		0.2	0.02*	
MW-6		<0.01	0.02(.05 dup)	
MW-7		0.6	0.10	
MW-8S		0.6	0.11*	
MW-8D		1.4	0.07	
MW-9		1.1	0.40*	

NOTES: All analyses for mixed Aroclors unless noted by * or **
All analyses in ug/l or parts per billion (ppb)
* Aroclor 1016/1242 *** Arcolor 1242
** Aroclor 1248
<# - Not detected, with respect to analytical detection level (#).
ND - Not detected, no detection level provided with analytical results.

1987 Low Flow Tracer Test

The first tracer test at Lemon Lane was conducted under dry, fall conditions. Injection occurred on November 10, 1987 and sampling continued until November 27, 1987. Monitoring wells MW-7, MW-1D, and MW-10 were selected as injection locations. Springs to be sampled were: Crestmont A and B, Detmer A, Illinois Central, Quarry, Packinghouse Road (PH Road), Pumping Station, Snoddy A and B, Stony East, Stony West A and B, and Urban (Figure 8). Monitoring wells to be sampled were: MW-5, MW-11, MW-8S, and MW-8D. Lithium bromide (Br-) was chosen as the tracing agent and two sets of background samples were taken prior to injection. Background concentrations of Br- were in the range of 0.01 to 0.39 ppm with a detection limit of 0.01 ppm.

Sampling frequency was every 6 hours for the first 72 hours and then every 24 hours for the remainder of the sampling period. Only Illinois Central Spring and Quarry Springs had definitive breakthrough curves for Br- that had a classic rising limb, a peak, and a receding tail. The peak at Illinois Central was 5.11 ppm and the peak at Quarry was 4.83 ppm. The time to peak from first detection at both springs was about 250 hours. Illinois Central rises and flows along the surface for 600 feet to where it sinks about 200 feet above where Quarry Springs resurges. Based on this and the tracer detection times, Quarry Springs was presumed to be second resurgence of the same water. This was later confirmed by a dye trace run from where the Illinois Central waters sink to their outlet at Quarry Springs. Crestmont A had one Br- sample within 15 hours of injection that was 0.50 ppm and Urban spring had a similar one time occurrence of Br- at 0.66 ppm within 1.5 hours after injection. Well MW-8D had 1.49 ppm Br- detected in it on November 25, 1987. It is possible that these Br- detections may be the tracer injected at Lemon Lane, but it is not considered likely because they are isolated occurrences.

1989 High Flow Tracer Test

A tracer test during high flow periods was scheduled to be performed in the spring of 1988. However, drought conditions prevailed and the test was rescheduled for the spring of 1989. A different fluorescent dye was injected in each well along with an aliquot of Br- in each well. Color Index (C.I.) Fluorescent Brightner 28 (FB28) was injected in MW-7. MW-10 received Fluorescein (C.I. Acid Yellow 73; AY73). Instead of MW-1D, well MW-1S was injected with Direct Yellow 96 (DY96) (Figure 2). Tracers were distributed throughout the bedrock column in these wells because injection occurred during rising water levels and because there was interconnection between nested wells. The trace was to be qualitative with detection to be accomplished visually for the fluorescent dyes. FB28 and DY96 are adsorbed onto sterile cotton fabric which is suspended in the resurgence. The cotton is washed and then observed under a hand-held ultraviolet lamp. FB28 will fluoresce blue, and DY96 will fluoresce yellow. If both

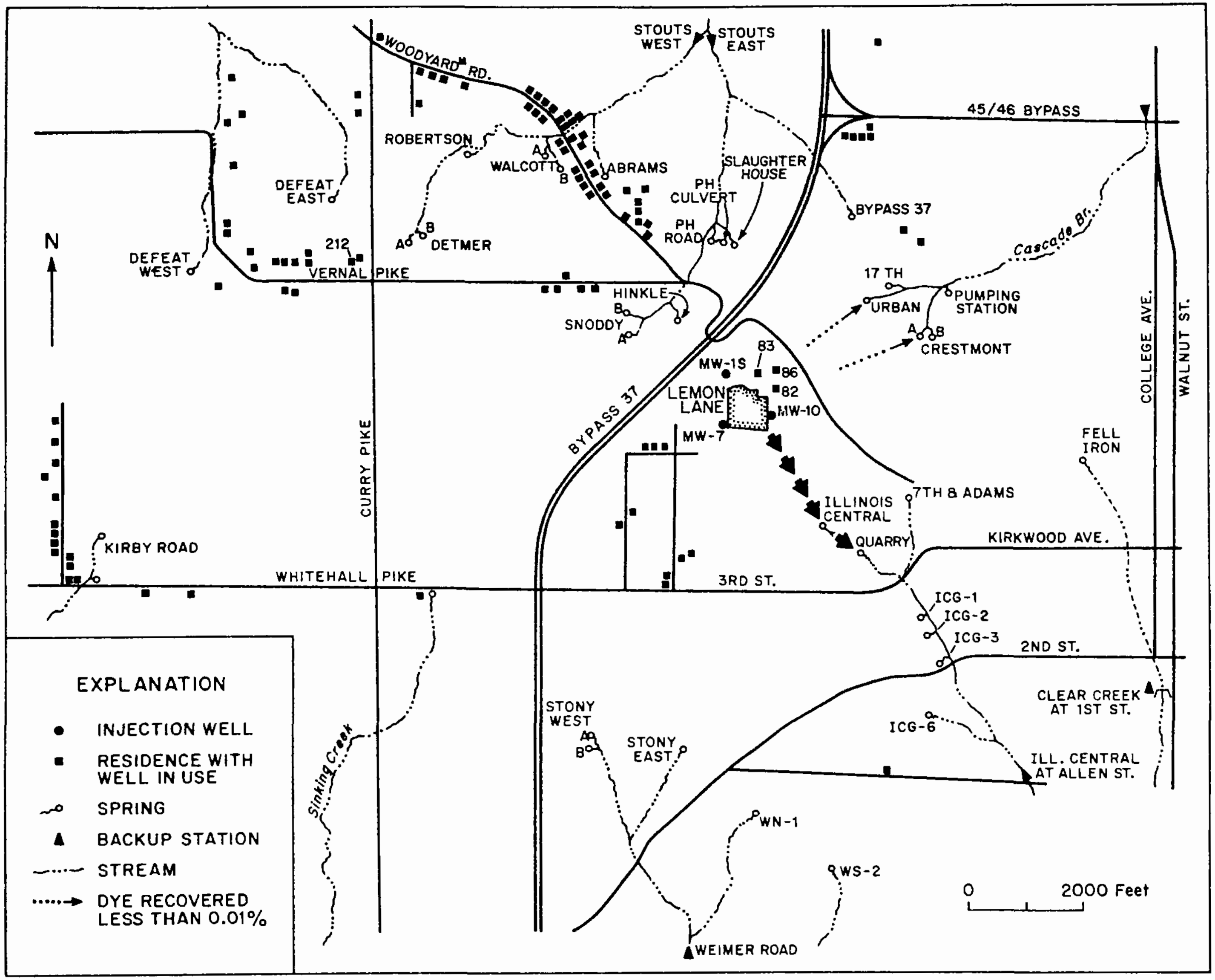

Figure 8 - 1987 Low Flow Tracer Test

tracers are present, the cotton will fluoresce a characteristic blue-white. AY73 is adsorbed onto activated charcoal which is suspended in screen packets in the resurgences. The dye is eluted from the charcoal with a hydroxide solution and will appear as a yellow-green layer in the elutant above the charcoal. Background samples taken before injection indicated minor amounts of FB28 at some stations. This is expected since FB28 is a component of laundry detergents. However, no background samples were taken during storm flows.

Injection of tracers took place on May 26, 1989 after a 1.35 inch rainfall. Several storms of 1" or greater rainfall occurred during sampling, including a 2.36" rain on July 12, 1989. Figure 9 shows the location of springs and streams that were sampled during the test until July 22, 1989. Monitoring wells that were sampled were MW-8S, MW-6, MW-5, and MW-11. Sampling began within 6 hours after injection, and Illinois Central and Quarry springs had distinct positives for FB28 on the first samples. Breakthrough curves for Br- were also detected at Illinois Central and Quarry springs with peak concentrations at Illinois Central of 8.7 ppm and 10.5 ppm for Quarry. Distinct visual positives for AY73 were detected at Illinois Central and Quarry springs beginning 55 hours after injection and continued throughout the 57 day sampling period. The tracer DY96 was not recovered visually at any station during the sampling period. Apparent detections of AY73 were recorded for all stations on samples collected following rain storms. However, the coloration was not the distinctive AY73 yellow-green and it was suspected other organic constituents were being adsorbed by the charcoal and making the definite identification of AY73 impossible by visual means. There was no visual detection of dye at any of the monitoring wells, although Br- was detected at MW-8D. AY73 was detected visually at residential well #83 (Figure 9). There were Br- detections at Clear Creek of 2.96 ppm at 839 hrs (35 days) and 1.45 ppm at 1147 hrs (47 days) after injection, but since they are isolated occurrences they are not considered conclusive.

Since the visual detection of the dyes at many stations was inconclusive, random samples were submitted to Quinlan and Associates for analysis on a scanning spectrofluorophotometer (SSFP). These analyses indicated AY73 and DY96 were present in some samples, even though they were not recognizable visually. A SSFP was purchased in February 1990, and most of the grab water samples taken for Br- analysis were analyzed for fluorescence. Table 2 shows the results of that analysis. These results confirmed the suspicion that Lemon Lane was at or near the divides of several subterranean drainage basins, and, that upper level conduits exist that would divert ground water, during high flow storm events, into those adjacent spring drainage basins. Actual dye concentrations were not calculated for these samples, but the relative intensity of fluorescence as measured by the SSFP indicated that Illinois Central and Quarry Springs received the majority of the tracers and all the other stations received only minor amounts. It was obvious that the proportional distribution

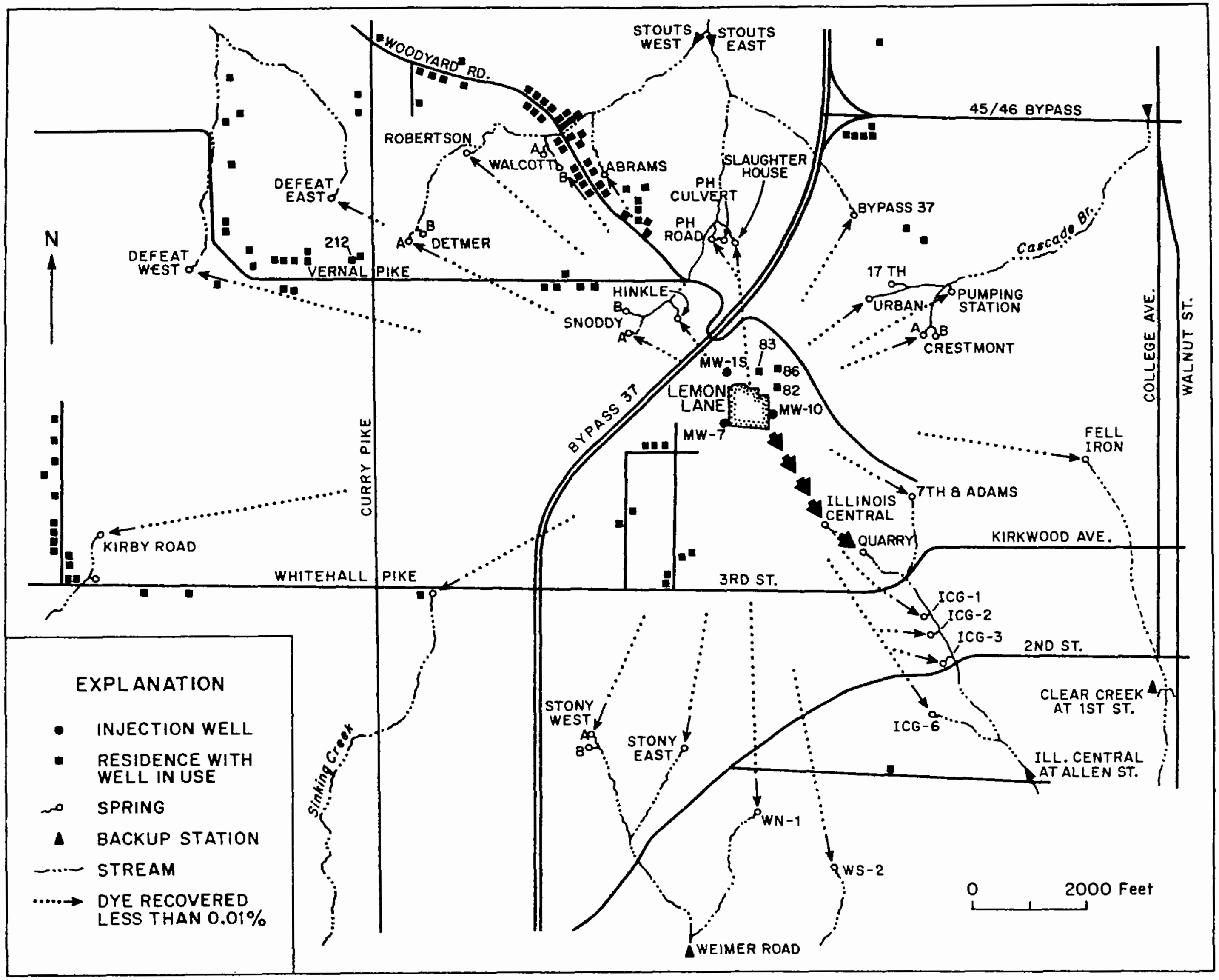

Figure 9 - 1989 High Flow Tracer Test

Table 2 - Results of Spectrofluorophotometer Analysis
1989 High Flow Tracer Test

Station	5/26	5/27 0600-0800	5/27 1800-2000	5/28 0000-0200	5/28 0600-0800	5/28 1200-1400	5/28 1800-2000	5/29	5/30	5/31	6/1	6/2	6/9	6/16	6/23	6/30	7/7	7/13	7/22
Detmer A	ND	ND	DY96	DY96	DY96	ND	DY96	ND	ND	ND	ND	DY96	DY96	ND	ND	DY96	DY96	DY96	ND
Detmer B	FB28	FB28	DY96	ND	ND	ND	DY96	ND	ND	DY96	DY96	ND	DY96	DY96	--	--	DY96	DY96	ND
Defeat E.	DY96	DY96	--	DY96	DY96	DY96	ND	DY96	DY96	DY96	DY96	DY96	--	DY96	ND	DY96	DY96	DY96	ND
Defeat W.	FB28	DY96	ND	DY96	ND	--	ND	DY96	DY96/ AY73	ND	ND	DY96	ND	--	--	--	ND	ND	ND
Kirby Rd.	ND	DY96	ND	ND	FB28	DY96	DY96	DY96	DY96	DY96	DY96	ND	DY96	ND	DY96	ND	DY96	ND	DY96
Stony E.	ND	DY96	--	FB28	ND	ND	ND	ND	ND	AY73	AY73	DY96	ND	ND	ND	ND	--	--	--
Stony W.	--	DY96	ND	ND	DY96	DY96	DY96	ND	ND	AY73	ND	AY73	DY96	ND	DY96	DY96	--	--	--
Sinking Cr.	FB28/ AY73	--	FB28/ AY73	AY73	AY73	AY73	AY73	DY96/ AY73	FB28/ AY73	DY96/ AY73	AY73	AY73	ND	DY96/ AY73	DY96	--	DY96	AY73	AY73
WN-1	AY73	AY73	ND	AY73	ND	ND	ND	ND	ND	ND	ND	ND	ND	--	--	--	--	--	--
WS-2	--	ND	ND	ND	--	ND	ND	ND	ND	ND	--	--	--	ND	--	--	AY73	ND	--
ICG-1	FB28/ DY96	ND	ND	DY96	ND	--	AY73	AY73	AY73	DY96/ AY73	AY73	AY73	DY96/ AY73	AY73	AY73	AY73	--	--	--
ICG-2	ND	--	FB28/ AY73	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
ICG-3	ND	--	AY73	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
ICG-6	FB28	FB28	ND	ND	ND	ND	ND	ND	ND	ND	--	AY73	ND	ND	ND	AY73	AY73	ND	ND
Pump St.	FB28	FB28	ND	ND	ND	ND	ND	--	ND	ND	ND	ND	ND	ND	ND	ND	DY96	ND	ND
Urban	DY96	DY96	DY96	ND	ND	ND	ND	DY96	ND	--	ND	DY96/ AY73	DY96/ AY73	AY73	AY73	DY96/ AY73	DY96/ AY73	DY96	ND
Bypass 37	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	DY96	ND	ND
S.House	FB28/ AY73	DY96	DY96	ND	AY73	AY73	AY73	AY73	AY73	AY73	AY73	DY96/ AY73	--	--	--	--	--	--	--
PHRoad	FB28	ND	ND	ND	ND	AY73	AY73	AY73	AY73	AY73	AY73	DY96/ AY73	AY73	--	--	--	--	--	--
Hinkle	FB28	ND	ND	DY96	ND	ND	ND	ND	DY96	--	--	DY96	--	--	--	--	--	ND	--
Snoddy A&B	FB28	FB28	ND	ND	FB28	DY96	FB28	ND	ND	DY96	DY96	AY73	ND	ND	ND	ND	DY96	DY96	ND
17th St.	FB28	ND	ND	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--	--
IL Central @ Allen	FB28	DY96	DY96	ND	--	ND	DY96	DY96/ AY73	DY96/ AY73	AY73	AY73	--	DY96/ AY73	--	AY73	AY73	AY73	AY73	AY73
Weimer Rd.	--	ND	ND	DY96/ AY73	AY73	DY96/ AY73	--	ND	DY96/ AY73	ND	AY73	ND	ND	DY96	ND	DY96	ND	ND	ND
Stouts W.	FB28	ND	FB28	ND	ND	ND	ND	ND	ND	ND	ND	ND	DY96	ND	ND	AY73	--	--	--
Stouts E.	DY96	FB28/ AY73	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	DY96	ND	ND	DY96	ND	ND
MW-8S	FB28	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
MW-6	ND	ND	FB28	ND	ND	ND	ND	ND	ND	AY73	ND	ND	FB28/ AY73	ND	ND	ND	ND	ND	ND
MW-11	ND	ND	ND	ND	FB28	ND	ND	FB28	ND	ND	ND	ND	FB28	ND	ND	ND	ND	ND	ND
MW-5	FB28	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

ND - Not Detected
-- - Not sampled or not analysed
FB28 Optical brighteners detected
AY73 Fluorescein detected
DY96 Direct yellow 96 detected

of groundwater would have to be ascertained. A quantitative high-flow tracer test was scheduled for the spring of 1990.

1990 High Flow Tracer Test

Background sampling was conducted at all monitoring stations to ascertain which fluorescent tracer would have the least amount of background interference and be most suitable for the quantitative trace. The sampling indicated that Rhodamine WT, (C.I. Acid Red 388; RWT) would be appropriate. Some stations however, did have background concentrations of RWT in grab samples of water analysed on the SSFP. The detection limit of RWT on the SSFP was 0.01 ppb (Table 3).

RWT was detected at 7th & Adams on 4/19/90 in the elutant from a charcoal packet. In addition AY73 was detected in Quarry Springs at 0.51 ppb. Eighty-two residential wells in the vicinity of Lemon Lane were monitored during the tracer test by teams from federal, state, and local agencies (Figure 10). The one residence, #83, that had AY73 detected in its well during the 1989 test, was subsequently connected to city water and this well was not monitored during 1990. Residential wells #82 and #212 had AY73 detected in them during background monitoring for the 1990 test.

Spring and stream stations were gaged, where possible, and Table 4 lists the stations and the flow monitoring method. Illinois Central spring, per se, was not monitored for this test, rather a gaging station was installed below Quarry Springs.

Table 3 - Background Determination of Rhodamine WT in Water Samples in parts per billion (ppb)

STATION	3/20/90	4/2/90	4/19/90	5/7/90
Quarry Spring	0.075	ND	ND	ND
Urban Spring	0.095	ND	ND	ND
Bypass 37	0.075	ND	ND	ND
Stony East	0.058	ND	ND	ND
Clear Cr @ 1st St.	0.31	0.125	ND	0.11
ICG-2 Spring	0.15	ND	ND	ND
Cascade Br.	0.095	ND	ND	ND
Weimer Rd.	0.058	ND	ND	ND
7th & Adams	ND	ND	ND	ND

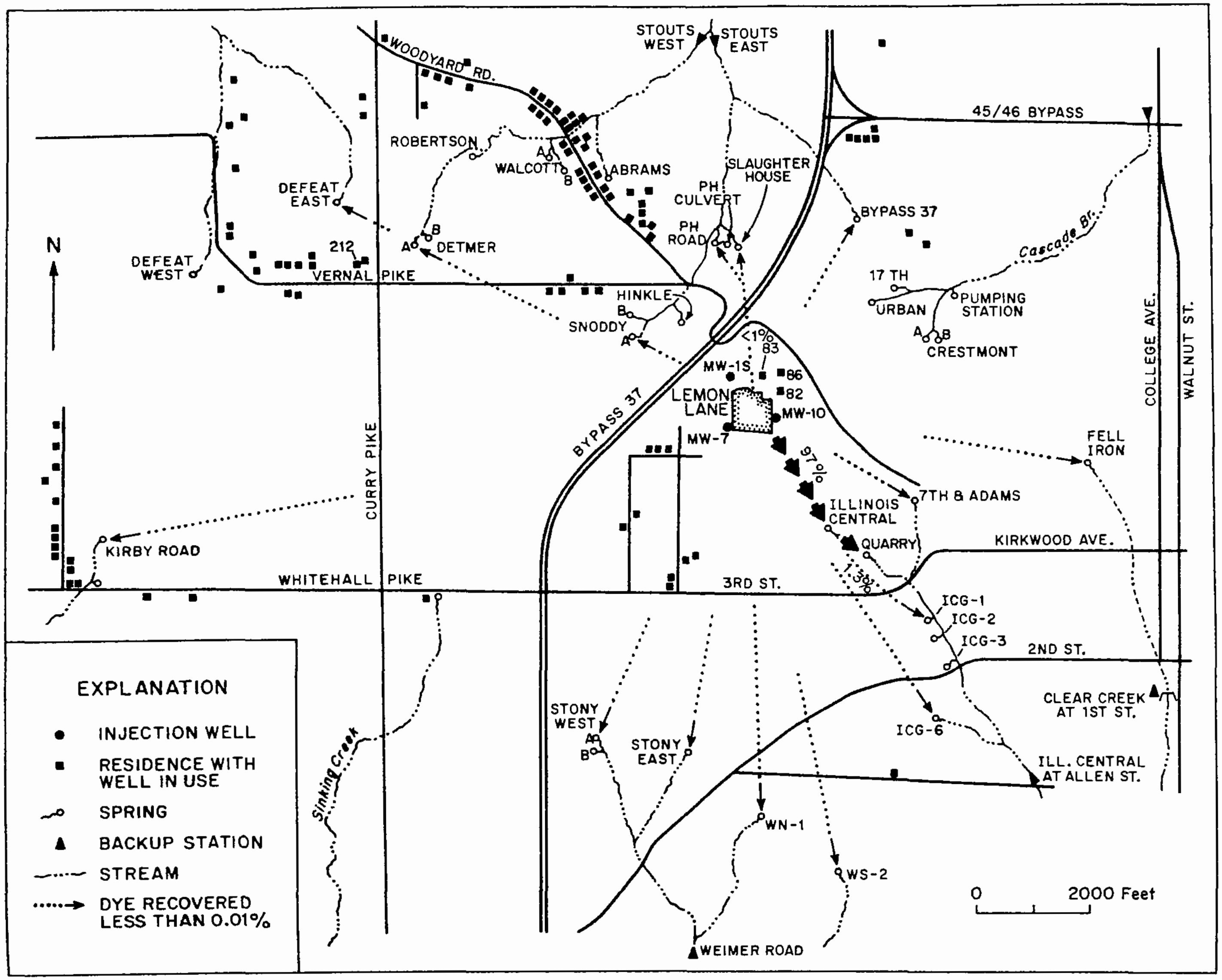

Figure 10 - 1990 High Flow Tracer Test

TABLE 4 - MONITORING STATIONS FOR 1990 TRACER TEST

Station is near a Spring	Station is downstream of a Spring	Station is a backup point downstream of several primary of several Springs
1. Quarry Spring (w)	2. 7th & Adams (rc)	Cascade Br. (rc)
3. ICG-1 (rc)	8. Clear Cr. @ 1st (rc)	Stouts Cr. W. (rc)
4. ICG-2 (w)	13. Bypass 37 (rc)	Stouts Cr. E. (rc)
5. ICG-3 (w)	26. Defeat Cr. E. (rc)	Ill Cen.@ Allen (rc)
6. ICG-6 (w)	27. Defeat Cr. W. (rc)	Weimer Road (nm)
7. Fell Iron (nm)	28. Kirby Road (rc)	Clear Cr. @ 7th (nm)
9. Crestmont (w)	29. Sinking Creek (rc)	
10. Pumping Stn. (nm)		
11. 17th Street (nm)		
12. Urban (w)		
14. S.House (w)		
15. PH Culvert (w)		
16. PH Road (w)		
17. Snoddy A (w)		
18. Snoddy B (w)		
19. Hinkle (w)		
20. Abrams (w)		
21. Walcott A (w)		
22. Walcott B (w)		
23. Robertson (rc)		
24. Detmer A (nm)		
25. Detmer B (w)		
30. Stony West (rc)		
31. Stony East (rc)		
32. WN-1 (w)		
33. WS-2 (w)		

(w) - 90° v - notch weir
(rc) - rating curve with staff gage
(nm) - not measured

Injection of tracers began on May 12, 1990 after a 1.45" rain fell. RWT was injected as a 20% solution and the injected amounts are corrected for this dilution. Well MW-1S received 9,545 grams of RWT, MW-10 received 9,545g, and MW-7 received 8,183g for a total of 27,273 grams of RWT injected. Large amounts of RWT were injected because RWT is known to suffer large adsorption losses and high suspended clay and organic matter loads were observed during storm flows at Quarry Springs.

RWT was first detected at Quarry Springs 3.75 hours after injection and reached a peak concentration of at least 4300 ppb (distinctly visible) at 5.25 hours. By 10 hours after injection the concentration had receded to 40 ppb and remained at or below that level through the last sample taken on August 24, 1990. Five other stations had repeated detections of RWT. They were ICG-1, Slaughterhouse, PH Road, PH Culvert, and Clear Creek at First St. However, Clear Creek did not have a breakthrough curve during the first 72 hours of sampling. Only two samples from Clear Creek were above the background level of 0.31 ppb. They were a 2.6 ppb detection at 584 hours after injection and a 0.7 ppb detection at 593 hours. The Clear Creek watershed is urbanized and the specific spring heads could not be located or monitored. A determination of Clear Creek's connection with Lemon Lane above its confluence with the Illinois Central-Quarry Spring Branch is not conclusive.

Ten stations had sporadic and low (near background level) detections of RWT in the grab water samples. They are listed in Table 5, along with the date, time, concentration detected, and amount detected in background samples. In addition the following stations on the following dates had RWT detected in the back-up activated charcoal samples: Fell Iron Spring on 6/8/90 and 6/15/90, Snoddy A on 7/6/90, Snoddy B on 7/6/90, Detmer A on 7/13/90, and Stony East on 7/14/90. Three residential wells, #82, #86, and #212, had RWT detected in them. Residential well #86 had been abandoned when the owner connected to city water in 1986, but was elected to be sampled anyway by state agency representatives. Well #212 also had AY73 detected in it. Wells #82 and #212 were analyzed for PCBs and none were found to a detection limit of 0.1 ppb.

Calculations for mass balance were performed to determine the dye recovery at the six stations that had repeatable RWT detections. Those results are listed in Table 6. Clear Creek at First St. was included even though the connection is problematic. Only 18% of the total amount of RWT injected was recovered. The amount not recovered is attributed to losses by adsorption, underestimation of peak flows, underestimation of peak concentrations due to quenching effects of high concentrations on the SSFP, and tracer in ground-water storage. Sampling frequency was sufficient to catch all peaks of dye detections and is not considered to be a significant contributor to the lack of dye recovery. Slightly less than 97% of the amount recovered occurred at Quarry Springs. ICG-1 appears to be related to Quarry, perhaps

TABLE 5 - RHODAMINE WT APPEARANCES AS OF 6/29/90

Station	Date	Time	Concentration	Background
Kirby Rd.	5/19/90	0148	360 ppt	ND
Stony West	5/12/90	2230	260 ppt	ND
WN-1	5/12/90	2250	290 ppt	85 ppt
Stouts W.	5/12/90	2033	115 ppt	ND
		2233	280 ppt	
ICG-6	5/18/90	1130	58 ppt	ND
	5/29/90	1130	58 ppt	
Clear Cr.	5/13/90	0520	85 ppt	ND
@ 7th		0720	75 ppt	
		0930	120 ppt	
	5/14/90	0525	42 ppt	
	5/15/90	0520	44 ppt	
Cascade Br.	5/25/90	1215	275 ppt	95 ppt
WS-2	5/15/90	0641	54 ppt	ND
Bypass 37	5/28/90	0920	42 ppt	75 ppt
Defeat E.	6/15/90	0930	65 ppt	ND

as a subsidiary overflow route during storm events or as a distributary resurgence. Based on their morphology, elevations, and comparison of dye breakthrough curves, it is concluded that Slaughterhouse, PH Road, and PH Culvert are distributary outlets of a main conduit draining in that direction. The conclusion then, is that about 98% of the ground water from Lemon Lane resurges in the vicinity of Quarry Springs, about 1% or less flows to the three distributaries to the northwest, and another 1% or less may flow to other headwaters of Clear Creek above the Illinois Central-Quarry Spring Branch confluence. Some of ground water from the vicinity of Lemon Lane may be diverted to numerous other spring basins during storm flows, but the amount of ground water is probably no more than 0.01% of the total from Lemon Lane based on the tracer recovery ratios (Figure 10).

Hydrogeology of Lemon Lane Landfill

An examination of the Cross-section in Figure 3 shows that shallow strata on the east dip to the south-west and strata on the west slope trend to the south-east. This is interpreted to be an effect of sinkhole subsidence and collapse, as well as development of the conduit along a structural sag and trough. Strata adjacent to the conduit that collapsed also subsided in the direction of the collapse. A zone of interbedded limestone, dolomites, and shales at approximately the 798'-805' elevation is where the major water producing fractures occur. The different strain properties of the different interbedded lithologies caused bedding plane openings, vertical jointing, and general fracturing of the rock units in response to the subsidence stresses. This produced the zone of enhanced permeability noted on the cross-section. A higher density of fracturing also occurs within the first 10' of the bedrock. This is in accordance with Williams (1983), Ford and William (1989, p. 206), and others, who report that the uppermost layers of bedrock constitute a suspended aquifer termed the subcutaneous zone (Williams, 1983) or the epikarst (Mangin 1975).

TABLE 6 - RHODAMINE WT RECOVERY

Station	Mass (grams) Recovered	% of Injected	% of Recovered
Quarry	4744.243	17.40%	96.85%
ICG-1	64.135	0.24%	1.31%
Clear Cr. @ 1st	60.192	0.22%	1.23%
Slaughterhouse	22.032	0.08%	0.45%
PH Road	5.589	0.02%	0.11%
Ph Culvert	2.562	0.01%	0.05%
	4898.753	17.97%	100%

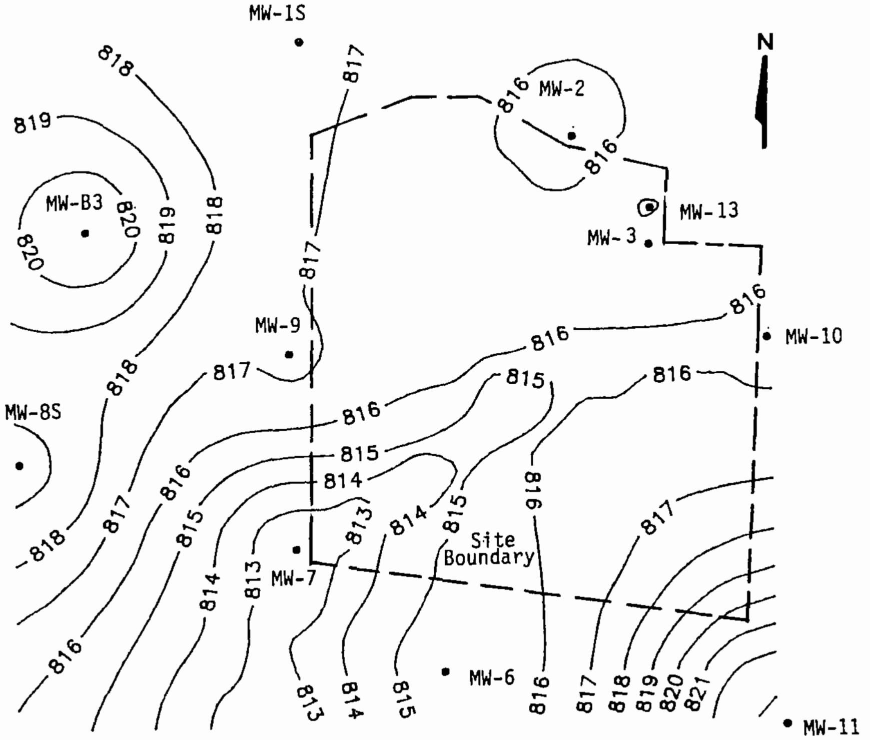

Figure 11 - Hypothetical Base Flow Map

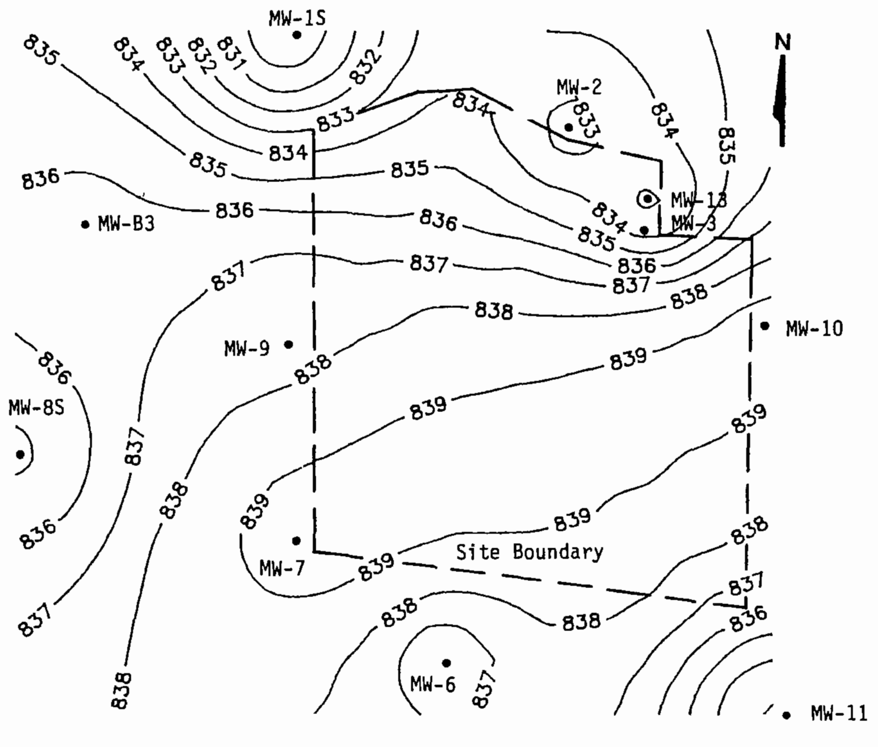

Figure 12 - Hypothetical High Flow Map

ground-water flow and contaminant movement based on that assumption erroneous. The original monitoring plan, which was predicated on the basis of intercepting a contaminant plume, with wells, is incapable of yielding reliable or accurate monitoring - except by improbably good luck.

(2) The aquifer has been shown to be a karst aquifer, one dominated by conduits in which flow is convergent; therefore, efficient, reliable, and time-sensitive monitoring of off-site movement of contaminants is best accomplished through the sampling of springs shown to be connected to the landfill by tracing as described by Quinlan and Ewers (1985) and Quinlan (1989). The tracer tests have clearly shown that most of the ground water flows rapidly via conduits to Illinois Central-Quarry springs during high flows. Off-site releases of contaminants in directions other than the Illinois Central-Quarry system would more efficiently and reliably be detected at some, or all, of those 32 springs shown by tracing to have intermittent connection with Lemon Lane rather than off-site wells.

(3) Since movement of ground water occurs in different directions during different flow regimes, monitoring schedules must be adjusted on the basis of hydrologic criteria. For example, during pre-excavation baseline sampling, it is proposed to collect samples during storm events as well as during base flow periods. When material is being excavated from Lemon Lane, it is proposed to sample Illinois Central Spring and Quarry Springs weekly and also during every storm event of sufficient magnitude and duration to elevate the discharge of those springs above their previous week's base flow discharge.

(4) On-site monitoring wells will provide water level data and samples to accomplish monitoring of ground water in storage near the site. This is a necessary component of a complete monitoring plan but, at Lemon Lane, the on-site wells were shown by tracing to be unreliable monitors for ground water leaving the site. This is because wells tend to intersect ground water in the diffuse bedrock aquifer and miss the conduits that convey most of the water. Part of that ground water is recharge from the backflooding of the conduits. However, the probability of wells intersecting conduits conveying the majority of the ground water, and, hence, the contaminants leaving the site is low. For the same reason, the use of proposed off-site wells to intercept contaminants that have left the site is considered to be inefficient and unreliable, especially since tracing has shown the springs to be time-sensitive and accurate monitoring stations for the potential off-site release of contaminants.

DEVELOPMENT OF A MONITORING PROGRAM AT A SUPERFUND SITE

IN A KARST TERRANE NEAR BLOOMINGTON, INDIANA

Michael R. McCann, Westinghouse Electric Corp.
Noel C. Krothe, Indiana University
Bloomington, Indiana

Q: Do you have any reason to believe that all the dye is out of the system?

A: On the contrary, we have evidence the dyes are still in the system. As late as September 23, 1991 Illinois Central Spring had 8.7 ppb Fluorescein and 12.3 ppb Rhodamine WT and Quarry Spring had 1.8 ppb Fluorescein and 8.0 ppb Rhodamine WT detected in grab water samples. We believe this is due to two reasons. One is that injection wells MW-1S and MW-10 were relatively "tight" and a portion of the dye simply remains in the fractures near the wells, draining slowly into the conduit system. This is particulary true of well MW-1S; during purging of this well for PCB sampling a great deal of concentrated Rhodamine WT was evacuated. Apparently, very little of the dye from this well entered the system at all. Secondly, a portion of the dye is thought to be in storage in the epikarst and adjacent fractured bedrock. This storage is either an artificial result of injection into rock borings under rising water levels, or a result of the backflooding of the conduit system during the storm event.

Q: When is it best to inject dye? Why didn't you catch the dye at its peak concentration?

A: If time and money will allow, it is best to inject dye first on a moderate flow event, followed by repeat injections on a low flow and then high flow event. At a minimum, one low flow event followed by one high flow event should be conducted. The low flow event is necessary to formulate your base-flow baseline sampling schedule. The high flow event is necessary, not only to establish your high-flow baseline sampling schedule, but also to make sure all off-site flow routes have been established, because upper level overflow routes may exist that operate only at the higher flows.

The dye pulse came through quickly, and even though the sampling frequency was every 15 minutes, it may not have been enough to catch the peak. In retrospect, when dye begins to appear visually, it is probably necessary to increase one's sampling frequency to every 5 minutes or less.

HETEROGENEITY IN CARBONATE AQUIFERS;

EFFECTS OF SCALE, FISSURATION, LITHOLOGY AND KARSTIFICATION

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Abstract

The flow transmission properties of carbonate aquifers may be determined and inter-compared using cumulative probability plots of log hydraulic conductivity obtained from slug and bailer tests in observation boreholes. Such tests are simple and inexpensive, and allow determination of both average hydraulic conductivity, and an indication of the extent of aquifer heterogeneity. Karstified aquifers, such as the Carboniferous Limestone of the Mendip Hills, exhibit a large range of hydraulic conductivities, while fissure flow aquifers, such as the Great Oolite of the adjacent Cotswold Hills, have both higher hydraulic conductivities and are less heterogeneous. The importance of fissure density and karstification can be illustrated using caliper logs and aquifer tests of different scales. Spatial variations of aquifer properties also occur, and can be related to variations in lithology and structure, and to the effects of quarrying.

Introduction

It is generally recognised that the behaviour of carbonate aquifers is dependent on the type of groundwater flow occurring, although the recharge and storage properties are also of importance (Smart and Hobbs, 1986). Various authors suggest a continuum of flow types ranged between two end members (conduit and diffuse flow; White (1977), Smart and Hobbs (1986)), three end members (conduit, diffuse and fissure network flow; Atkinson (1985)) and even four end-members (granular, fracture, diffuse and conduit flow; Quinlan (1988)). Hydrodynamic (Aley, 1975; Atkinson, 1977) hydrochemical (Schuster and White, 1971) methods have been used to characterise flow type for particular aquifers, and a list of possible characteristic criteria was presented by Smart and Hobbs (1986). Another approach is to consider the spatial variation of aquifer transmission properties which can be obtained from borehole aquifer tests (specific yield, transmissivity or hydraulic conductivity). Diffuse flow aquifers would be expected to exhibit only limited spatial variations while karstified aquifers are characteristically heterogeneous and would show much greater variation (Daoxian, 1986; Hobbs and Smart, 1988), in this paper we present data derived from simple slug and bailer tests, which can conveniently be displayed using cumulative probability plots, a tool often used by hydrogeologists to display well specific capacity data (Csallany and Walton, 1963). The data may also assist in understanding the factors controlling the spatial distribution of aquifer hydraulic conductivity, and provide a data-base suitable for parametarisation of numerical aquifer models.

Methods

Slug and bailer tests (Ferris and Knowles, 1963) provide a simple, quick and cheap method for estimating aquifer hydraulic conductivity. Unlike the pumping tests, more conventionally employed to determine aquifer parameters, they do not require installation of submersible pumps, and can thus be undertaken in small diameter observation boreholes. The logistic support required to install, power and discharge water from the pump is thus eliminated. Typically slug and bailer tests can be completed in a few hours and require only one or at the most two operators they are therefore ideal for geotechnical and other low costs investigations in which development of a pumping supply borehole is not the prime objective. In these tests, a slug of water is either introduced into or removed from the borehole, and the return of the water level to pre-existing conditions is monitored, preferably using a sensitive pressure transducer and chart recorder or data logger. The initial change in water level should theoretically be instantaneous, and it is often easier to approximate this condition by either inserting or removing a long weighted float to generate the disequilibrium in head conditions. A number of theoretical solutions to this problem have been presented which permit direct calculation of hydraulic conductivity in a variety of different situations (see review by Karasaki et al, 1988). In our study of water-table aquifers the solutions of Bouwer and Rice (1976) and Van der Kamp (1976) for the underdamped case were employed. It should be noted that these solutions are based on the assumption of laminar flow within the aquifer, wheras in aquifers with dissolutionally enlarged voids turbulent flow may develop under the test conditions.

Hydraulic conductivity data follow a log-normal distribution and thus can be linearised by plotting log hydraulic conductivity on cumulative frequency probability paper. The mean and standard deviation are determined for the normally distributed log data, and can be used to calculate the coefficient of variability. The standard deviation can also be converted back to non-logarithmic form to provide upper and lower limits which are asymmetric about the mean, but are expressed directly in the units of hydraulic conductivity (here m/d).

Comparison of Hydraulic Conductivities from Four Contrasting Carbonate Aquifers

Hydraulic conductivities have been obtained from slug and bailer tests (henceforth slug tests) in four contrasting aquifers. The Carboniferous Limestone aquifer of the Mendip Hills is a maturely karstified aquifer developed in massively bedded, well jointed limestones of low primary porosity (< 0.1 %) (Green and Welch, 1965). Flow in the aquifer is predominantly via conduits, but fractures and fissures provide the majority of the saturated zone storage in the diffuse flow zone adjacent to the conduits (Atkinson, 1977). Abstraction is wholly from spring sources. The slug tests were conducted in the eastern Mendips (Hobbs and Smart, 1988; Atkinson et al, 1973), where surficial karst features are less well developed than in the better known and more mature central Mendips area

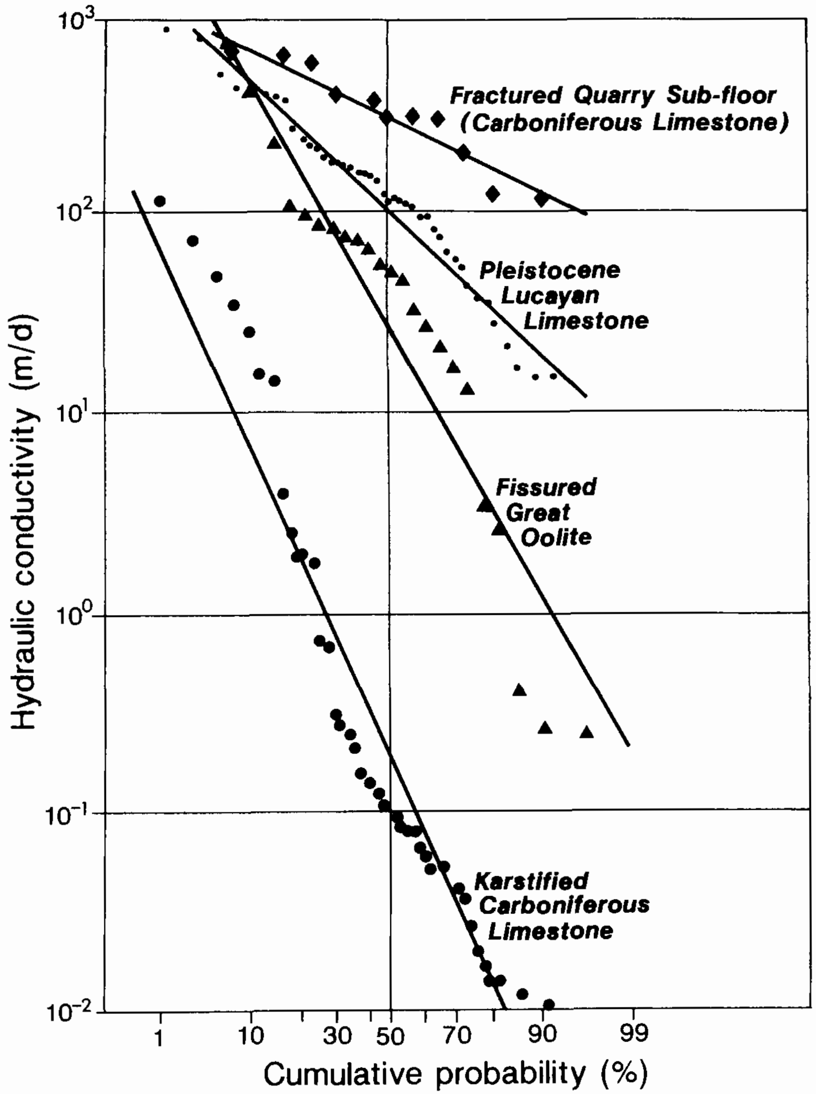

Figure 1 *Cumulative probability plot of log hydraulic conductivity determined from slug tests for quarry sub-floor, Lucayan Limestone, Great Oolite and Carboniferous Limestone aquifers.*

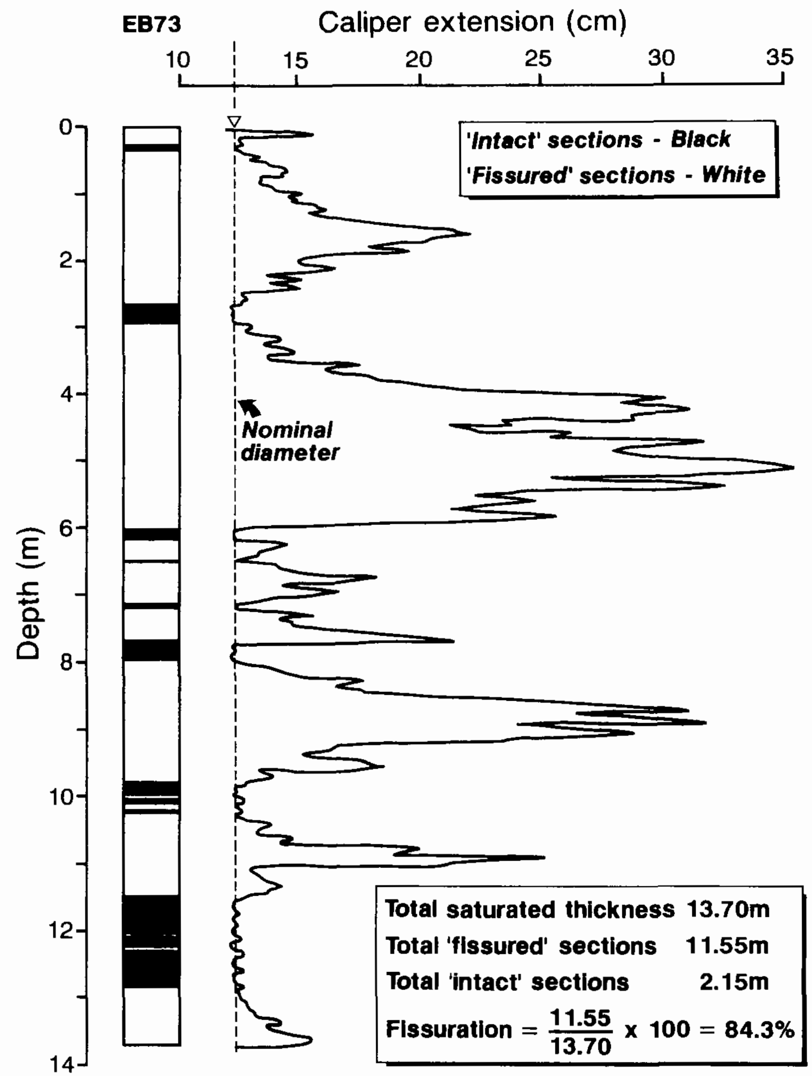

Figure 2 *Caliper log for Grand Bahama borehole EB 73, and calculation of fissuration index.*

described by Atkinson (1977). Within the East Mendip area, the Carboniferous Limestone has been extensively extracted by hard-rock quarries, which extend below the natural water table (Edwards et al; this volume). Modern blasting techniques, which are designed to comminute the rock, also cause extensive fracturing of the Carboniferous Limestone both below the quarry floor and laterally away from the walls (Holmberg and Maki, 1981; Gunn and Gagen, 1987). Where it is below the water table the sub-floor blast zone can be considered a separate aquifer characterised by high fracture density, low primary porosity and negligible conduit development. Our data are derived from shallow (< 10m) boreholes, drilled into the sub-watertable quarry floor.

The Great Oolite aquifer of the southern Cotswolds comprises massive oolitic and bioclastic limestones (Kellaway and Welch, 1980), and has been extensively developed by large abstraction boreholes (> 10 ML/d). Transmissivities are high and are dominated by development of dissolutional fissures upon the bedding planes and extensive joint systems. These fissures provide the majority of the storage (specific yield c 3%), but there is also significant intergrannular porosity (up to 15%). Test data has been derived from a network of observation boreholes. Unlike the Great Oolite and Carboniferous Limestone aquifers, the Pleistocene Lucayan Limestone aquifer of the Bahamas has never suffered deep burial (Beach and Ginsberg, 1980), and has a high intergrannular porosity (typically 40%) and specific yield (5-6 %). It has however been karstified, with vadose meteoric leaching during times of low Quaternary sea-level, and dissolution in the freshwater phreatic and mixing zones at times of high sea-stands (Smart and Whitaker, 1988), resulting in vuggy porosity and development of coastal (dune-flank) caves (Mylroie and Carrew, 1990). The data presented here have been derived from observation boreholes and drainage wells on the island of Grand Bahama.

Hydraulic conductivity data from these four contrasting carbonate aquifers is presented in Figure 1, while summary statistics are given in Table 1. The densely fractured blast zone aquifer has the highest mean hydraulic conductivity, and is also spatially the most homogeneous of the aquifers. The Lucayan Limestone and Great Oolite aquifers attain similar maximum hydraulic conductivities but exhibit a much greater range of test values, indicating greater heterogeneity. In the case of the fissured Great Oolite this greater heterogeneity results from a relatively small proportion of sites having either unusually high, or very low hydraulic conductivities, the lower quartile spanning two orders of magnitude. The former are associated with fissure network zones which are responsible for the rapid transmission observed in the aquifer from tracer tests (Smart 1976), while the latter represent tight zones where bedding plane and fracture density are low, and extensive dissolution to generate fissures has not occurred. A similar pattern is seen for the karstified Carboniferous Limestone, the data from which also span four orders of magnitude. None of the boreholes tested intersects on open conduit. Had this been the case, much higher maximum apparent hydraulic conductivities would have been obtained, significantly increasing the range. The mean hydraulic conductivity of the Carboniferous Limestone is much less than that of the Great Oolite, both because of the more massive bedding, and because concentration of flow into conduits at an early state in the aquifer development limits the extent of dissolution in the diffuse zone (Ewers 1978). In the more densely fractured Great Oolite, competition between flow routes of similar size ensures a more uniform distribution of dissolution (Palmer, 1984).

Thus conduit flow aquifers are characterised by high heterogeneity and a rather low transmissivity dominated by the diffuse flow zone. Fissured aquifers have a higher transmissivity and significantly lower heterogeneity, as indicated by the coefficient of variation (Table 1), and the gradient of the log probibility plot. Both the Pleistocene Lucayan Limestone, in which intergrannular flow is significant, and the fracture flow quarry sub-floor have high transmissivity and low heterogeneity. The mean transmissivity

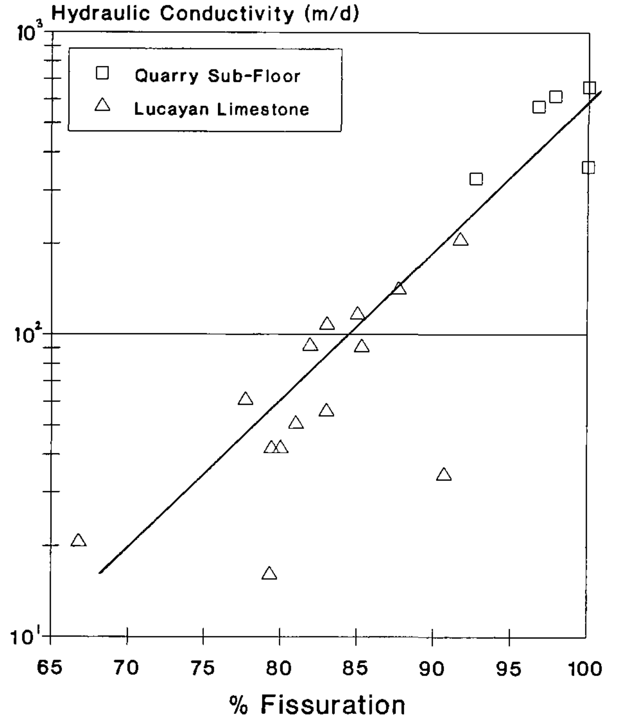

Figure 3 *Relation between fissuration and log hydraulic conductivity for Lucayan Limestone and quarry sub-floor aquifers.*

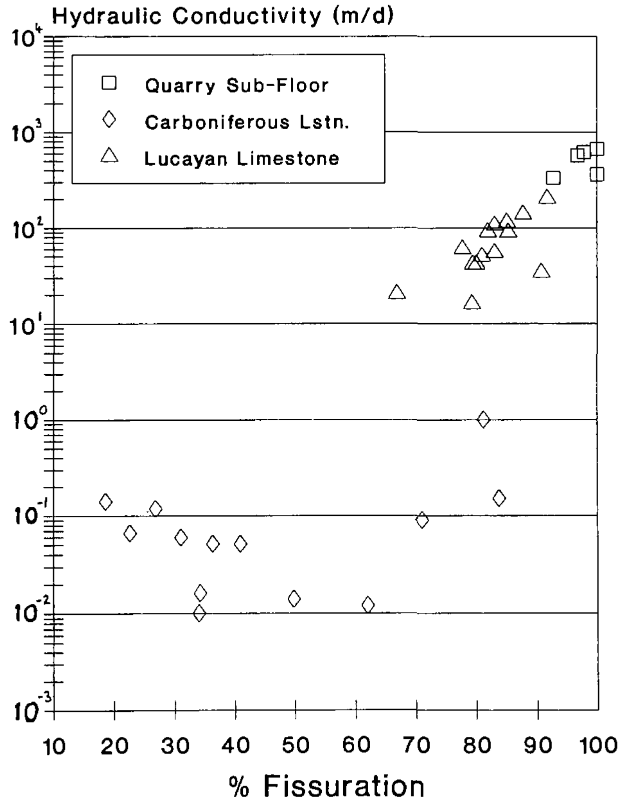

Figure 4 *Comparison of relation between fissuration and log hydraulic conductivity for Carboniferous Limestone and Lucayan Limestone and quarry sub-floor aquifers.*

is thus an indication of the extent of dissolution enlargement of voids within the aquifer, while the coefficient of variation (or slope of the log probability plot) indicates the degree of flow concentration occuring on the continuum, from diffuse to conduit flow (Smart and Hobbs, 1986).

Role of Dissolution and Void Integration

If the aquifer hydraulic conductivity at a particular site is controlled by the development of dissolutional fissures, it should be possible to predict hydraulic conductivity directly from the aperture and number of fissures penetrated by the borehole. In order to investigate this relationship, selected boreholes were caliper logged using a Gerhart T-450 borehole logging system (Keys and Maccary, 1971). An index of fissuration was then derived from the caliper logs by summing the length of borehole segments which were greater than the nominal drilled diameter, and expressing this sum as a percentage of the total saturated borehole depth (Figure 2). The index should thus depend both on the number and size of fissure openings, although in practice the latter will be overestimated by 'break-out' of the drill bit into the fissure. It is also important to remember that widening of the hole may not simply be due to the presence of open fissures, but can occur in softer rock units, such as poorly cemented limestones in the diagenetically inmature Lucayan Limestone, or shales in the Carboniferous Limestone. Furthermore, the amount of 'break-out' which occurs into a fissure is strongly dependent on method of drilling, drilling rate and rock matrix strength, which are not controlled in our data set.

Figure 3 presents data from the Lucayan Limestone aquifer of Grand Bahama, and the quarry sub-floor aquifer. For these there is a remarkably good relation between percentage fissuration and the measured hydraulic conductivity (which is not dependent on the coincidence of the quarry and Lucayan Limestone relationships). Thus even in young limestones with high intergrannular porosity, the transmission properties of the aquifer are governed by development of dissolutional fissures. In the Bahamas these form along the frequent exposure surfaces which terminate early carbonate sub-unit and are developed during low sea-stands; such surfaces are laterally continuous and thus generate very high horizontal hydraulic conductivities.

In Figure 4, the Bahamas and quarry sub-floor data are supplemented by a data from the karstified Carboniferous Limestone aquifer. Because the intergrannular porosity of this limestone is very low, secondary voids would be expected to dominate the transmissivity. In fact there is not a significant relationship between the fissuration index and borehole hydraulic conductivity (it should however be noted that unfortunately the logged boreholes did not include any sites with above average hydraulic conductivities). The probable explanation for this unexpected finding is related to the connectivity of individual fissure openings. Packer testing has shown previously that some fissures yield hydraulic conductivities similar to the bulk rock, indicating that they do not interconnect laterally with other voids in the aquifer (Hobbs and Smart, 1988). They thus contribute to storage within the aquifer by delayed leakage, but are not important for flow. Thus as emphasized by Sendlein and Palmquist (1977) void intergeration is critical in controlling transmission in karstified rocks. Whilst in the blast zone and Bahamas fissure voids integrate laterally, in the Carboniferous Limestone this is not the case and the resulting aquifer is both less transmissive and more heterogeneous (Figure 1).

Careful examination of the data for the individual aquifers (Figure 1) suggests that there may be sub-populations present in the data, as indicated by linear segments of differing gradient on the cummulative probability plot. This is illustrated for the Carboniferous

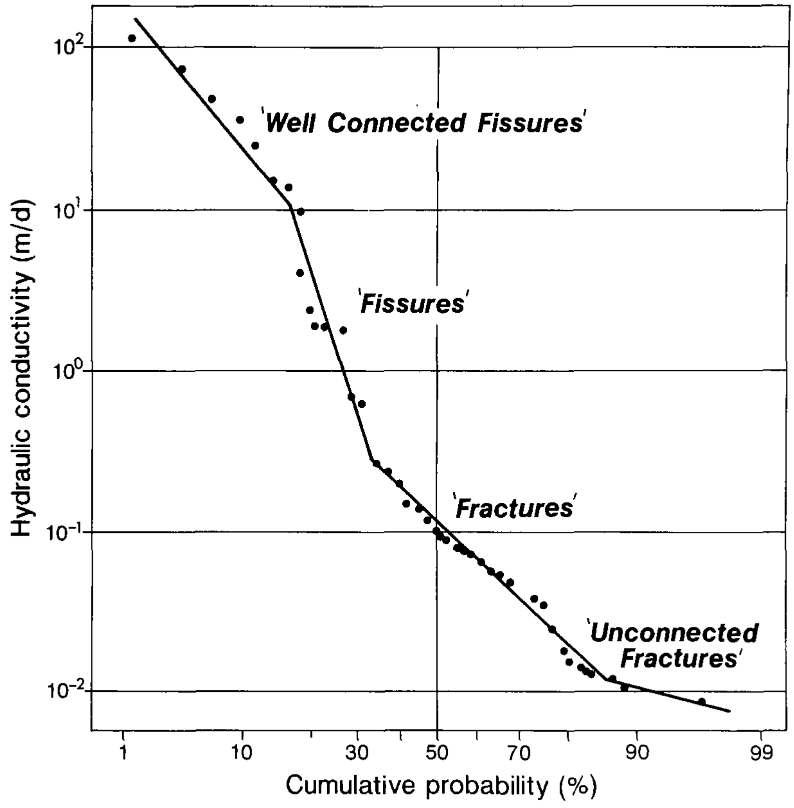

Figure 5 *Slug test data for the karstified Carboniferous Limestone sub-divided into linear components.*

Table 1 Comparison of hydraulic conductivity data for the four aquifers

Carbonate Aquifer Type	Mean (m/d)	Coefficient of Variation (%)	Total Number of Boreholes
Fractured Quarry Sub-Floor	305	10.4	11
Pleistocene Lucayan Limestone	96.2	29.5	44
Fissured Great Oolite	31.8	70.1	24
Karstified Carboniferous Limestone	0.214	179	46

Limestone aquifer (for which we have most data) in Figure 5, which shows four sub-populations represented by straight line segments. These sub-populations may simply be a statistical artefact, but could also represent the combined effects of void integration and dissolution. Fractures (width mm) are unmodified joints and beddings developed by unloading an re-exposure of the limestone, while fissures (width cm) have suffered dissolutional modification, and in some cases may have turbulent flow. They have, however, not expanded to the size of conduits (width tens of cm to m). Thus while fractures give lower hydraulic conductivities than the larger aperture fissures, the degree of void integration is also critical. Unconnected fractures give very low hydraulic conductivities, which increase significantly once integration of the net occurs, for instance by continued unloading (or in the quarry sub-floor case by blasting). The enhanced circulation permits dissolution generating fissures, but initially these are interconnected only by fractures which effectively limit the increase in hydraulic conductivity. Continued circulation expands these fracture links giving a net of highly connected fissures with high hydraulic conductivity. In fact it is precisely such a net which characterise the Great Oolite aquifer. These suggestions could be more rigourously tested by application of simulation models, such as those developed by Foster and Milton (1974) for the Chalk aquifer of southern England.

Controls on the Spatial Variation of Aquifer Hydraulic Conductivity

Because slug tests are relatively rapid and simple they can be used to generate sufficiently large data sets that controls on the spatial variations of aquifer hydraulic conductivity can be investigated (provided sufficient suitable boreholes are available). For the East Mendip Carboniferous Limestone data set, an initial sub-division was made into boreholes on the steeply dipping (c 70°) northern limb of the Beacon Hill pericline, and those from the more gentle southern limb (dip c 30°) (Figure 6). These differ significantly in hydraulic conductivity (at the 95% confidence interval), the northern limb having an average hydraulic conductivity on order of magnitude less than the southern limb. The northern limb also has markedly less sites in the 'fissure' range (10^{0} to 10^{1} m/d). These differences are paralleled by a much greater degree of fissuration on the southern (average fissuration index 78.6%) than the northern limb (34.6%). As both areas have a similar geomorphic history, this difference may be attributed to the difference in dip which controls the number of bedding planes intersected by a vertical borehole. This is much lower in the steeply dipping beds, of the northern limb, than for the more gently dipping southern limb. Because it is the bedding planes which provide laterally continuous and thus interconnected openings, the steep dips yield lower hydraulic conductivities on the northern limb.

The Carboniferous Limestone data was also sub-divided with respect to the lithological group within the Carboniferous Limestone Series (Green and Welch, 1965). Again significant differences are apparent, both with regard to the average hydraulic conductivity and the heterogeneity indicated by the coefficient of variation (Figure 7, Table 2). Although there are lithological differences between the groups, (the Vallis and Hotwells Limestones are coarse, pure bioclastic limestone, the Clifton Down contains a major mudstone unit, and the coarse bioclastic Blackrock Limestone has both significant shales and chert beds), these lithological factors appear to be less important than disposition in the aquifer. Thus the impure Blackrock Limestone would be expected on lithological grounds to be less transmissive than the pure Vallis Limestone. The converse is in fact the case, because the Blackrock Limestone has been exposed to much greater dissolution as it is at the base of the limestones, and receives aggressive allogenic runoff and diffuse leakage from the topographically higher siliclastic rocks forming the core of the anticline. The

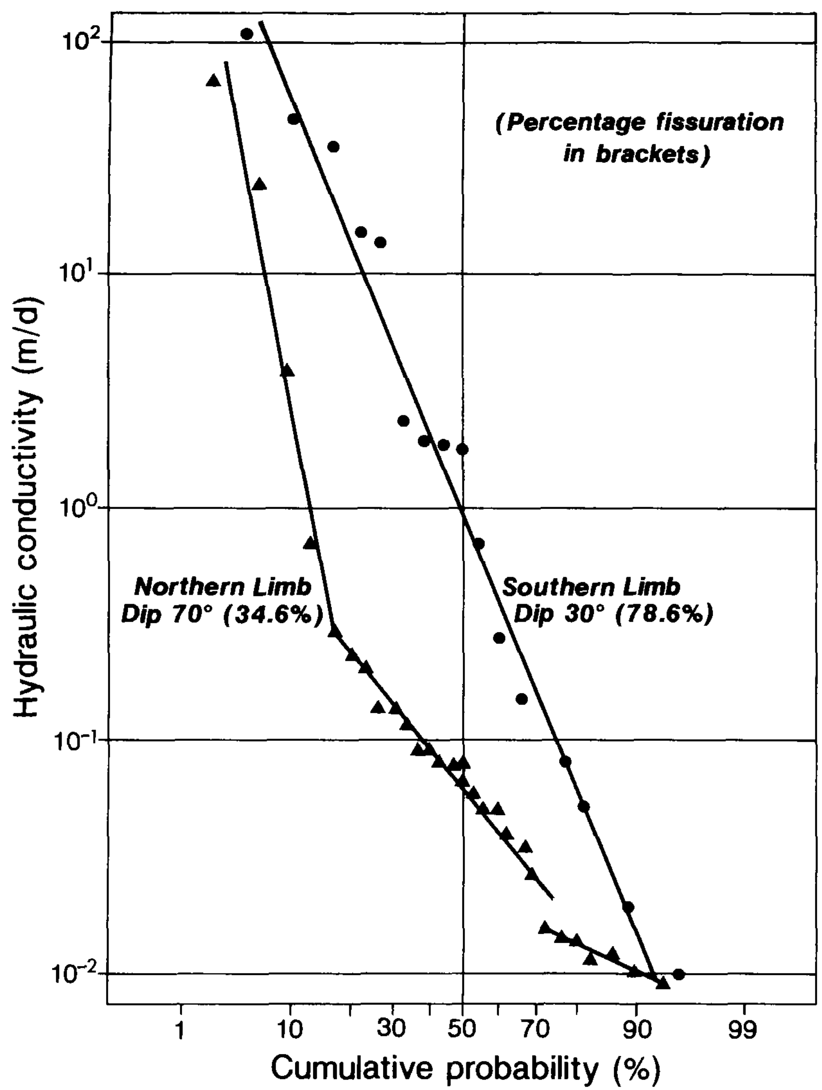

Figure 6 *Comparison of hydraulic conductivity data for boreholes from the northern and southern limbs of the Beacon Hill pericline. Carboniferous Limestone aquifer.*

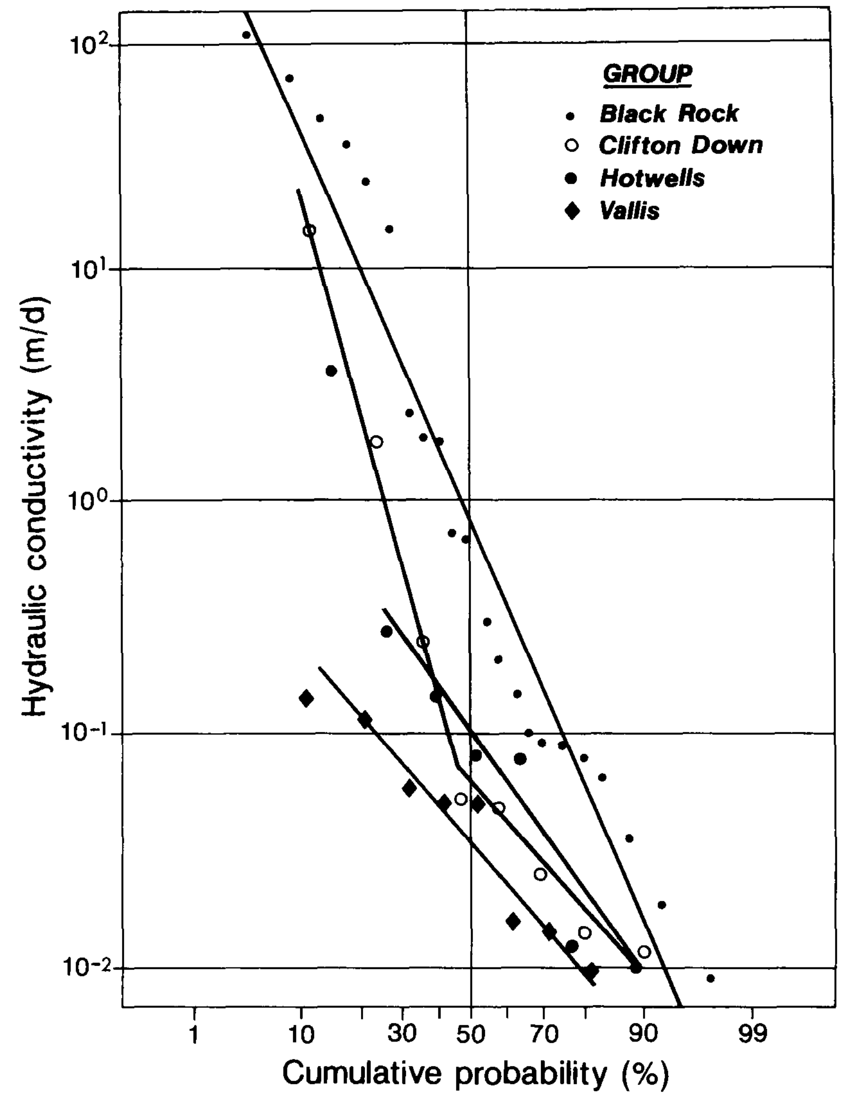

Figure 7 *Slug test data for the Carboniferous Limestone aquifer sub-divided into lithologic groups.*

Vallis Limestone in contrast has received only autogenic recharge, and dissolutional integration of the initial fractures has been limited.

Interestingly, the Blackrock Limestone has also been more extensively quarried than the other limestone groups, and some of the tested boreholes are in fact drilled within the quarries. If we only consider sites which have water-levels well below (< 10 m) the sub-floor blast zone, these have hydraulic conductivities much higher than boreholes which are remote from quarries (>150 m; Figure 8). Whilst this may be explained by the lithological controls discussed above, the implications for geotechnical aquifer parameterisation studies which draw data only from the quarry sites are potentially serious.

Table 2 **Hydraulic conductivity data for the Carboniferous Limestone sub-divided into different lithological groups**

Group	Mean (m/d)	Coefficient of variation (%)	Total Number of Boreholes
Blackrock	0.97	9330	22
Clifton Down	0.123	120	7
Hotwells	0.102	88.4	7
Vallis	0.025	43.8	9

Effect of Scale

Kiraly (1975) has suggested that in karstified aquifers there is a systematic relationship between the effective transmissivity of the aquifer and the scale considered. Thus at the catchment scale conduits dominate transmission, which at the other extreme in core samples only intergrannular voids are present, and measured hydraulic conductivities are very low. Slug tests involve displacement of only a limited volume of water, and thus 'sample' only a relatively small zone of the aquifer within 1 to 2 m of the test borehole (depending on slug size and aquifer hydraulic conductivity). In contrast pump tests may cause more extensive drawdown over distances from tens to hundreds of metres from the test borehole. Figure 9 contrasts our slug test results with pump test data for Grand Bahama obtained by Little et al (1975). Over 60% of the pump test results have higher hydraulic conductivities than the maximum recorded in the slug tests, but the two data sets give similar minimum values. This suggests that in pump tests the cone of depression expands to intersect dissolutional conduits, whose spacing is sufficiently wide that they are not penetrated directly by random boreholes. It should also be born in mind that at high hydraulic conductivities such as these turbulent flow may develop and the transmissivities determined may be erroneous.

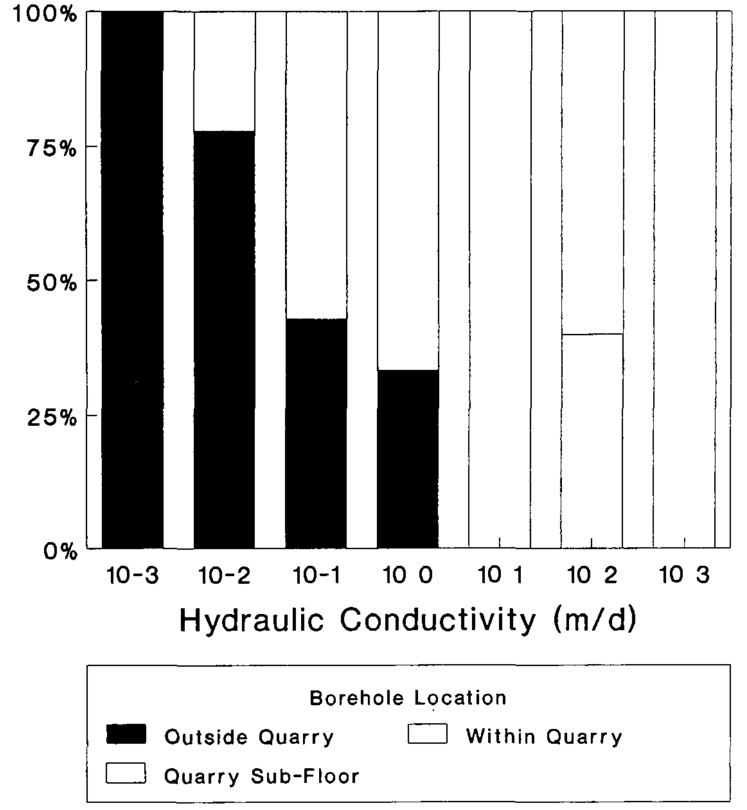

Figure 8 *Comparison of hydraulic conductivities for boreholes within quarries, in the quarry sub-floor, and greater than 150 m from quarries, Carboniferous Limestone aquifer.*

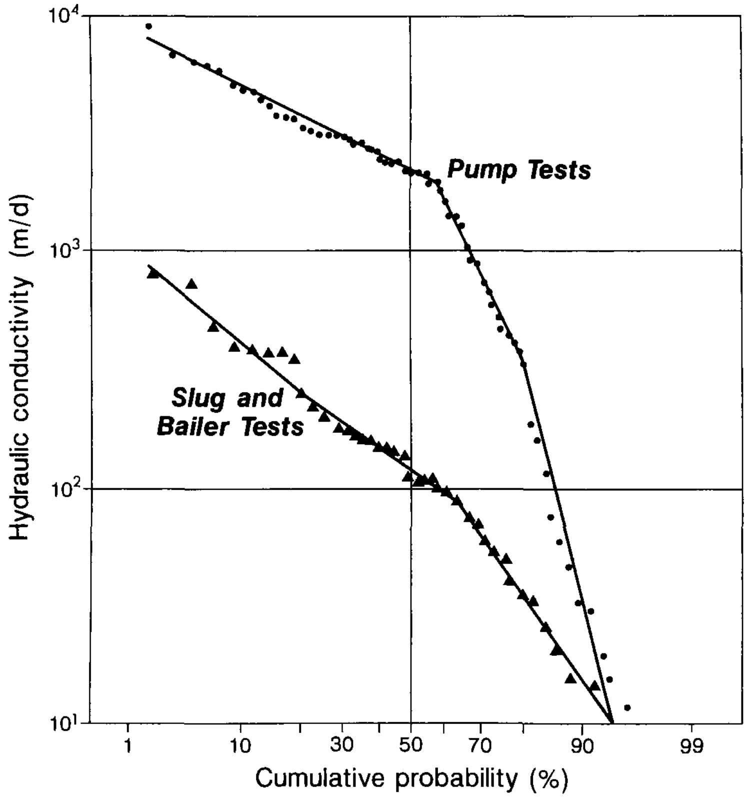

Figure 9 *Comparison of hydraulic conductivity data for the Lucayan Limestone aquifer, Grand Bahama derived from pump and slug tests.*

Conclusions

In our study we have made use of small diameter observation boreholes which do not have pumps installed, and are therefore readily tested by slug and bailer methods. In other areas where large numbers of domestic abstraction wells fitted with pumps are available, use could be made of borehole specific capacity data (or transmissivity derived from this data) see for instance (Sallany and Walton, 1963). In both cases log probability plots offer an easy and effective method for quantification of the position of the aquifer on the flow spectrum from difuse to conduit within the conceptual model of Smart and hobbs (1986). Different carbonate aquifers can be readily compared and contrasted, and insight gained on the flow properties. Hydraulic conductivity in carbonate aquifers is controlled by void size and void integration. In the more densely fractured aquifers, void integration is effectively complete, giving a relatively homogeneous aquifer, and the density of fissuration and degree of dissolution enlargement are the main factors controlling mean hydraulic conductivity. In less densely fractured aquifers, void integration becomes much more important, and where dissolutional development is not extensive, the aquifer is much more heterogeneous, especially where flow concentration into conduits has occured. Lithology and structure may be important in controlling aquifer transmissivity, but disposition of the unit within the aquifer may be equally important where dissolutional void enlargement is important. Thus, the approach to aquifer characterisation exemplified in this paper has two important applications. On a practical basis it provides a means of describing the transmission properties of carbonate aquifers, providing data in a format suitable for stochastic parameterisation of aquifer models. It also provides a simple design tool useful for prediction of borehole yields, and the preferred mode of aquifer development (Smart and Hobbs, 1986). On a more academic level it provides data to elucidate the controls on transmission and structure of carbonate aquifers. This also has implications for aquifer management, for instance transmission of pollutants from waste disposal sites via the diffuse flow zone in karstified limestones (Edwards and Smart, 1989) and the validity of geotechnical measurements of aquifer parameters.

Acknowledgements

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HETEROGENEITY IN CARBONATE AQUIFERS: EFFECTS OF SCALE, FISSURATION, LITHOLOGY, AND KARSTIFICATION

By: Peter L. Smart, Alan J. Edwards, and Steve L. Hobbs

Q. **Relative to the use of caliper logs to characterize the degree of fissuration, how was fracture aperture determined? What about single solutionally enlarged fractures? The caliper log would seem to be an indicator of borehole instability and be a measure of factors other than single fracture density.**

A. Borehole caliper logs are controlled by the degree of aquifer fissuration as well as the drilling method and borehole instability. The percentage fissuration index described in this paper is a crude method of measurement. However, it offers a pragmatic and simple technique which we believe generates interesting and useful results.

Q. **Did you use different methods of slug-testing? Did you observe any differences between the hydraulic conductivities determined with various slug-test methods and a bailer? If so, how do you account for any observed differences?**

A. In all slug and bailer tests, weighted floats were inserted and removed from the water within the tested borehole. This technique was utilized in all the tests. Hydraulic conductivities determined from the slug rather than the bailer test were generally higher. This was because the rise in water level, due to float insertion, resulted in an immediate water loss to the surrounding unsaturated rock. In order to overcome this disparity, the average hydraulic conductivity from 6 slug and bailer tests was calculated for each tested borehole.

CAUSTIC WASTE CONTAMINATION OF KARSTIC LIMESTONE AQUIFERS IN TWO AREAS OF JAMAICA

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Abstract

Jamaica's bauxite deposits occur in association with the karstified White Limestone Formation of Tertiary Age. The refining of bauxite to alumina results in a waste known locally as "red mud", which is stored in unsealed mined out karstic depressions. The red mud, which is more than 70% water is highly caustic and infiltrates to the groundwater table. Groundwater contaminated by "red mud", shows increased sodium concentration and increased pH. Monitoring of groundwater quality in two areas of the island, where bauxite/alumina plants are located, indicate significant contamination over the last 20 years. In these two areas, caustic contamination has resulted in over 20 km^2 of aquifer being unsuited for groundwater development. The mud disposal ponds are located in the direct path of groundwater flow and still poses a serious threat to groundwater reservoirs and consequently the groundwater reserves of the island. Remedial measures to reduce contamination have included sealing of disposal ponds, thickening of mud and solar drying and recycling of liquid fraction.

Introduction

A highly caustic waste known locally as "red mud" is a by-product of the extraction of alumina from Jamaican bauxite, which occurs as an interfingered blanket deposit in association with the limestone deposits forming the exploitable aquifer of the island. Four plants located on the south coast of the island, produce approximately 3Mm^3 of red mud waste annually and an increase to 4.5Mm^3 is expected with expansion of the present plants and construction of one new plant. The red mud waste is stored in mined out or topographic depressions in the limestone.

TABLE 1 GROUNDWATER RESOURCES AVAILABILITY

	Mm^3/Yr	Migd	Percent of Total
GROUNDWATER SAFE YIELD	3418	2062	
FROM LIMESTONE AQUIFERS	3294	1987	96
FROM ALLUVIUM AQUIFERS	124	75	4
PRODUCTION (WELLS & SPRINGS)	839	506	
FROM LIMESTONE AQUIFERS	706	426	84
FROM ALLUVIUM AQUIFERS	133	80	16
UNUTILIZED RESOURCES	2579	1555	75

Limestone rocks in Jamaica have a high risk factor for pollution because of the development of relatively large channels for groundwater infiltration and percolation (karstification). The limestone aquifers in Jamaica are a major source of water supply because of the high transmissivities and storage especially along fault and fracture lines and in areas where karstification has produced a network of solution openings. Limestone rocks underlie 67% of the island (figure 1) and important groundwater reservoirs are to be found in the limestone close to major urban and agricultural areas. There are more than 400 wells tapping the limestone aquifer and abstracting approximately 706 $Mm/^3$ yr of water or 84% of the total water produced (Table 1).

The disposal sites (figure 1) are directly in the flow path of groundwater to major reservoir areas. No structural or lithologic barriers exist in the limestone and changes in groundwater abstraction patterns could induce a more wider spread of the contaminant. At sites 1 and 4 (figure 1) caustic contamination of groundwater has been detected. Several springs and wells have been abandoned and a significant portion of the limestone aquifer adjoining these sites is unsuitable for further groundwater development.

Geology/Hydrogeology of the Limestone

The White Limestone Formation is a thick series of pure calcium carbonate ranging in age from middle Eocene to Miocene (47M yrs - 10M yrs). The entire sequence is estimated to be more than 3,000 meters thick and has been subdivided into ten (10) biostratigraphic and four (4) hydro-stratigraphic units. Hydrogeologic properties of the entire series are broadly similar and there is hydraulic continuity between all the units. Extensive tectonism has resulted in folding, faulting and fracturing of the rocks.

Figure 1 SIMPLIFIED GEOLOGY OF JAMAICA

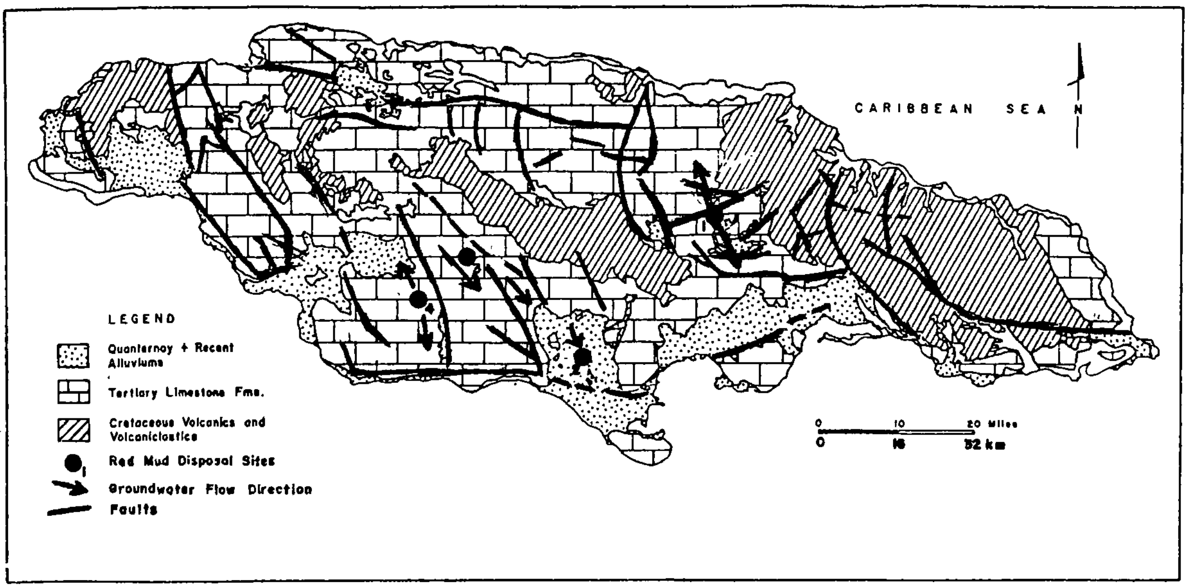
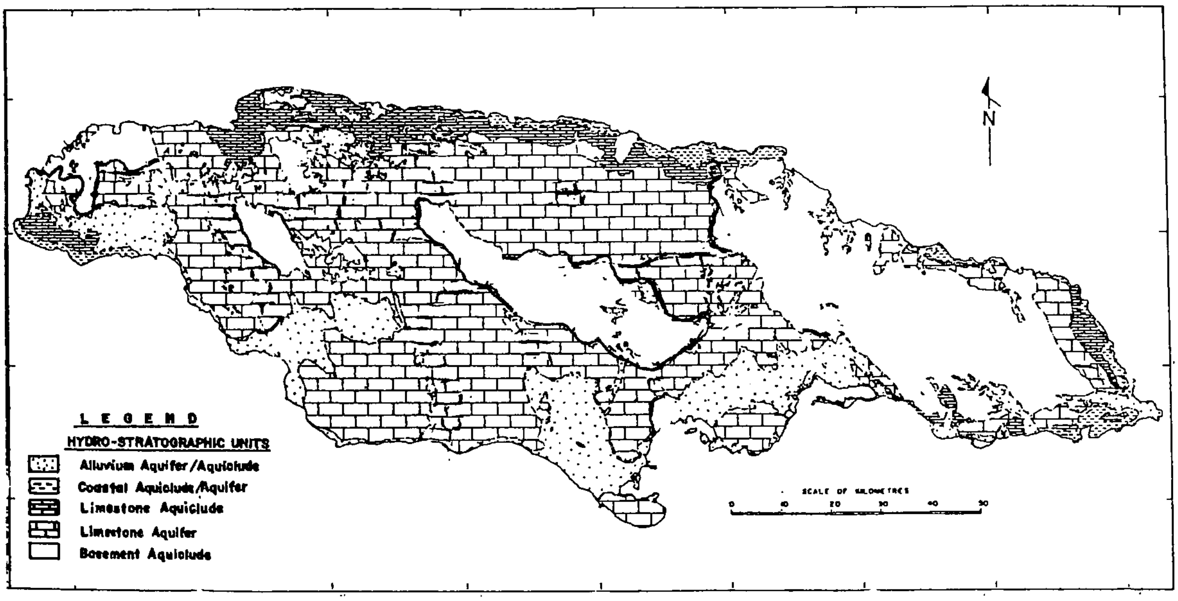


FIGURE 2 HYDRO-STRATIGRAPHIC MAP OF JAMAICA



The regional dip of the rocks is north and south towards the sea from the central anticline. The regional dip combined with the dominant alignment of faults and fractures, the topographic gradient and the higher rainfall on the central highlands impose a regional pattern of groundwater flow from the central highlands northwards and southwards to the sea.

In the limestone uplands, faults and associated fracture zones are major conduits for groundwater and transmissivities are substantially higher than in intervening blocks.

In the lowlands, solution of the limestone has resulted in an integrated network of solution openings, imparting to the aquifer almost homogenous and isotopic properties. In the lowlands conduits which originate in the uplands may continue as cave systems or subterranean streams. Most of the recharge of the limestone aquifer takes place in the uplands, where soil cover is absent and rainfall is the highest, while storage is in that portion of the aquifer underlying the coastal plains. Lateral flow of groundwater takes place both in the vadose and phreatic zones. In the vadose zone, lateral flow is rapid and confined to narrow channels which may continue into the phreatic zone. Most of the water in the phreatic zone moves through the interconnected network of solution openings and fracture joints.

Limestone Hydrostratigraphic Units

The limestone is subdivided in four (4) hydrostratigraphic units (figure 2) viz:

- Limestone Aquifer
- Limestone Aquiclude
- Coastal Aquifer
- Coastal Aquiclude

The limestone aquifer is comprised of members of the White Limestone Group. They form a sequence of moderately compacted, well bedded, partially crystallised bioclastic and micritic limestones with thickness in excess of 2,000 metres. The aquifer exhibits mature karstic features, typified by a very high infiltration capacity, predominant sub-surface drainage and highly compartmentalised sub-surface conduit flow. Its karstic nature makes the limestone aquifer very susceptible to contamination.

The limestone aquiclude is a soft, fine grained chalk of low permeability which fringes the coast ponding groundwater within the juxta posiitoned limestone aquifer. Thickness approaches 1000 metres.

The coastal aquifers are raised reefs patchily deposited along the north coast of the island. These reefs form highly karstified limestone aquifers of high permeability and low groundwater storage potential. Thickness is usually less than 50 meters.

The coastal aquicludes are soft marls fringing the coast of the island. It dams groundwater flow which occurs along faults through the limestone aquiclude such that springs issue along the limestone/coastal aquicludes boundary. Thickness approaches 300 meters.

TABLE 2 RED MUD INSOLUBLE SOLIDS

CONSTITUENT	PERCENT
LiO	11.0
SiO_2	5.5
Al_2O_3	12.0
Fe_2O_3	49.5
P_2O_5	2.0
CaO	8.0
Na_2O	3.5
TiO_2	5.0
MnO_2	1.0
Miscellaneous	1.5

TABLE3 RED MUD SOLUBLE SOLIDS

CONSTITUENT	AMOUNT
Al_2O_3	2.5g/kg liquid
NaOH	3.7g/kg
Na_2CO_3	1.6g/kg
Na_2SO_4	0.4g/kg
Nacl	0.7g/kg
$Na_2C_2O_4$	0.1g/kg
Spec. Gravity	1.008
pH	10.5 units
BOD	6 ppm
COD	148 ppm

Table 4

ANALYSES OF RED MUD LIQUID FRACTION AND RECOMMENDED CONCENTRATION LIMITS

Constituent	Concentration in Red Mud Liquid mg/litre			Recommended Limits: Domestic *		Irrigation &** Fisheries		Industrial**	
Calcium	2		mg/litre						
Magnesium	1442		"						
Sodium	2400 -	5000	"	200					
Potassium	16 -	18	"						
Carbonate	3600		"	-					
Bicarbonate	3267		"						
Sulphate	230 -	620	"	200	400 mg/l	960	1920 mg/l	680	mg/l
Chloride	82	170	"	200	600 "	177	355 "	500	"
Hardness as $CaCO_3$	5860	8130	"					850	"
Alkalinity as $CaCO_3$	6280	10600	"					500	"
Total Dissolved Solids	9900		"	500	1500 "	700	2100 "	100	"
pH	11.7	12.6	"	6	8.5	6.5	9	3.5	9.1
Turbidity	750	units		30	units				
Colour	47	"		5	50"			1200	units
% Sodium	90					60%			-

* WHO Drinking Water Standards

** Water Quality Criteria by Mckee and Wolf, 1968

From Wedderburn 1975

Nature of Contaminant

The "red mud" is a thick fluid suspension consisting of the residue of bauxite after extraction of the alumina and the residual caustic and organic substances in solution used in the Bayer extraction process. The red mud has a water content of 70-80 percent and due to the small solid particles settles to about 30 -35% clear, supernatant liquid. Jamaican red mud has physical characteristics that make its disposal difficult. One of these characteristics is its fine grain size which explains its tendency to retain so much water.

Its potential for contamination of groundwater lies in the concentration of sodium and hydroxide ions; the presence of iron oxide; and the organic substances which on decomposition impart an unpleasant smell to the waste. Tables 2 and 3 show respectively the chemical analyses of insoluble and soluble solids of Jamaica red mud.

For domestic, industrial and agricultural water, colour, taste, smell, high soda content and high pH makes the "red mud" a potential agent for degrading groundwater quality. Table 4 shows an analysis of the red mud liquid fraction plus the recommended concentration limits.

Contamination Criteria

Analyses to detect above average concentrations of the chemical constituents present plus esthetic indices such as colour, taste and smell determine the degree of contamination.

Five indices were specifically used to detect contamination.

(1) Sodium to chloride concentration ratio exceeding the maximum ratio encountered in uncontaminated groundwater in Jamaica of 1.5

(2) High sodium content. This alone is not a precise indicator as sodium chloride waters are found in the limestone aquifers as a result of saline intrusion. However, in this form of contamination high sodium concentrations are associated with high chloride concentrations, not the case with caustic contamination.

(3) Sodium to Calcium concentration ratio in excess of the ratios generally encountered in uncontaminated groundwater of 1.0.

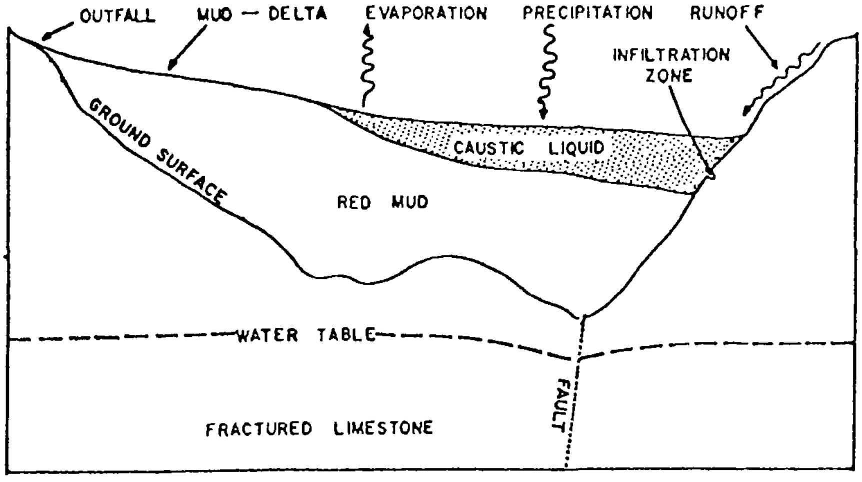
(4) High pH values in excess of 8.0 units, the maximum encountered in uncontaminated groundwater.

(5) The presence of suspended solids, red discoloration, poor smell and unpleasant taste.

Disposal Methods

The two plants, around which contamination has been detected, have similar disposal methods. The red mud waste is piped to and ponded in a natural karstic depression from where the bauxite had been previously mined. The two sites are located atop highly karstified limestone with numerous solution openings. No special precautions were taken to prevent or reduce infiltration before storage of waste began. Infiltration was seen as one way to extend the life of the disposal pond. No proper geological, hydrogeological and engineering studies were done before the selection of disposal site was made.

Figure 3: Diagrammatic Section through a Red Mud Pond



Movement of Contaminant into Aquifer

A diagramatic section through a red mud pond is shown as Figure 3. The pond is formed by a sloping, elongated, steep sided karst depression with a narrow outlet at the downstream end. Storage is improved by the construction of a rock fill dam across the outlet with increases in dam height as necessary. Limestone is exposed along the surface and at the base of the pond. The depression is naturally drained by sinkholes or other solution openings. Lateral and vertical percolation takes place through the network of fractures and solution openings typical of the karstified limestone. Leakage also occurs through the rockfill dams. With time, the accumulation of solid material will form a seal over the limestone, reducing the percolation rate. Calculations indicate that only 30 - 40% of the waste is retained in the pond.

The formation of vertices, rapid fall of mud level and overnight disappearance of mud and water, indicate rapid and high infiltration losses. The formation of mud deltas at the outfall pushes the liquid faction towards the rockfill dam where the capacity for leakage is high. Evaporation from the pond surface concentrates the pollutants in the waste. The ponds act as collectors of rainfall runoff, mixing of this water with the waste and then infiltrating into the limestone.

Contaminated Areas

Of the four sites shown on figure 1, three have been constructed atop karstic limestone and one atop alluvium. Of the three sites atop karstic limestone, contamination has been detected at two sites - site 1 and 4. Site 1 is associated with the Alcan Jamaica Company (ALJAM) Ewarton Plant while site 4 is associated with Alumina Partners (ALPART) Nain Plant.

Site 1 - ALJAM Mt. Rosser Mud Pond

This site has the largest mud disposal pond in the island with 13.17 Mm^3 of waste in storage up to September 1991, and a surface area of 40 hectares. The pond occupies a large elongated karst depression approximately 6.5 km north of the plant (figure 4). This pond known as Mt. Rosser or Schwallenburgh, had no special preparation before mud storage began. At the lower eastern end a rockfill dam of limestone blocks was constructed. Massive material losses were recorded during the early disposal period. However, it was felt that the impact on the surrounding environment would not be significant. There was also the expectation that with increasing submergence of the pond floor, the mud would form a sufficiently good seal against further effluent loss. This has not been the case and effluent losses have continued unabated. At first, attempts were made to locate and seal all the sinkholes and fractures with clay to stop the leakages. Attempts have have also been made to create beaches of tailings around the face of the dam to keep the liquid faction from seeping through but this was also unsuccessful in reducing effluent loss.

The Mt. Rosser Pond is located on a faulted anticline, the northern limb of which continues into the Moneague Sub-basin (a syncline) and the northern limb continues into the Linstead Sub-basin (also a syncline). The latter merges with a faulted anticline at Bog Walk, where impervious strata are brought to the surface and forms a barrier to groundwater flow.

The Moneague syncline forms a large depression into which groundwater discharges through springs. This depression has no surface water outlet and the spring discharge enters the aquifer via sinkholes and flows northwards to the headwaters of the rivers on the north coast of the island. Caustic contamination of groundwater has been detected at a number of springs and wells to the north and south of Mt. Rosser Pond. To the north the Rio Hoe Spring and the Walkers Wood well show evidence of caustic contamination.

The Rio Hoe Spring rises at the southern edge of the Moneague Sub-basin, flows for 2km and enters the Moneague Lake. Water from the Moneague Lake enters the aquifer via the Walton Sink and flows to the Walkers Wood well. Figure 5 shows the

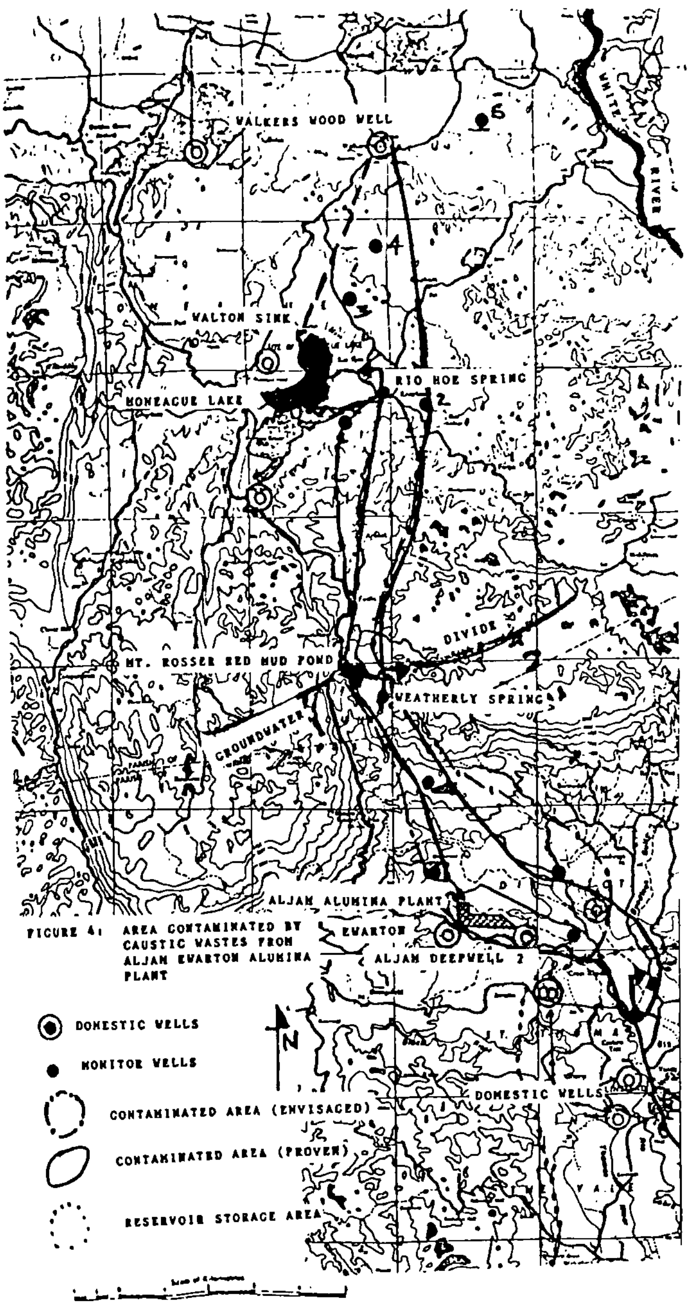


FIGURE 4: AREA CONTAMINATED BY CAUSTIC WASTES FROM ALJAM EWARTON ALUMINA PLANT

plot of sodium concentration for the period 1973-1991 for Rio Hoe. The contamination was first detected in 1971 and rose progressively to peak at over 350 mg/l sodium in 1983, and the spring which was a source of domestic water has been declared unfit for human consumption. The Moneague Lake fed by the Rio Hoe Spring is also contaminated with maximum sodium concentration of 65 mg/l (Mar. 1990).

The contamination in the Walkers Wood well was detected in 1977. The well was drilled in 1976 and the sodium concentration during the pump test was 5mg/l. The contamination has increased and monthly monitoring, which began in 1990, shows a worsening situation, (figure 6).

The elevation of the surface of the red mud waste is at an altitude of 466.3 metres while the spring rises at 305 meters and there seems to be continuity between the pond and the spring by a system of conduits above the regional water table. The spring is above the regional water table and during high rainfall the discharge becomes brown and turbid, while the water table does not exhibit this change.

The area is now being assessed and so far 5 monitoring wells varying in depth from 200-350 metres have been completed. To date, only 3 of the monitoring wells have tapped contaminated groundwater. The probable area of contamination is shown on figure 4. To the south the Weatherly Spring and Alcan's Deepwell 2 have shown evidence of caustic contamination. The plots of sodium concentration for Weatherly Spring and Deepwell 2 for 1973-1991 are shown as figures 7 and 8 respectively.

Figure 5 : SODIUM CONCENTRATION RIO HOE SPRING 1973 - 1991

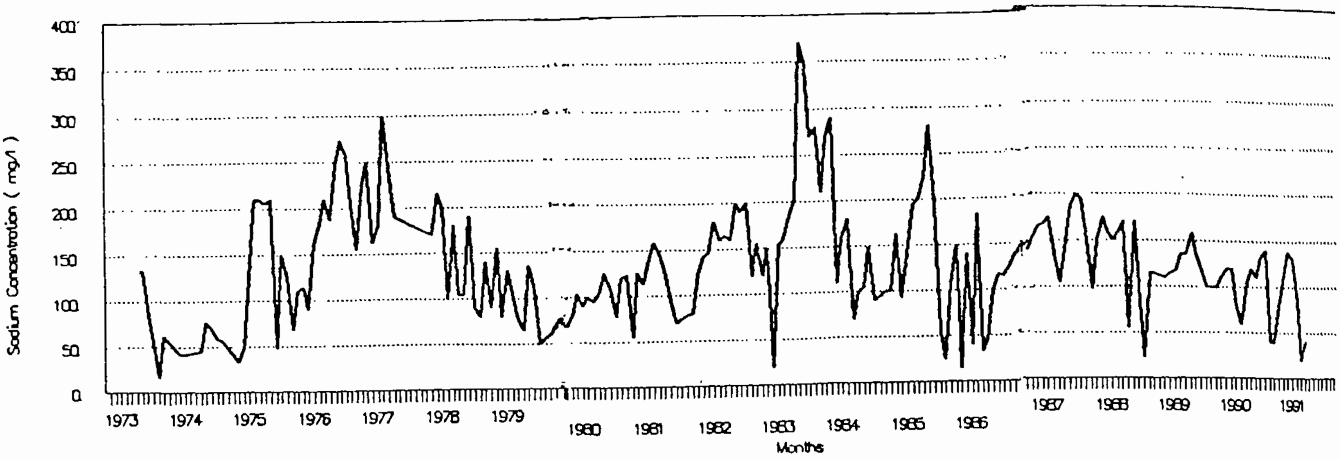
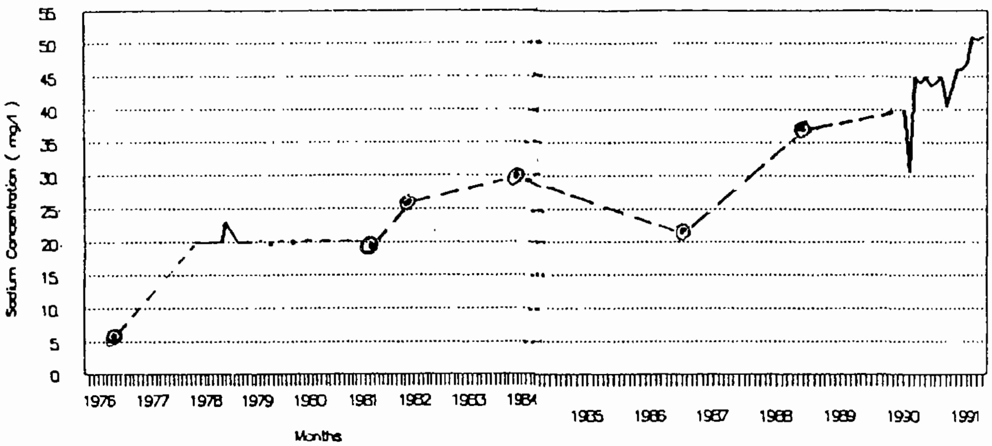


Figure 6 : SODIUM CONCENTRATION WALKER'S WOOD WELL 1976 - 1991



In 1989/90 the Underground Water Authority and ALJAM collaborated on a project to assess the level of contamination. Six monitoring wells were drilled and average sodium concentration varied from 9-30 mg/l. The results of the project indicate:

(1) the area of contamination has been better defined and has been found to be smaller than originally envisioned.

(2) two zones of contaminated groundwater separated by clean water were intercepted and confirmed the more significant role that compartmentalised flow plays in the movement of the pollutant than diffuse flow.

(3) The upper contaminated zone has been interpreted as being the direct result of the plant, its attendant dumps and other adjacent effluent generating facilities.

Figure 7 : SODIUM CONCENTRATION WEATHERLY SPRING 1973 - 1991

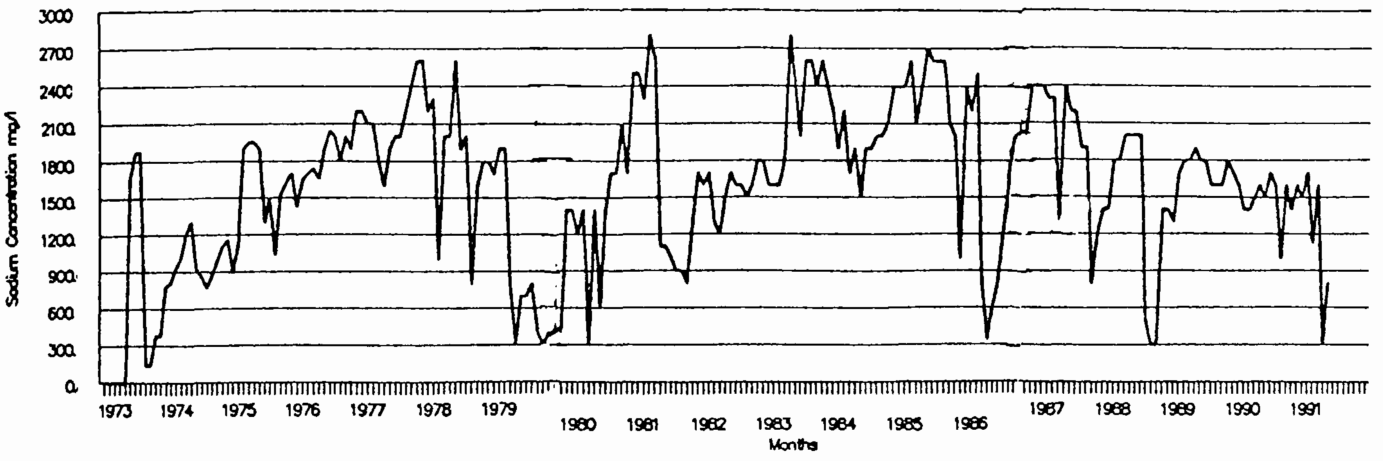
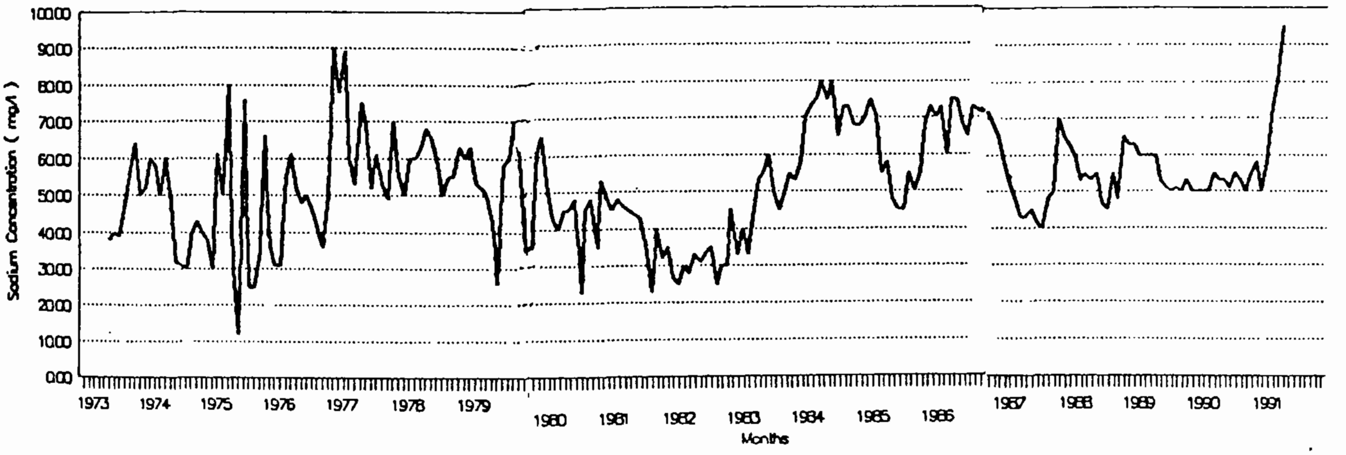


Figure 8 : SODIUM CONCENTRATION DEEPWELL 2 1973 - 1991



To the north and south the total area of the aquifers contaminated is approximately 18 km^2 (figure 4).

Site 4 Alpart -Nain Mud Pond

There are two (2) disposal ponds at this site, a south mud pond and a north clear effluent pond (figure 9) . The ponds are located on a thick sequence (> 1500m) of faulted and fractured brecciated limestone. The presence of numerous sinkholes and the absence of surface drainage indicate the potential for rapid infiltration. Thesites are located close to the groundwater divide which separates northerly flow to the Upper Morass from southerly flow to the sea. The surface area of the ponds are 90 hectares and the volume of mud in storage is estimated at 12.8 Mm^3. Red mud disposal was being restricted to the south lake but caustic enriched clear effluent is being disposed of in the north lake. The formation of tailings "deltas" were being restricted to the northern and eastern perimeters of the south lake with the resultant ponding of supernatant liquor directly against the limestone along the southern and

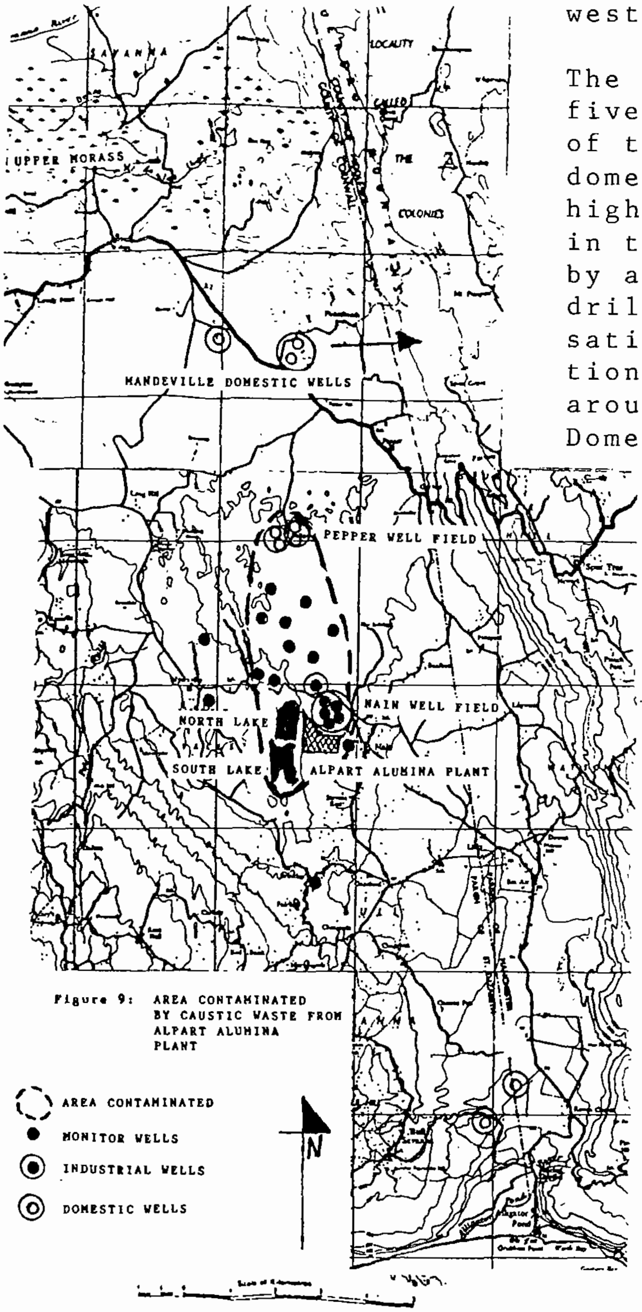


Figure 9: AREA CONTAMINATED BY CAUSTIC WASTE FROM ALPART ALUMINA PLANT

western lake perimeter.

The plant began operations in 1969 and five wells were drilled in the vicinity of the plant to supply industrial and domestic water. As early as March 1970 high sodium concentrations were noted in the well water. This was followed by a decrease in well yields and the drilling of a replacement well (#6) to satisfy demands. The sodium concentrations increased until all the wells around the plant are now contaminated. Domestic water now comes from a new well field established 6.4 km north of the plant at Pepper. The Pepper well field began operations in 1974 and has since been showing a trend to increasing sodium concentration (figures 10, 11, 12, 13).

To the south a 1990/91 investigation has indicated a low level of contamination of domestic wells at New Forest. This contamination is masked by saline intrusion into the aquifer but once the molar ratio between sodium and chloride have been reconciled, excess sodium remains in the groundwater and its source can only be the mud pond. The area of the aquifer contaminated is approximately 20 km^2 and the domestic wells that supply the large urban centre of Mandeville are now threatened.

Reduction of Contamination

The Underground Water Authority has been working with both Alpart and ALJAM to reduce groundwater contamination.

At ALJAM a new mud disposal system called "mud stacking and drying" has been implemented. This consists of thickening the red mud to 22% solids, spraying it thinly on a sloping, sealed drying bed, where it is dried by the sun. All run-off is collected at the toe of the bed and transferred to a sealed holding pond, from where it is recycled into the plant. Once the mud has dried on the bed (it does not reslurry) the bed can be cleaned and returned to use. This has allowed the Bauxite Company to dispose of less than 5% of the annual waste generated in the Mt. Rosser red mud pond, and only in the very high rainfall

Figure 10 : SODIUM CONCENTRATION ALPART 1 WELL 1975 - 1991

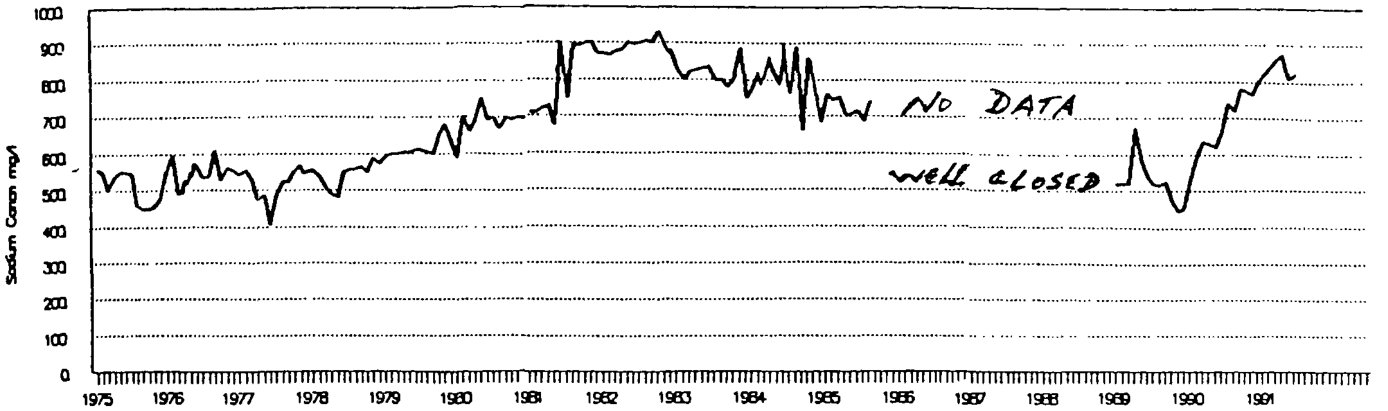


Figure 11 : SODIUM CONCENTRATION PEPPER 1 WELL 1975 - 1991

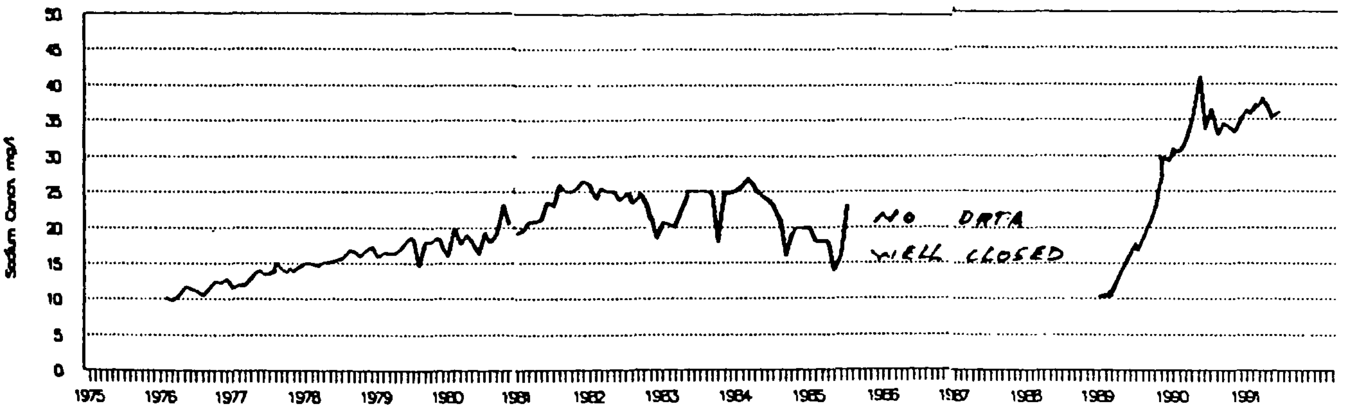


Figure 12 : ANNUAL MAXIMUM Na/Cl RATIO ALPART 1 1975-1991

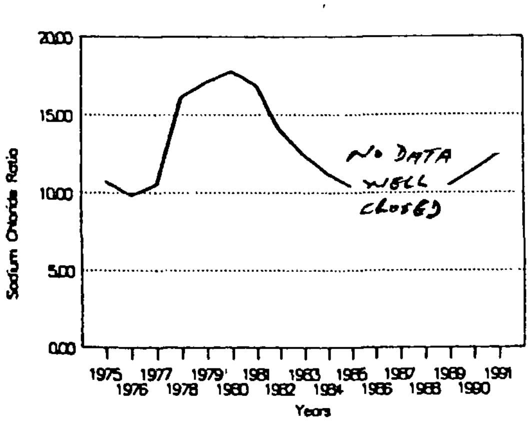
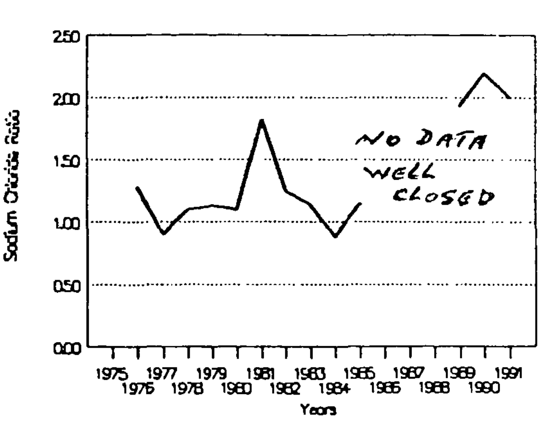


Figure 13 : ANNUAL MAXIMUM Na/Cl RATIO PEPPER 1 1975-1991



periods when the drying cycle becomes longer. Alpart is now investigating the suitability of this method of disposal for implementation at its plant site.

Summary and Conclusions

The red mud wastes produced from the bauxite alumina operations contain sufficiently high concentrations of pollutants to contaminate the groundwater, making it unusable. The method of disposal of the waste utilizing natural or mined out depressions in the limestone creates a potential hazard in view of the high degree of karstification of the aquifer. Contamination of the groundwater in the vicinity of the two sites have been proven. The solutions to reducing the contamination lie in the sealing of the disposal pond and possible total impoundment of the effluent. Pond management needs to be improved to minimize the possibilities and effects of further contamination.

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Question

Are there any springs near the plants that are used as water supplies? If so, are they hydraulically connected to the plant?

Answer

Yes, there are 4 springs located near to the plant that can be used for water supplies. One spring is located to the north, and 3 are located to the south of the plant (see figure 4).

Of the 4 springs, 3 are hydraulically linked to the mud disposal pond, and show signs of contamination.

These are:

(a) North - Rio Hoe Spring
(b) South - Weatherly Spring
Cashew Tree Spring

The fourth spring is not hydraulically linked, and is not contaminated. It is used as a source of domestic water for a small village in the hills above the plant.

The Interaction of Flow Mechanics and Aqueous Chemistry

in a Texas Hill Country Grotto

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Abstract

The physical characteristics of flow, or *flow mechanics*, through a carbonate terrane affect the chemical signatures of groundwater inputs to Hamilton Pool, a grotto in the Central Texas Hill Country portion of the Trinity Aquifer. Because flow mechanics influence geochemical processes, their identification is necessary to protect water quality at this unique site, which is used for both recreation and habitat preservation.

During baseflow, two groundwater sources supply all the water in the pool: 1) small perennial springs half a kilometer upstream from the pool are the headwaters for Hamilton Creek, which enters the pool via a 25-meter waterfall (0.1-0.3 ft^3/s), and 2) stalactites hanging from the ceiling of a large overhanging limestone ledge drip water directly into Hamilton Pool (0.05 ft^3/s). The two sources exhibit differences in pH, calcium concentration, Ca/Mg ratio, and temporal variation in chemistry. The chemistry of the drip water is affected by degasing of carbon dioxide and equilibration with ambient temperature upon contact with the atmosphere. Backmodeling of these environmental processes with PHREEQE resulted in a subsurface chemistry of the drip water very similar to that of the spring water. Analysis of conductivity variation in response to precipitation revealed that the chemistry of the upstream springs is controlled by conduit flow, while the ceiling water is controlled by diffuse flow.

These results suggest that a) both groundwater inputs to Hamilton Pool are from the same recharge source, and b) the conduit-controlled upper springs render Hamilton Pool vulnerable to contamination should the catchment area be urbanized.

Introduction

Carbonate aquifers are distinguished from other groundwater media by their physical flow characteristics. Flow rates at different points within a single carbonate aquifer can vary many orders of magnitude, from centimeters per day to kilometers per day. The mechanics controlling flow rate influence both the chemistry of the water and the susceptibility of the water to contamination. This paper highlights the findings of a more extensive study of geochemical response to flow mechanism in a Texas Hill Country grotto (Mahler, 1991).

Diffuse flow and conduit flow represent opposite extremes of carbonate flow. Diffuse flow is similar to flow through a granular, porous media; water flows through extremely tiny joints, partings, bedding planes, and fractures, the diameters of which are on the scale of millimeters or less. Conduit flow, in contrast, involves flow through integrated, solutionally-developed conduit systems with diameters on the scale of centimeters to meters (e.g., Shuster and White, 1971).

The risk of contamination of conduit-controlled water sources is high. Conduits behave as pipes rather than as filters; water moves through them with high velocities and short residence times, and without undergoing dilution. The processes that normally cleanse groundwater (e.g. filtration by soils, adsorption of contaminants by mineral grains, dilution and dispersion through large volumes of water, and decay and die-off during long residence times) often do not affect conduit flow.

Springs controlled by the two different flow mechanisms can be distinguished by their chemistry. Shuster and White (1971) found that diffuse flow springs showed much less variation in hardness than conduit flow springs. Similarly, Jacobson and Langmuir (1974) found that diffuse flow springs showed less variation in solute amounts than conduit flow springs. They also found that after a storm, conductance remained the same or increased slightly for diffuse flow springs due to salts and solutes swept in from soil, while conductance decreased in conduit flow springs due to dilution with low conductivity meteoric water. They therefore suggested using specific conductance as an indicator variable of subsurface flow mechanism.

The objective of the investigation presented here is to characterize the aqueous geochemistry of inflows to Hamilton Pool, Travis County, Texas, to determine the flow mechanism controlling each source, and to investigate each inflow's genetic relationship to the other baseflow sources.

The Study Site. Hamilton Pool is a pristine waterfall-fed grotto in rural western Travis County, Texas, approximately 20 miles southwest of the city of Austin. The pool is the focal point of Hamilton Pool Preserve, a county-operated facility. Water quality at the pool is utmost important both for recreational use and species habitat. Although current land-use in the Hamilton Creek watershed is primarily agricultural, commercial and residential development are rapidly expanding westward from the Austin area.

The Hamilton Creek watershed is located in the rocks of the Lower Cretaceous Trinity Division. Five outcropping formations of the Trinity Division can be viewed at Hamilton Pool Preserve. Sycamore Sand outcrops in the bed of Hamilton Creek from the Pedernales River to approximately one kilometer upstream, where it slopes into Hammett Shale. The shale forms the eroded walls of the cliff surrounding the east side of Hamilton Pool, and Cow Creek Limestone forms the overhanging ledge. The rolling grasslands surrounding Hamilton Pool are on Hensel Sand, which outcrops along the creek bed and smaller drainages. Glen Rose Limestone outcrops in the hills east of the pool, and is the most areally extensive formation cropping out within the watershed (Figures 1 and 2).

There are two principal inflows to Hamilton Pool. The main source is Hamilton Creek, which empties into the pool via a 25-meter waterfall. During baseflow, the sole source of this water is a group of small springs a few hundred meters upstream from the waterfall. A second inflow consists of groundwater dripping from a large travertine formation ("Drippy Rock") and stalactites which line the ceiling of a cave-like structure which overhangs the pool. Both these inflows issue from the Cow Creek Limestone. During baseflow, Drippy Rock and stalactite flow total approximately 0.035 ft^3/s, while flow from the upper springs varies from 0.1 to 0.5 ft^3/s.

Approach

The aqueous chemistry within the Hamilton Pool Preserve was examined to determine the water source, flow mechanism, and subsurface processes. Water samples were collected weekly for six months from the four springs comprising the Upper Springs site, from Drippy Rock, and from Hamilton Pool itself. Field measurements of pH, temperature, alkalinity, and conductivity were recorded, and analyses of major anions and cations performed in the laboratory. The chemistries of the four upper springs, all located within three meters of one another, were compared to see if one spring could be chosen as representative of all four. Because all four sites displayed similar solute concentrations and temporal variation, it was decided to use analyses of East Spring as indicative of the chemistries of all the upper springs.

Mineralogy of solid phase samples was analyzed to estimate the contribution from each formation to aqueous chemistry. Fresh samples of the Glen Rose Limestone, the Hensell Sand, the Cow Creek Limestone, and the Hammett Shale were collected in the field and analyzed for major cations.

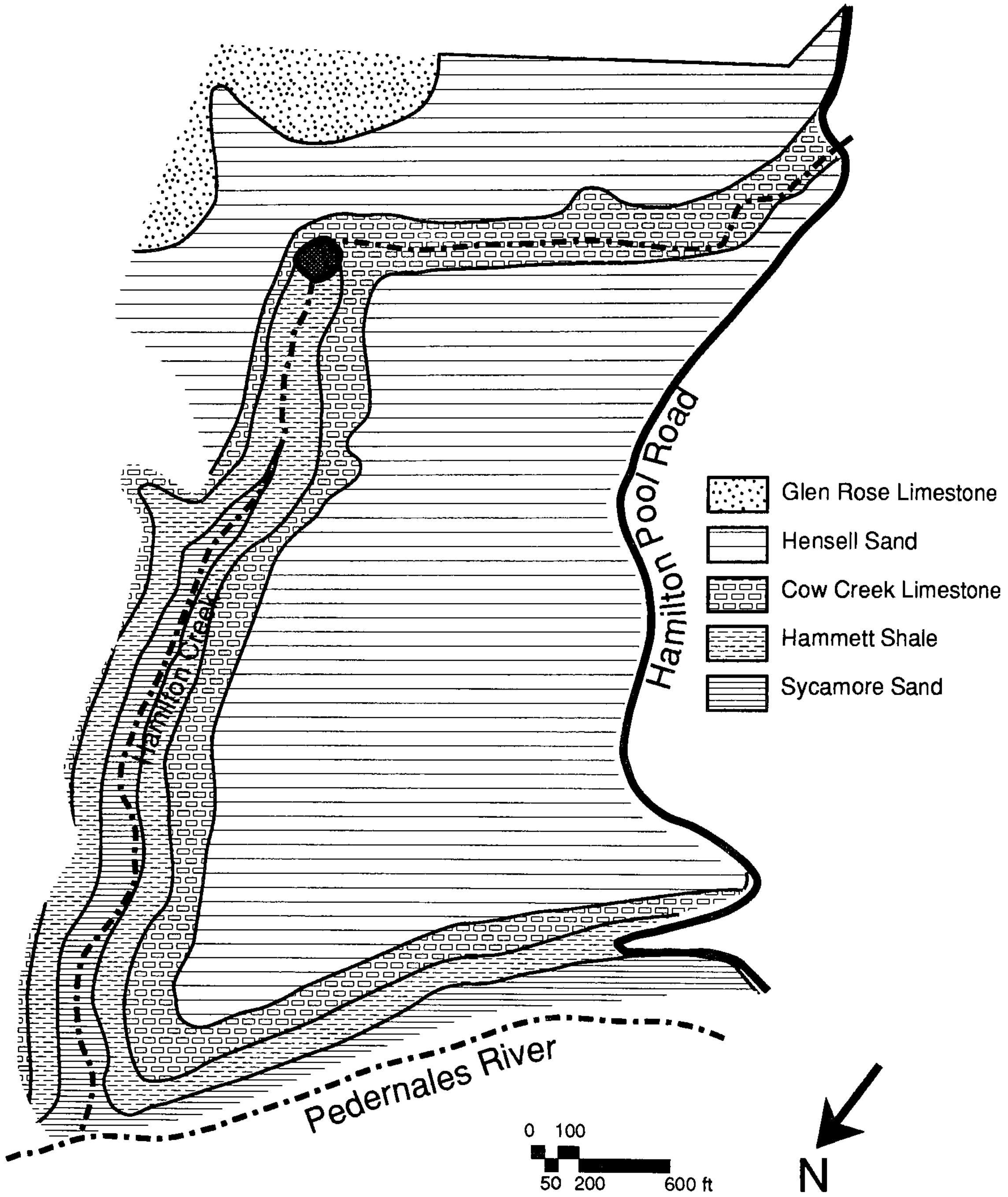

Figure 1. Plan view of Hamilton Pool Preserve.

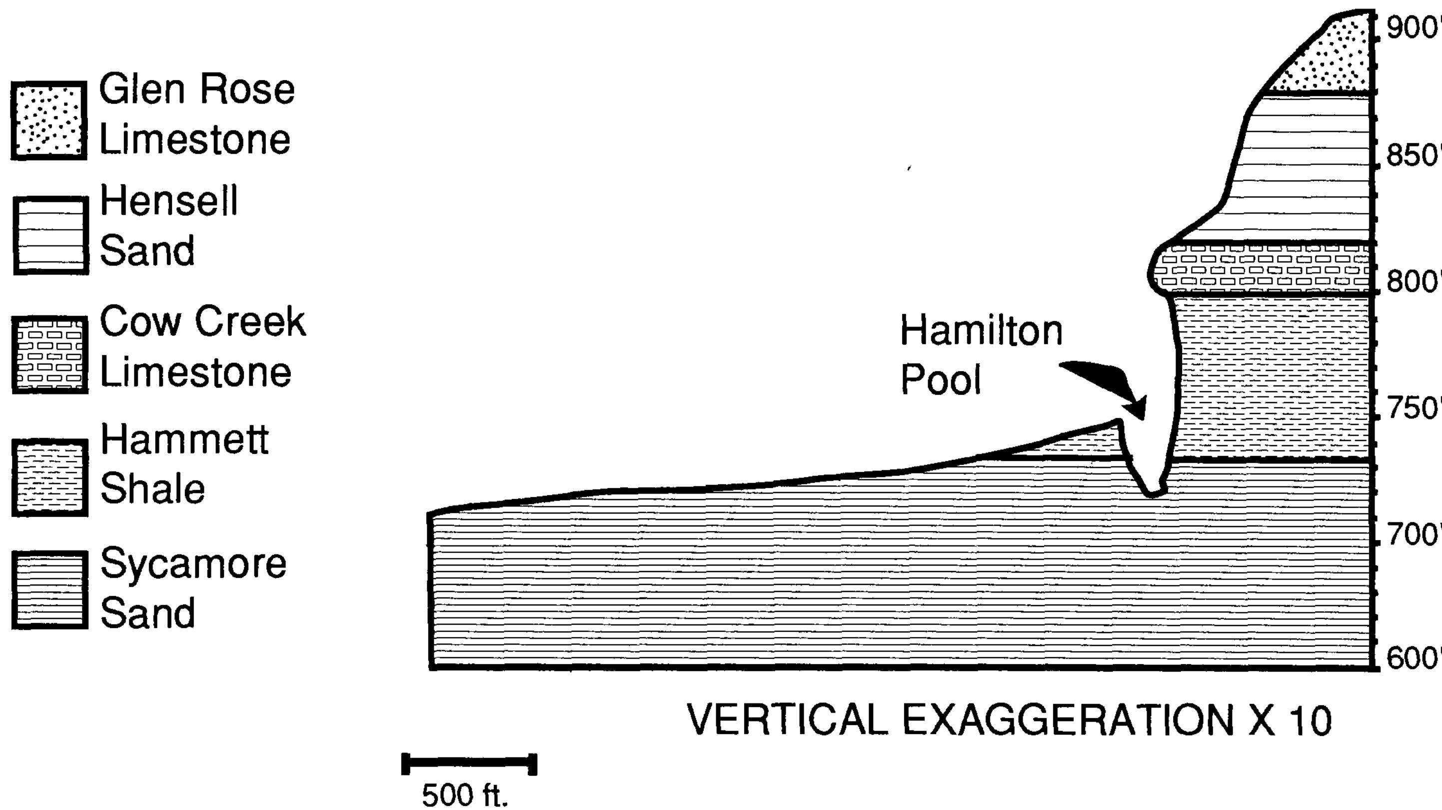

Figure 2. Cross Section of through Hamilton Pool Preserve.

Results

Environmental processes and flow mechanics cause differences in aqueous chemistry at Drippy Rock and East Spring. The formation of travertine where Drippy Rock water issued from the subsurface indicated that the drip water became supersaturated with respect to aragonite upon contact with the atmosphere. Additional carbon dioxide may have been lost to the atmosphere from the collection vessel in the amount of time that it took to accumulate enough drip water for a sample, as suggested by White (1988). Saturation indices for aragonite, calculated with WATEQF (Plummer et al., 1976), an equilibrium speciation program, showed that Drippy Rock water was indeed supersaturated with respect to aragonite, while East Spring was slightly undersaturated. Analyses of Drippy Rock revealed that Drippy rock was depleted in calcium ion and had increased pH relative to East Spring, as expected. In order to compare the subsurface chemistries of the two sources, PHREEQE (Parkhurst et al., 1980) was used to back-model the environmental processes affecting the drip water. Loss of carbon dioxide was reversed by reacting Drippy Rock water with a fixed pCO2 equal to that of baseflow at East Spring (-1.70), the effects of ambient temperature were reversed by holding temperature fixed at the baseflow temperature of East Spring (23°C), and the water was held in equilibrium with calcite. Figure 3 compares the chemical analyses for Drippy Rock and East Spring with the results of the simulated subsurface Drippy Rock water. Modeling Drippy Rock water in the subsurface eliminated most of the difference in pH and calcium concentration for the two waters, but had no effect on magnesium concentrations.

Drippy Rock and East Spring had contrasting responses to rainfall. Conductivity and rainfall for a three-month period are shown in Figure 4. Conductivity at East Spring drops after a major rain, while the conductivity at Drippy Rock remains the same or increases slightly.

In general, the back-modeled Drippy Rock water and East Spring water had similar overall chemistries. Both were both calcium/magnesium bicarbonate waters. They contained similar levels of magnesium, chlorine, sulfate, and silica, and had very low levels of iron, phosphate, boron, and potassium. These similarities suggest that the source of these waters was similar. To investigate possible rock-water interactions, solid samples of Glen Rose Limestone, Hensell Sand, Cow Creek Limestone, and Hammett Shale were analyzed for major cations. The results are displayed in Table 1, which shows that the Cow Creek Limestone did not have any detectable magnesium, in contrast to the other formations, which contained magnesium levels similar within an order of magnitude. All four formations had similar proportions of calcium and strontium. These results indicate that 1) the Cow Creek Limestone has been entirely recrystallized into a pure calcite limestone, and 2) that the magnesium in water samples issuing from the Cow Creek

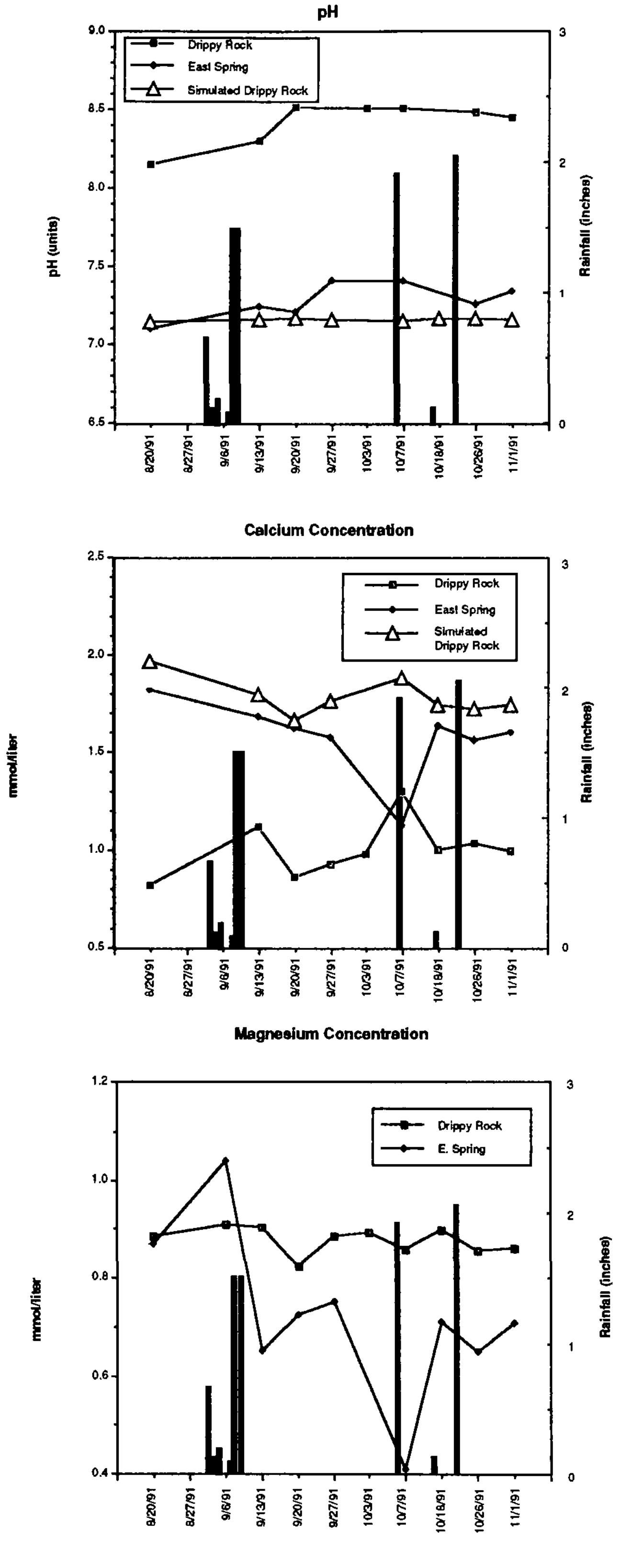

Figure 3. Temporal variation in pH, calcium, and magnesium versus rainfall. Simulated water is fixed at pCO2 = -1.70, T = 23.0° C, and is in equilibrium with calcite.

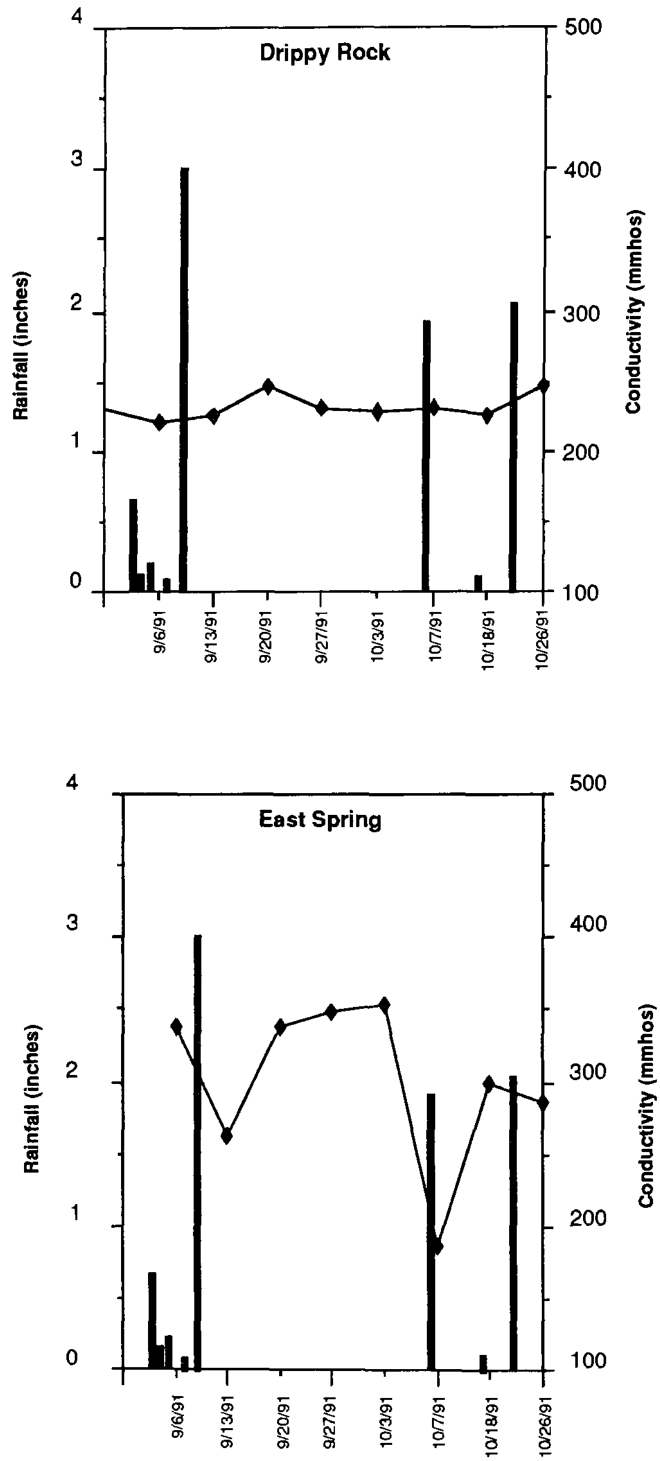

Figure 4. Conductivity versus rainfall at Drippy Rock and East Spring.

Table 1. Cation analyses for solid phase samples.

	Ca	Mg	Sr
Hammett Shale 1	0.324	0.043	0.00038
Hammett Shale 2	0.442	0.008	0.00033
Hammett Shale 3	0.284	0.012	0.00051
Hammett Shale 4	0.476	0.003	0.00056
Cow Creek Limestone 1	0.443	0.000	0.00063
Cow Creek Limestone 2	0.536	0.000	0.00028
Glen Rose Limestone	0.393	0.028	0.00047
Hensell Sand	0.245	0.075	0.00019

Units given in gram constituent per gram sample.

Limestone must be derived from other formations, most likely the Glen Rose Limestone, which covers the majority of the catchment area. These results suggest that water at both East Spring and Drippy Rock originate from recharge water falling on the Glen Rose Limestone.

Discussion

The results of the chemical analyses suggest that water at Drippy Rock is chemically similar to water from the upper springs, but that each is controlled by contrasting flow mechanics.

The chemical behavior of East Spring suggests that it is controlled by conduit flow. The decrease in conductivity and calcium concentrations after rainfall at this site indicates dilution of baseflow by low-conductance meteoric water. Once baseflow water has been diluted by rainfall in a conduit system, it moves swiftly through the aquifer and emerges from a spring before it has reached equilibrium with the surrounding rock matrix (Jacobson and Langmuir, 1974). An interesting physical manifestation of conduit flow at this site was the build-up of small mounds of sand and gravel at the mouths of the upper springs after a particularly heavy rain; the sediment had evidently moved through the conduits as bedload. The conduit nature of flow at this site greatly increases its susceptibility to contamination, as water which enters the conduits undergoes minimal filtration and adsorption of contaminants.

In contrast, diffuse flow mechanics control the water chemistry at Drippy Rock, as evidenced by the lack of variation in conductivity. The slow rate of diffuse flow allows water to approach equilibrium with the rock matrix, and water at this site remains unaffected by dilution from rainfall. In diffuse flow systems, rainfall may also flush soil water with high carbon dioxide concentrations into the rock matrix. Such an increase in carbon dioxide could cause the increase in dissolved calcium concentrations observed at Drippy Rock after rainfall.

Conclusion

The two groundwater inflows to Hamilton Pool are genetically very similar. The contrasting chemistries of these inflows to Hamilton Pool are caused by physical differences in flow as the water comes into contact with the atmosphere. Drip water from Drippy Rock undergoes much more degasing upon atmospheric contact than spring water at East Spring; it also responds more quickly to ambient temperature. As a result, Drippy Rock water undergoes an increase in pH accompanied by a reduction in calcium and bicarbonate concentrations as it emerges from the subsurface. After removing the effect of these processes through geochemical modeling, the simulated Drippy Rock subsurface water and the East Spring water both appear to originate as recharge through the overlying Glen Rose Limestone.

Contrasting response to rainfall indicates that the two sources are controlled by contrasting flow mechanics. East Spring is controlled by conduit flow. As a result, after rainfall East Spring water shows an increase in conductivity and a decrease in concentrations of several ionic constituents including calcium. Because conduit-controlled springs are susceptible to contamination, Hamilton Pool is vulnerable to water quality degradation from this source. In contrast, the second source (Drippy Rock) is controlled by diffuse flow. After rainfall, water at Drippy Rock undergoes a slight increase in conductivity and an increase in calcium concentration due to flushing of soil carbon dioxide into the water.

Acknowledgements

Very special thanks go to Travis County Parks and Recreation staff at Hamilton Pool Preserve, particularly Terri Siegenthaler and Dan Chapman. Additional thanks to Roger Lee, Raymond Slade, and Colleen Stapleton.

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Biographical Sketch

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The Interaction of Flow Mechanics and Aqueous chemistry in a Texas Hill Country Grotto.

By Barbara Mahler and Philip Bennett

Q: Do either or both of the chemistries of the diffuse and conduit-dominated flow regimes vary seasonally?

A: Water samples were collected from June through October, 1990, a sampling period insufficient to verify existence or lack of seasonal variation in chemistry. During this period no seasonal variation was detected in the conduit-controlled regime. In contrast, the samples collected at the diffuse-controlled site had calcium concentrations which were inversely related to ambient temperature, due to rapid equilibration of the drip water with surface conditions. However, we assume that within the subsurface the diffuse-controlled regime would display the same sensitivity to seasonal variation as the conduit-controlled regime.

The sampling period was necessarily limited due to the scope of the project (field work for a M.A. thesis). However, regular sampling over a two year period may reveal differences in seasonal variation between the two regimes, which would provide more information about the relation between the two water sources.

THE ORONOCO LANDFILL DYE TRACE III: RESULTS FROM A SUPERFUND REMEDIAL INVESTIGATION IN A GLACIATED, DIFFUSE-FLOW KARST.

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ABSTRACT

The Oronoco Landfill in Olmsted County, Minn., a Superfund site 10 miles north of Rochester, is built on the Ordovician Prairie du Chien Formation, and important regional karst aquifer. As part of the site's R.I., a dye trace was conducted to determine: 1) the local directions(s) and speed of ground water flow, 2) if an existing monitoring well network effectively intercepted ground water flow beneath the site, and 3) which local private wells were most likely to be affected by potential ground water contamination attributable to the landfill.

Samples were collected from 11 on-site monitoring wells, 12 springs, and 201 private wells. Continuous pulses of dye were detected in 6 on-site monitoring wells, and 2 private wells located 2.2 and 2.9 km northeast of the landfill. The continuous pulses of dye detected in the wells have persisted for up to two years. The concentration of dye in the wells periodically increases after major recharge events. This karst is a diffuse-flow system with ground water flow to the northeast at velocities of the km/yr. The monitoring wells

appear to effectively intercept ground water flowing under the site. Conventional quarterly sampling will detect the largest, chronic hypothetical landfill releases but may miss short-term, acute pulses of pollutants. The continued presence of dye in the system has proven to be a useful guide during the installation of additional monitoring wells at the site.

Introduction

The Olmsted County Sanitary Landfill is located approximately 10 miles north of Rochester , Minnesota (Figure 1) and was purchased by the City of Rochester in 1969. A permit to construct a landfill on the site was granted by the Minnesota Pollution Control Agency (MPCA) in 1970. Site development began in 1972. The City of Rochester operated the site until 1983, when the City and Olmsted County agreed to transfer the permit to Olmsted County.

The landfill received waste from local refuse collection haulers serving residences throughout Olmsted and adjacent counties. Local commercial establishments, institutions, and industries also used the site. In addition to the usual refuse from these generators, the site also received incinerator ash, utility ash, wastewater treatment plant sludge and sludge ash, miscellaneous industrial process sludges, and debris resulting from a 1978 flood. In March 1987, the landfill was closed to municipal solid waste, but continues receiving demolition, coal ash, and asbestos waste.

In 1983, analysis of water samples from monitoring wells at the site indicated the presence of VOCs. As a result, MPCA and EPA began Superfund actions at the site in 1986. Subsequently MPCA and EPA agreed to establish MPCA as the lead agency for remedial action. The MPCA negotiated a Response Order by Consent with the City of Rochester and Olmsted County which became effective in December 1989.

As part of the site's Remedial Investigation (R.I.), the Olmsted County Landfill Dye Trace Study was conducted to determine: 1) the local directions(s) and velocity of ground water flow, 2) if an existing monitoring well network effectively intercepted ground water flow beneath the site, and 3) which local private wells were most likely to be affected by potential ground water contamination attributable to the landfill. The Study was initiated in April, 1989 and continues at the present time (October, 1991). This report is based primarily on data gathered between April, 1989 and September, 1990 and is excerpted from Alexander et al. (1991). The Study was divided into two phases. Phase 1 extended from the initiation of the project through March 31, 1990. Phase 2 extended from April 1, 1990 to the present.

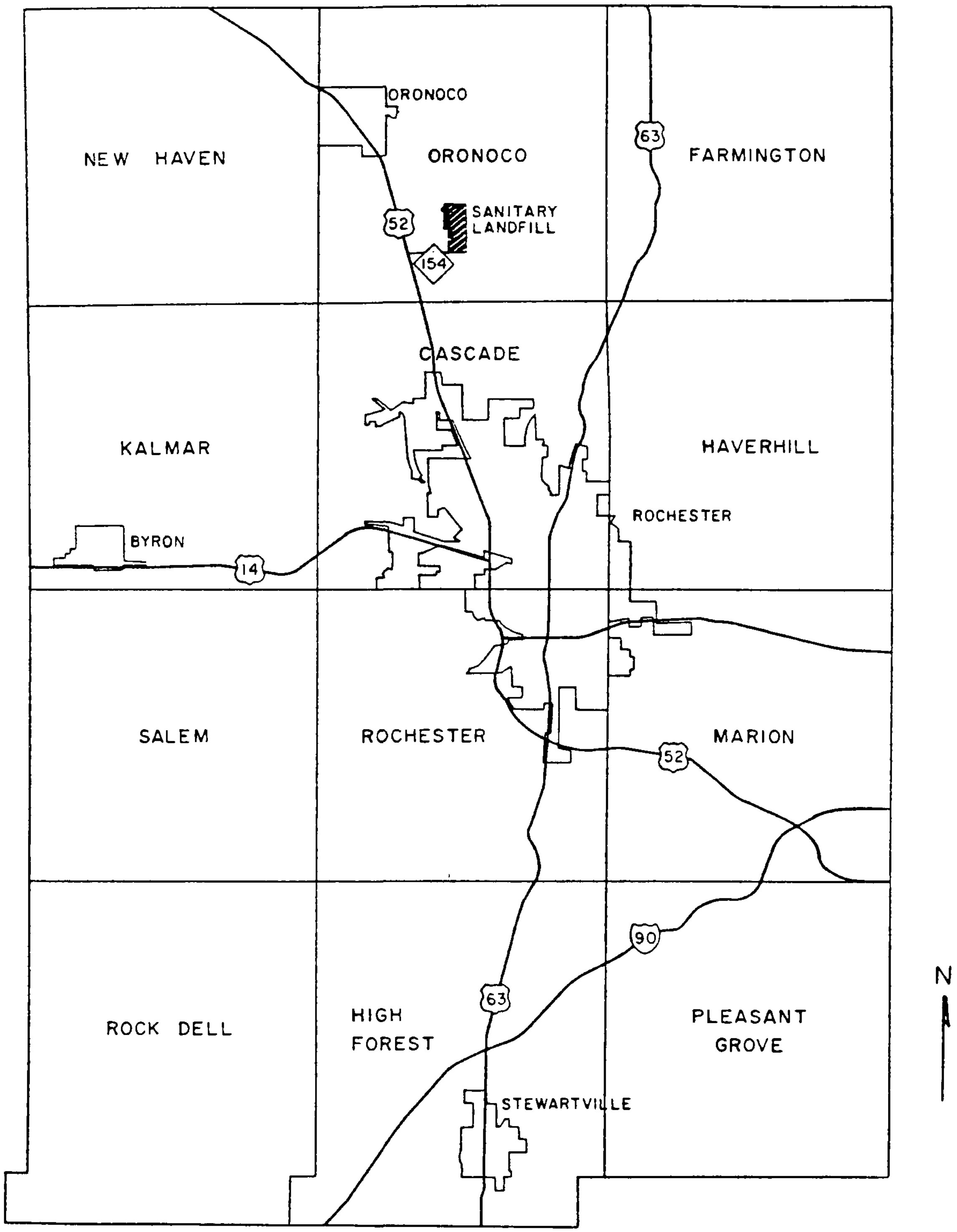

Figure 1. Map of Olmsted County Minnesota with the townships, cities, major highways and County Sanitary Landfill near Oronoco shown.

Site Geology and Hydrology

In the vicinity of the site, glacial overburden generally is less than 100 feet thick. Within the site, the overburden is 0 to 60 feet thick and includes sands and gravels, clay and silt. Clay and silt units predominate in the southern part of the site. Sand and gravel underlies the central and northern part of the site, including the waste cells.

The first bedrock encountered under the site is the Prairie du Chien Group. It is up to 250 feet thick and is predominantly dolomitic limestone. It is fractured, jointed, and contains numerous karst features. Underlying the Prairie du Chien Group is the Jordan Sandstone (Olson, 1988a, 1988b).

The water table is encountered in the Prairie du Chien Group (Kanivetsky, 1988). In the vicinity of the waste cells, groundwater monitoring wells completed at the water table have static water levels at depths of 80 to 115 feet below ground surface. Ground water flow in the vicinity is generally to the east or northeast. Residential wells in the vicinity draw water from both the Prairie du Chien and the Jordan aquifers.

Dye Trace Methods

Groundwater flow velocity and direction in limestone is aquifers difficult to predict because the exact nature of the fractures, joints and solution cavities in the limestone is unknown. Groundwater flow in these conditions can be studied by introducing dye at one or more locations and then monitoring the groundwater at many locations to determine the speed and direction of groundwater and dye movement.

Rhodamine WT dye was selected for use in this study principally because of its ease and economy of analysis. Dye was introduced into carbonate bedrock in a quarry west of the landfill in May 1989 (Phase 1). A second introduction of dye was made in March 1990 in a sinkhole located east of the site (Phase 2).

A study area approximately 1.5 miles square was established. Groundwater samples were taken daily from approximately 100 residential wells and from the landfill monitoring wells. Springs and sites along the Zumbro River were sampled less frequently.

The residential wells were distributed throughout the study area. They included up-gradient wells, generally west and south of the landfill site, and wells north and east of the Zumbro River, a hydrogeologic boundary. Residential well samples were taken by the residents.

Springs were sampled because they were considered to be potential discharge points for dye. The Zumbro River was sampled in an effort to detect dye which might seep directly into the river. These samples were taken by County staff.

Results and Conclusions

Rhodamine WT dye reached the first monitoring well less than 24 hours after introduction (see Figure 2). Dye reaching a total of six monitoring wells in the following weeks and months documented a complex pattern of groundwater flow roughly west to east under the waste cells. The continued presence of dye in the monitoring wells documents that they are adequately sampling the local groundwater flow. All of the positive monitoring wells are in the Prairie du Chien aquifer.

Rhodamine WT dye reached private well 151 north-northeast of the landfill 77 days after introduction (see Figure 3). That well is 1.2 miles northeast of the dye introduction point. The minimum speed the leading edge of the dye traveled to reach well 151 is 5.7 miles per year. Dye continues to emerge from that well in October 1991. Dye reached private well 108, 1.5 miles northeast of the introduction point, in 207 days. Dye also continues to emerge from that well. Both wells are older Prairie du Chien wells. These two positive wells document a groundwater flow path from the landfill to the north-northeast in the Prairie du Chien aquifer.

Rhodamine WT from the first dye introduction has not been confirmed at any of the other original monitoring wells, residential wells, springs, or river stations. However, trace levels of Rhodamine have appeared at spring 412 (see Figure 4). Rhodamine WT from the first dye introduction has been detected in several of the new monitoring wells drilled into the Phase 1 dye plume. Rhodamine WT from the second introduction of dye was detected during the installation of a new monitoring well nest in June 1991. No other detections from Phase 2 dye have occurred.

The dye pulses in five of the six positive monitoring wells respond rapidly to recharge events producing large fluctuations in the dye concentrations. The monitoring well breakthrough curves illustrate the complex nature of the groundwater system beneath the landfill. The Prairie du Chien aquifer under and surrounding the landfill has flow characteristics that are intermediate between those of mature, conduit-flow karst aquifers and porous-media aquifers.

In addition to the continuous dye pulses discussed above, a low level contamination problem complicated and increased the cost of the Study. During the summer of 1989, dye detections

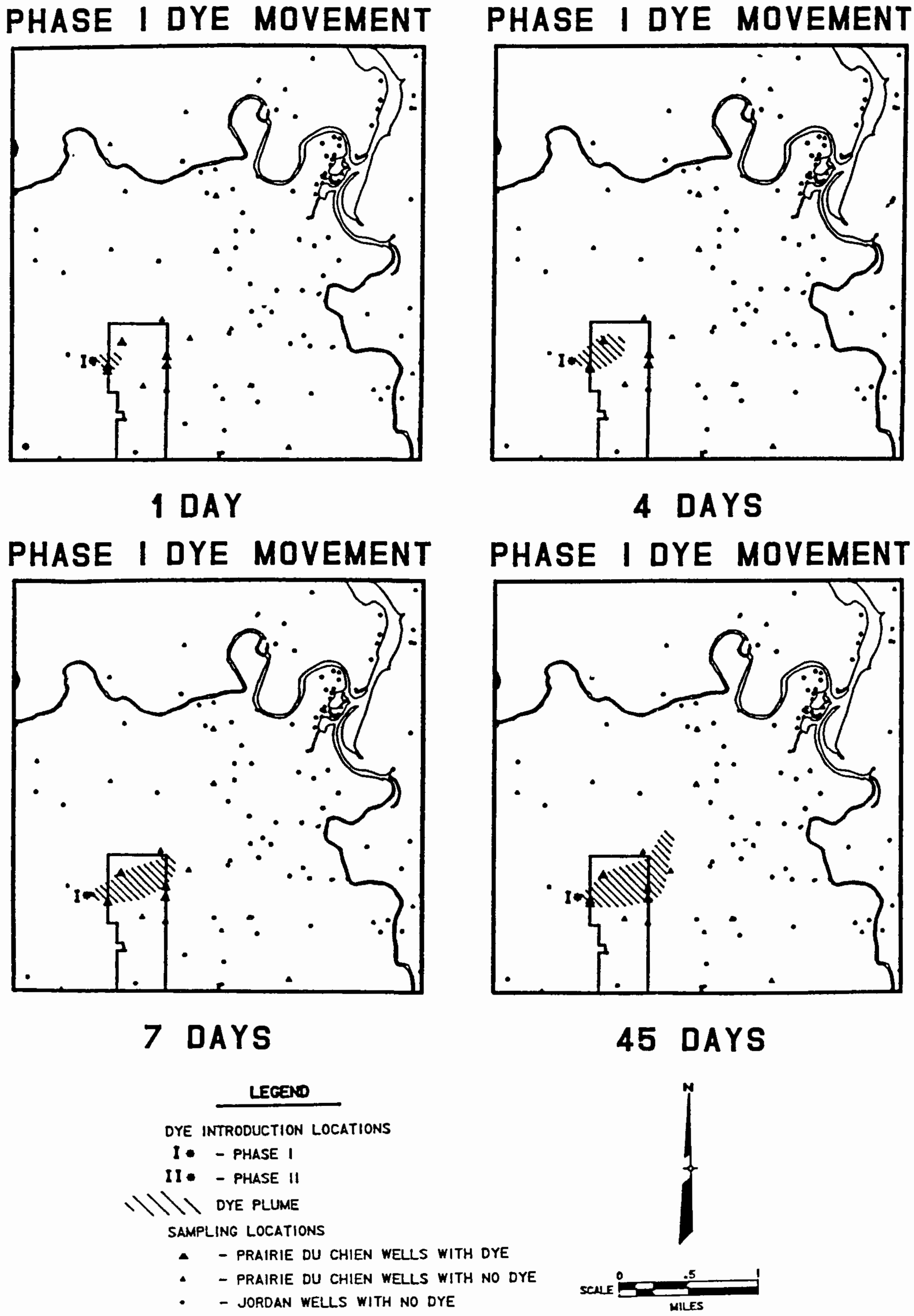

Figure 2. Phase 1 dye movement during the first 45 days.

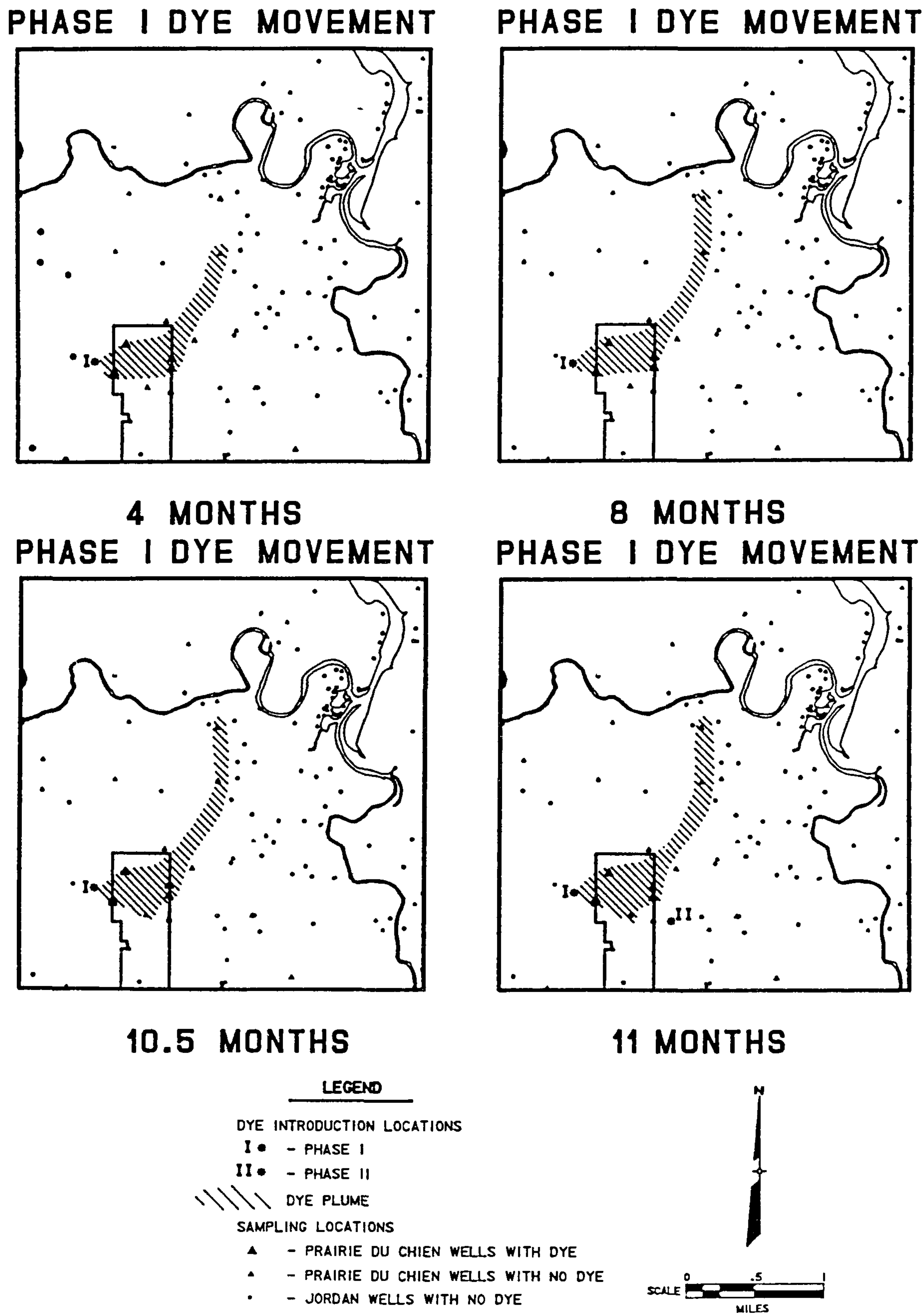

Figure 3. Phase 1 dye movement, four to eleven months.

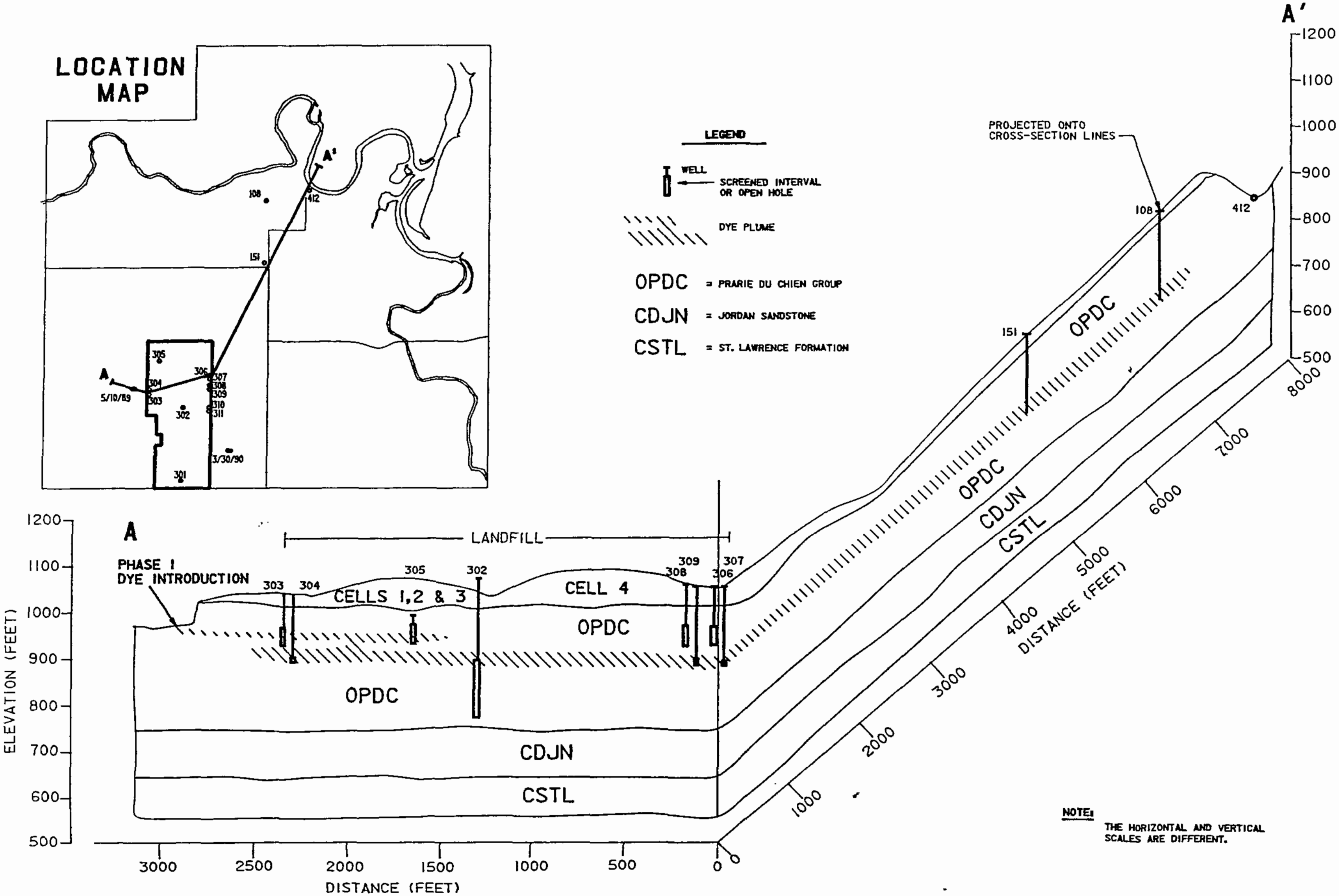

Figure 4. Cross section of dye movement.

began to occur erratically at isolated, random locations in the residential wells. The number of residential wells in the study was expanded. Ultimately the number exceeded 200. These scattered, short-term detections were eventually shown to be a random phenomenon unrelated to the Rhodamine WT used in the study.

The results of an anion survey conducted as part of the Study indicates that the Jordan aquifer in the Study Area is hydrogeologically isolated from the overlying Prairie du Chien aquifer. In many parts of southeastern Minnesota these two aquifers behave as single aquifer. In the Study Area, the Prairie du Chien aquifer contains elevated chloride and nitrate contents presumably from a variety of surface sources. The Jordan aquifer, in contrast, shows very little evidence of surface impact. The pattern of dye movement revealed by the positive wells is consistent with the chemical data and with Study Area scale potentiometric mapping.

Recommendations

Based on dye trace study results through September 1990, additional geotechnical investigations were recommended for the site. Specific recommendations included: 1) slug and pump tests on the existing monitoring wells, 2) investigation of the source of the vertical head gradients at the site, 3) gamma logging of the monitoring wells and selected nearby private wells, and 4) investigation of the existence and size of rapid pulses of contaminants in the monitoring wells.
It was also recommended that all of the existing monitoring wells should be maintained as part of the expanded Environmental Monitoring System and private wells 151 and 108 should be added to the array. There is a danger that access to samples from wells 151 and 108 could be lost as their owners seek alternative water supplies. The ability to sample wells 151 and 108 should be maintained.

Several new monitoring wells were recommended. Several shallow water-table, mid-level Prairie du Chien and at least one Jordan monitoring well were needed in the northeastern part of the landfill site. Mid-level Prairie du Chien monitoring wells and another Jordan monitoring well were needed in the area between the landfill site and private well 151.

Finally, if the anion evidence for leachate pulses in the monitoring wells is confirmed, the ability of routine quarterly sampling to adequately define the contamination potential will need to be examined.

Remedial Investigation

During the summer of 1991, work was conducted on the site as part of the Remedial Investigation (RI) phase of the Superfund process. The second stage of field work for the R.I. is currently underway. Seven new monitoring wells have been installed and geophysical logging was conducted on a subset of the new and old monitoring wells and residential wells 151 and 108. A "maintenance mode" of sampling for dye has been maintained throughout the R.I. The presence of dye in the aquifer was useful during the installation of the new monitoring wells to confirm correct siting of the wells and to assist in selecting the depth for screen placement.

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James F. Quinlan is an independent consultant and former research geologist for the National Park Service at Mammoth Cave Kentucky. He has a B.S. in geology (1959) from Virginia Polytechnic Institute and a Ph.D. in geology (1978) from the University of Texas at Austin. His field experience includes: 32 years of research and observations in karst terranes of more than 20 countries and 25 states; more than 500 dye-traces in the Mammoth Cave region of Kentucky and in various states; the environmental applications of dye-tracing; evaluation of waste-disposal sites in limestone terranes; design of ground water monitoring networks; and analysis and remediation of sinkhole development. He and

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Questions

1. There is an explicit vertical path followed by the first trace. What is the possibility of there being an additional vertical "step" and a deeper, undetected (untested) flow system?

<u>Response</u>: The possibility exists. The monitoring wells at the landfill do not sample the lower part of the Prairie du Chien aquifer and there was no way to monitor that part of the aquifer in this study. The major horizontal flow seems, however, to be in the middle of the aquifer, and we did not see any dye exiting at springs that appears to have missed the wells. Finally, the flow system does not extend locally into the underlying Jordan (sandstone) aquifer as evidenced by the lack of dye detection in any of the numerous Jordan wells monitored.

2. In retrospect, how could (should?) your investigation have been organized to maximize efficiency and lower costs without sacrificing reliability?

<u>Response</u>: Given the initial state of hydrogeologic and dye-tracing knowledge for this site, coupled with the administrative requirements and the residents' concerns, there is little that we could have done to maximize efficiency and lower costs. Armed with three years and half a million dollars worth of experience, requirements for future traces can be refined. We would remove the requirements for immediate public response and pre-set, automatic sampling plan expansion. Rather, necessary changes in the project plan should be technically driven.

The people who pay for dye traces and those who are affected by the results find it dificult to accept unknowns (costs, time-spans, out-comes, etc.) However, unknowns can not be completely removed and will continue to make dye traces complicated and expensive.

Deducing Karst Aquifer Recharge, Storage, and Transfer Mechanisms Through Continuous Electronic Monitoring - A Confirmation With Tracers

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ABSTRACT

Rainfall events produce significant changes in the stage, conductivity, and temperature of the groundwater mass in a karst aquifer. In theory, the arrival time of these thermal and conductivity pulses, relative to the stage pulses, should reveal the nature of the aquifer recharge, storage, and transfer mechanisms above the point of measurement.

An analysis of these fundamental aquifer properties was made utilizing continuous monitoring records from a karst aquifer in Virginia. These records suggested that there is no allogenic water entering this portion of the aquifer. This was evidenced by the subdued response in temperature during precipitation events. The monitoring record also suggested that the basin contributing flow to the part of the aquifer under consideration does not contain a significant conduit storage component. This was shown by the simultaneous response of stage, conductivity, and temperature to even the smallest of storm events. The rapid response to storm events and the limited rise in stage indicates that the basin is of very small dimensions.

Dye tracing of the two sources of allogenic water, in the area, confirmed that they do not contribute to the portion of the aquifer that was continuously monitored. The basin that contributes to groundwater flow at the monitoring point has been shown by dye tracing to be an area elongated along the strike of the local rocks, collecting only autogenic waters.

THEORY

In theory, the nature of the flow path or paths that groundwater in a karst aquifer traverses, to reach a particular site will be reflected in the physical and chemical adjustments each path imprints upon the water passing through it. Ashton (1966) proposed that the flow at any one point in the system will be the sum of the response of the portions tributary to that location. Given the most simple configuration the response of a measured parameter would be a deviation from base flow conditions to a peak and then a recession to the base flow condition. Response for a system that consists of two simple inputs that combine flow up stream from the monitored location will result in a record that reflects this as the sum of the characteristics of the two individual components (Figure 1). The resultant record will thus have peaks that reveal the existence and nature of each of the inputs, including their phase relationships. Ashton suggests that in addition to the flow rate other parameters of utility may be turbidity, pH, hardness,and temperature. He also suggests that specific conductance as a measure of the ionic character of the water is a practical alternative for hardness. The response for each of these parameters during the passage of a flood pulse will be modified by the various components of the aquifer that are encountered. The changes that the original pulses undergo and the relative timing of these pulses should reveal the nature of the storage, transfer mechanisms and the magnitude of each component of the aquifer system.

STUDY SITE

The Cathedral Hall Passage in Unthanks Cave has a free surface cave stream that flows through it. The Cathedral Hall Cave Stream contains an extraordinary assemblage of cave adapted aquatic invertebrates. One of these troglobitic organisms is an endangered species of Isopod Crustacean Caecidotea recurvata that the Virginia Chapter of The Nature Conservancy is committed to preserving. This study was conducted to determine the extent of the drainage basin and the transfer mechanisms for the water which contributes to the Cathedral Hall Cave Stream.

Karst by it's very nature presents certain problems that must be overcome by the groundwater scientist. The methods that have been used in this study have been implemented to understand the special nature of karst groundwater flow.

LOCATION OF STUDY AREA

The study area is located in Lee County Virginia (Figure 2), within the Powell Valley. The Powell Valley is one of the most western valleys of the Appalachian Valley and Ridge physiographic provence. The valley is bounded on the northwest by Cumberland Mountain and to the southeast by Wallen Ridge.

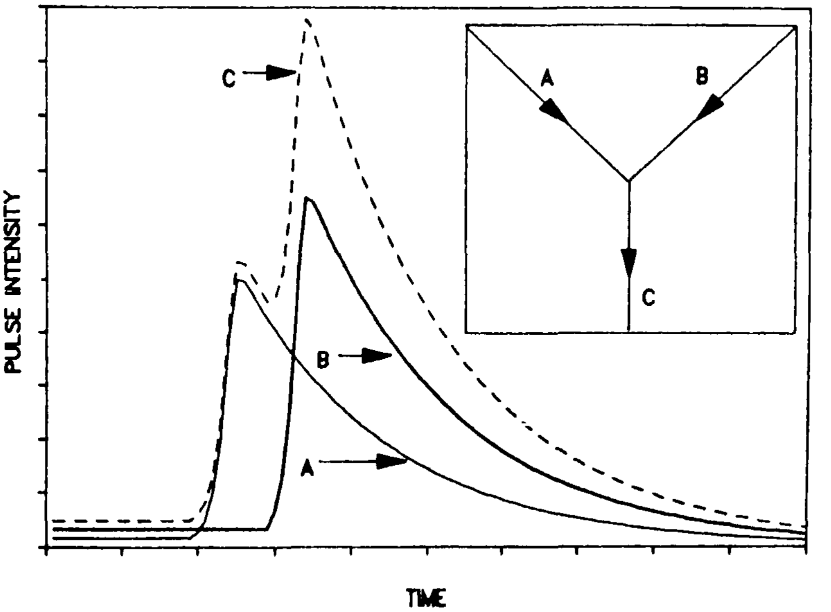

Figure 1. Flood wave complexing from multiple inputs (modified from Ashton 1966).

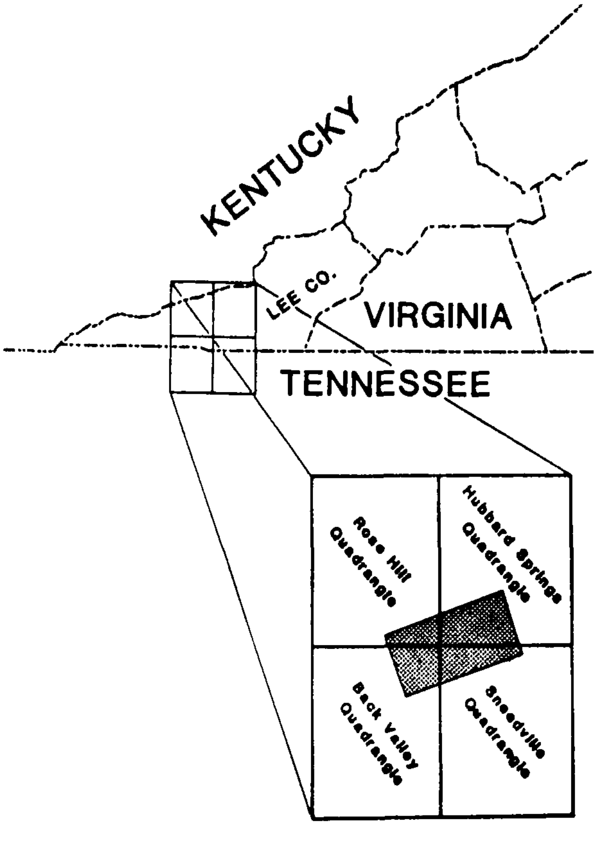

Figure 2. Location of the study area.

The limit of the investigation was controlled by the location of the Powell River and Wallen Ridge.

REGIONAL GEOLOGY

The rocks in the region are Ordovician carbonates with some shales in the valley and Silurian siltstones on the ridge. Unthanks Cave is formed in the upper portion of the Martin Creek Limestone near the base of the Hurricane Bridge Limestone. The Sandy Ridge Anticline follows the regional structure and is located between the cave and the Powell River. Dip of the rocks in the area ranges from horizontal to over 45 degrees, but within the cave, dip never exceeds 20 degrees. There are no major mapped faults with any surface expression in the study area.

CONTINUOUS MONITORING

Continuous monitoring of the Cathedral Hall Cave Stream was undertaken using a Campbell Scientific Incorporated digital micrologger model 21X. This is the central core of a Karst Water Instrumentation System (KWIS). The micrologger is a very versatile device that can be used to monitor many different kinds of sensors and physical parameters. The KWIS used in Unthanks Cave had three sensors that were monitored by the micrologger for the time that the system was installed in the cave. These sensors were used to monitor the stage (water level), temperature and conductivity (specific conductance) of the water in the Cathedral Hall Cave Stream.

The stage of the cave stream was monitored using a Druck pressure transducer. This device possesses a silicon wafer that varies in resistance as the pressure around it changes. This change is proportional to the height of water that is above the sensor. This transducer can detect changes in stage as small as .001 feet.

Water temperature was monitored using an Platinum Resistance Thermometer (PRT) supplied by Omega Engineering Incorporated. The resistance of a platinum wire changes with the water temperature. This resistance is measured relative to a 100 ohm precision resistor that is attached to the micrologger. Temperature is measured in degrees celsius with a resolution of .01 degree.

The specific conductance (conductivity) sensor that was used at this site was designed and constructed specifically for this site. The new design was developed in order to avoid the large current demand of earlier instruments. The new design has greatly reduced current demands and is thus able to be powered from the micrologger directly. Another advantage to the new design is the elimination of the "ground looping" problems associated with most other conductivity measurement systems.

The datalogger and sensors were located in the Cathedral Hall stream passage of Unthanks Cave. This passage is oriented along the strike of the rocks in the area and is apparently controlled by the fractures in the Martin Creek Limestone. The cave stream is about one inch deep by one foot across. There are numerous pools in the bottom of the accessible potions of the cave stream that are up to 18 inches deep. The sensor cluster used in this study was located in one of these pools to ensure that they always remain covered with water. The datalogger was located five feet above the cave stream on a ledge.

TYPICAL RESPONSE OF THE SYSTEM

A typical response for the aquifer system at Unthanks Cave can be seen in Figure 3. This record is related to the day 193 rain a single short uncomplicated event. The system responds with a rapid change in stage as the pulse of water progresses through the aquifer. Temperature of the water experiences a rapid rise at the same time as the stage changes are seen. Water temperature for this event responds in a increasing manner, events during colder seasons are represented by a decrease in temperature. There is also an immediate decrease in the specific conductance of the water from the base flow values.

A short term reversal of the trend of temperature and conductivity occurs within a few minutes of the initial system response. The detail of this occurrence can be seen most clearly in Figure 4.

The peak (the maximum deviation from base flow values independent of direction) is reached for each of these parameters at nearly the same time. Recession of stage is typically a simple return to base level without any abrupt changes in slope. The return of temperature to base flow values usually experiences a noticeable change in slope during its recession. Conductivity may also experiences a change of slope during the recession to base flow values.

All of the responses to storm events take this general form. Deviations from this form are believed to be due to variations in antecedent conditions. These changes may be due to the system being in a state of recovery from a previous storm event and thus its response is that of the combined pulses. Another factor connected with these deviations is the present state of saturation of the soil and rock. This may vary depending upon the length of time between precipitation events and the seasonal variations of evapotranspiration. Further variation may also arise from the stage increase associated with large storm events which bring other flow paths into operation that were not active during smaller events.

The simultaneous response of temperature, conductivity and stage is believed to be an indication that fresh storm water is arriving at the monitoring site from inputs very nearby and without significant restriction of flow. Hess and White (1973,1988) and Meiman et. al. (1988) observed a lag between the arrival of the stage response and a response in temperature and conductivity. This has been attributed to the propagation of

Figure 3.

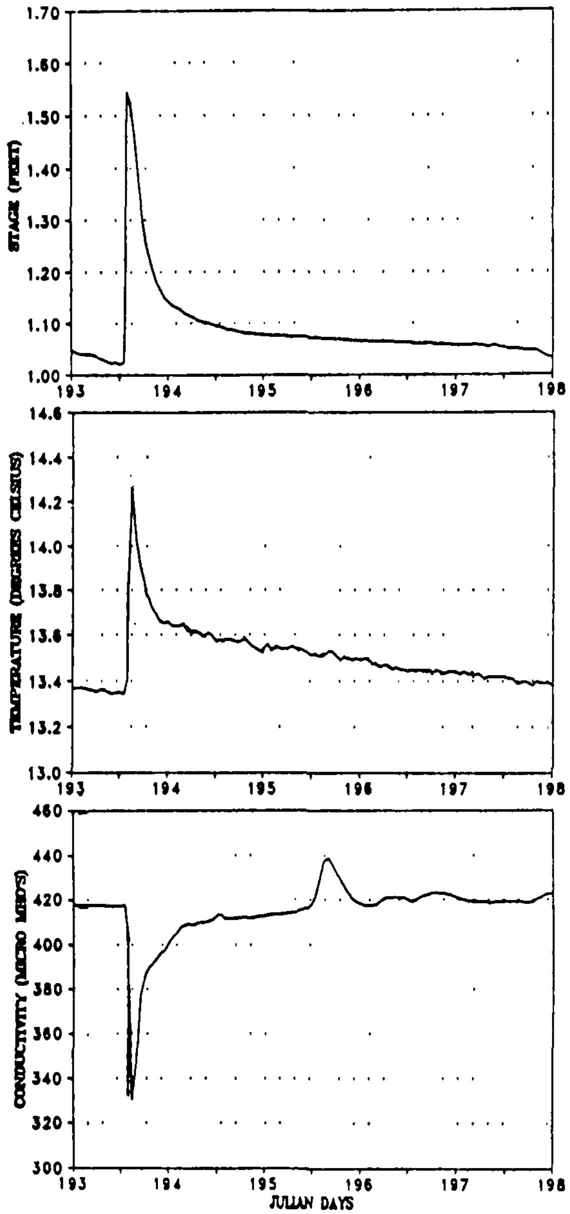

Figure 4.

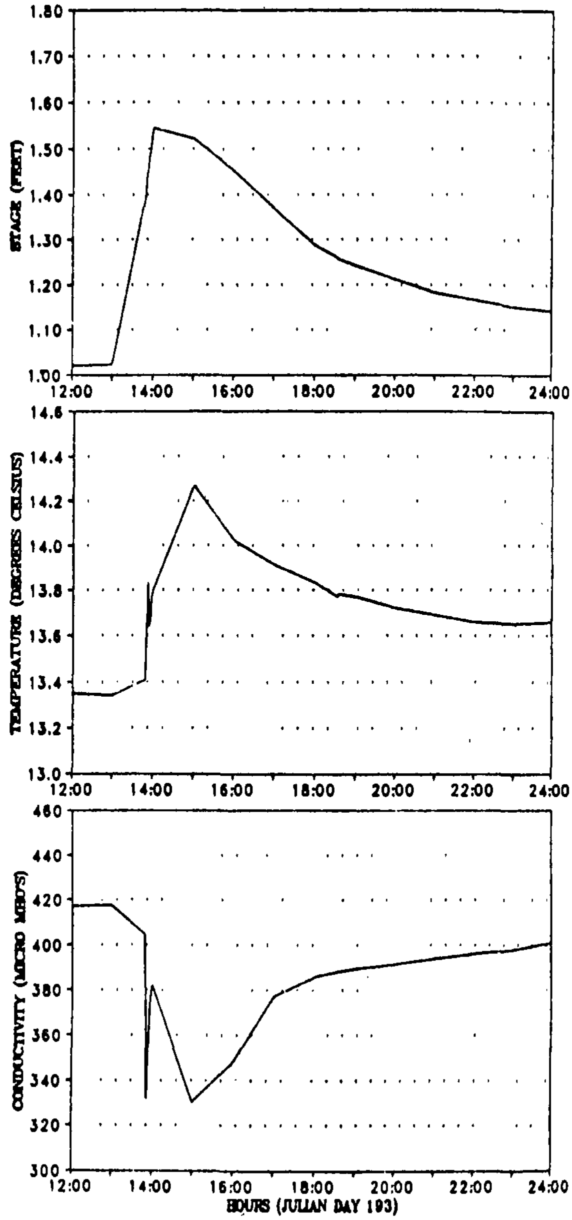

Figure 3. Aquifer response to rain event of julian day 193. Representative of typical aquifer response.

Figure 4. Detailed inspection of the fluctuation in specific conductance and temperature shortly after first response to the julian day 193 rain event.

the stage pulse as a pressure wave through the phreatic portion of the aquifer in advance of the fresh input waters. This lag effect has not been observed at this site, thus it is asserted that there is no significant phreatic contribution to the flow that passes the monitoring site.

The reversal of trend in temperature and conductivity (Figure 4) a short time after the initial response to the storm event is presumed to be the arrival of higher conductive water with a temperature signature resembling that of base flow values. This water is assumed to be the flushing of stored water as the fresh input waters affect other parts of the system. This may be the arrival of a flow component from further up stream or displacing water from nearby storage that exhibits more restricted flow of extended residence time than the initial response component. The cessation of this reversal of trend is interpreted as the flushing of the stores of this second component or the overwhelming of the influence by an increasing volume of fresh input water.

Meiman et. al. (1988) saw multiple slope changes in the conductivity response that was attributed to the input of separate concentrated allogenic sinking streams in the basin. There is normally only one slope change seen in the conductivity response with the present study. The arrival of this pulse is seen as a more conductive water than the fresh input waters. This high conductive signature is not likely to be associated with an allogenic sinking stream.

Continued departure from base flow values for all parameters is seen as the fresh input waters dominate the flow. Maximum stage occurrence is associated with the cessation of the source of fresh input waters to the system. The return to base flow conditions occurs as the excess water moves through the aquifer. Conductivity and temperature peaks are reached slightly after the stage crest is reached. Recession to base flow conditions for conductivity and temperature is marked by an initially rapid change that corresponds to the swift recession of stage. The slope of the recession of conductivity and temperature changes as the stage response approaches base flow. This change in slope is due to reduced significance of the quick flow component of the fresh input water and an expansion of the influence of the restricted flow extended residence time component.

SYSTEM RECOVERY AFTER STORM EVENT

The response to the day 259 precipitation event is similar to the record for all single storm events. This event clearly displays a temperature phenomenon that exists in the record of most events that have been observed in this study (Figure 5). After the peak response of the event has been attained for all parameters and the recession to base flow conditions begins, the return of temperature to base flow values lags behind that of stage, and to a lesser extent it lags behind conductivity.

Figure 5.

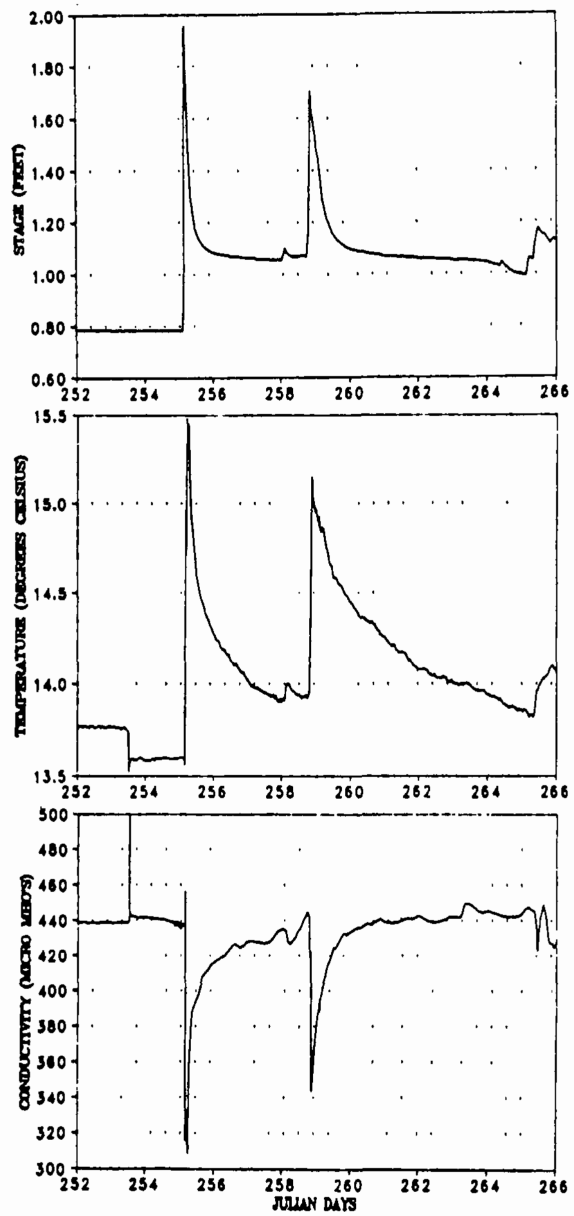

Figure 6.

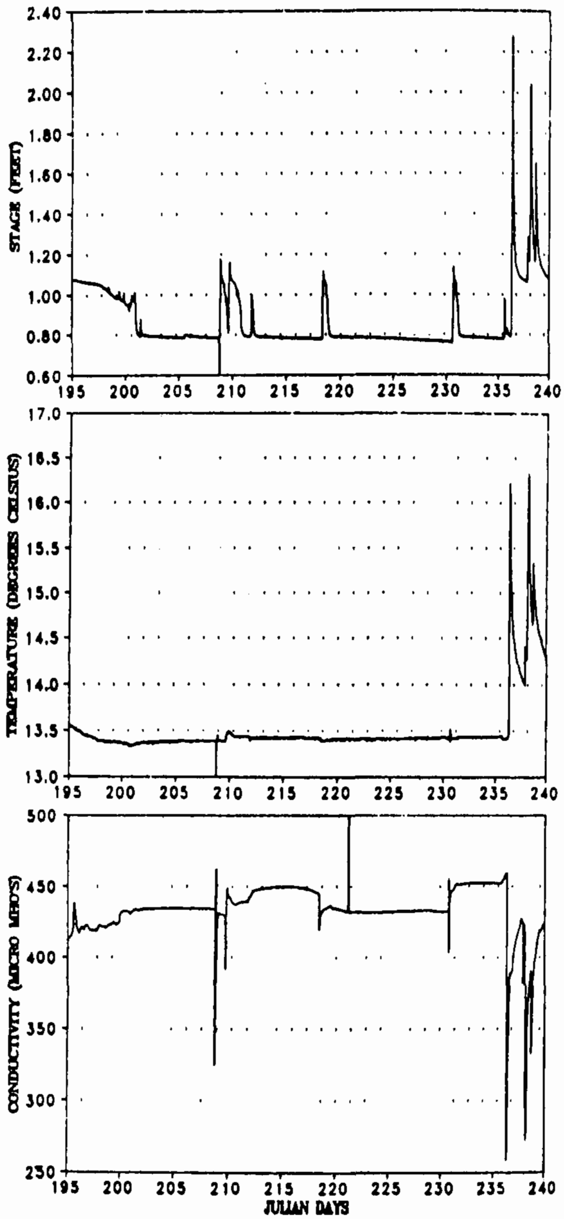

Figure 5. System response to storm events of julian days 259 and 255. The response to the day 259 event exhibits the delay in recovery of temperature, conductivity and stage relative to each other. The initial response to the day 255 event show an initial increase in conductivity.

Figure 6. System response to drought conditions.

In contrast to this study Meiman et. al. (1988) observed that the return to base flow values for temperature was more rapid than for conductivity. Storage of thermal energy in the rock mass in the vicinity of the active flow routes is the likely reason for any lag effect in temperature. The release of this thermal energy from the rock back into the water slows the return of temperature to base level until the thermal stores have been depleted.

This difference in response in the two studies may be related to the presence or absence of phreatic passages or the distance between the monitoring site and the source of input. The diffuse stores in the subcutaneous zone and saturated rock mass with restricted flow conditions will have a relatively high conductance due to an extended residence time. The aquifer at Mammoth Cave has been shown to receive flow from both concentrated and diffuse sources under both flood flow and base flow conditions. The combining of multiple sources will slow the return to base flow values of the conductance at a down stream monitoring point. Additionally, the aquifer at Mammoth Cave has a significant phreatic zone. As the storm pulse is transmitted through the aquifer a significant portion of the fresh input waters may be pushed from the conduit into the conduit adjacent stores. These short residence time waters will be released from the conduit adjacent stores as the flood event wanes. The low conductivity waters from the conduit adjacent storage will slow the recession to base flow values. If an aquifer receives the majority of its base flow from extended residence time water the response will be such that the conductivity may return to a base flow condition before the thermal stores in the rock from the storm event can be depleted.

Meiman et. al. (1988) saw that as a storm pulse was transmitted through the system, there is a lag in the initial response of temperature relative to conductivity and stage. They also note that the amount of lag is affected by the antecedent conditions. The absence of a thermal lag at the onset of the storm event, in the present study, suggests that this initial response is dominated by a quick flow component. This may reflect concentrated autogenic contribution into enlarged fractures and vadose shafts. Through this route of transit the water would not have enough time to react chemically with the rock to significantly change its conductance. The leading edge of the thermal response would not be delayed relative to the other values if the input is near the monitoring site. Additionally there is sufficient contact with the rock that a reduction of the temperature of the water without a significant delay in the arrival of the initial temperature change is likely. If flow at the monitoring site received contribution from the allogenic sinking streams in the area this water would not have sufficient contact with the rock to subdue the thermal response.

CONDUCTIVITY RISE AT THE ONSET OF AN EVENT

Meiman et. al. (1988) saw an initial rise in conductivity at the onset of storm events. This is explained as the flushing of high conductive stores from portions of the aquifer that are not typically active during base flow. The storm event of day 255 (Figure 5) does show a small initial rise at the onset of the storm event. This is the most noticeable occurrence of this effect seen during the time this site was monitored. The typical response has been, as stated before, an immediate drop in the conductance at the same time that the other parameters respond. This suggests that the flow route for the fresh input water must be very direct and possessing very limited stores in order to dominate the initial response, and that only periodically are there sources of high conductive water to be flushed ahead of this storm pulse.

LOW FLOW CONDITIONS

The flow of the cave stream during the period from day 195 to day 235 was seriously reduced due to the dry conditions (Figure 6). Starting about day 201 the stream flow was reduced enough that the level of water in the pool, where the probes had been located, was no longer sufficient to maintain a normal flow out of the pool. The reduced pool level appears to be controlled by fractures at a location lower than the normal pool level.

In response to several small storm events, that occur during this period of time, the pool level is temporarily raised to the overflow point that signifies the return to stream flow. This pool height is only maintained for a short period of time until the flow is cut off and the surface is returned to a level controlled by the fractures. Water temperature during this period is very stable and there is only a limited response that can be associated with the small storm events that have been recorded. This thermal stability indicates the limited extent of these storm events and/or reflects the efficiency of the thermal stores. Conductance of the water fluctuates with each storm event but is stable between storm events. When the stream ceases to flow, the pool is once again isolated and so measurement of the conductance represents the character of the last water to be flowing in the stream before flow stopped.

During a portion of the time that the stream in the Cathedral Hall Cave Passage had ceased to flow, the sinking streams that flow from the ridge still maintain a low level of flow. If these streams had contributed flow to the portion of the cave that was continuously monitored during this time period the flow of the cave stream would have been maintained.

DYE TRACE RESULTS

ALLOGENIC SINKING STREAMS

Two allogenic sinking streams flow off of Wallen Ridge in the area of the Cathedral Hall Passage. The "Scott Farm Sinking Stream" (SFSS) was found to sink at the contact of the Hurricane Bridge Ls. and the Martin Creek Ls. Water flowing in this stream includes runoff from a few farms and a winding road that cross the ridge, State Route 758. The sinking stream "SFSS" (Figure 7) was traced at three different times due to it's proximity to the cave stream of interest. At no time during the study was this sinking stream found to connect to the Cathedral Hall Cave Stream. Dye from "SFSS" has been detected in the cave at monitoring points down stream from the Cathedral Hall Passage.

The water flowing in the Horn Farm Sinking Stream (HFSS) was shown to flow away from Unthanks Cave parallel to strike but in the opposite direction of all other traces. Water Sinking at this location on the Horn Farm was never shown, under the conditions investigated, to contribute to any portion of Unthanks Cave.

AUTOGENIC SOURCES

During one of the research trips into the Cathedral Hall Passage a strong odor of "diesel" fuel was detected. After exiting the cave an investigation of local sources of this product was initiated. It was determined that the most likely source was due to dumping of excess product from large metal drums along side a sinkhole located behind "Bacons Store" (BSD). This is the site of a small country store that also sells gasoline. This sinkhole was later used for dye injection in order to further understand its connection to the cave system. Water entering at this point flowed to the Cathedral Hall Passage and was detected at the monitoring site. Dye introduced at this site was also detected at dye monitoring sites in the cave down stream from the Cathedral Hall Cave Stream.

"JWD1" is a sinkhole located a short distance south of State Route 758 in a wooded rocky portion of a farm. Dye input at this location was detected in the Cathedral Hall Passage. This dye was also detected at dye monitoring sites down stream from the Cathedral Hall Cave Stream. This site was only investigated during high flow.

Sinkhole "JWD2" is also located a short distance south of State Route 758 on the same farm. This sinkhole is located a few hundred feet East of JWD1. Dye introduced at this site was detected at dye monitoring sites down stream from the Cathedral Hall Cave Stream. This site was tested under high flow conditions but no dye from this site was detected in the Cathedral Hall Passage.

"RSD" is a sinkhole that is located very close to and just south of State Route 758. It is about half way between "BSD"

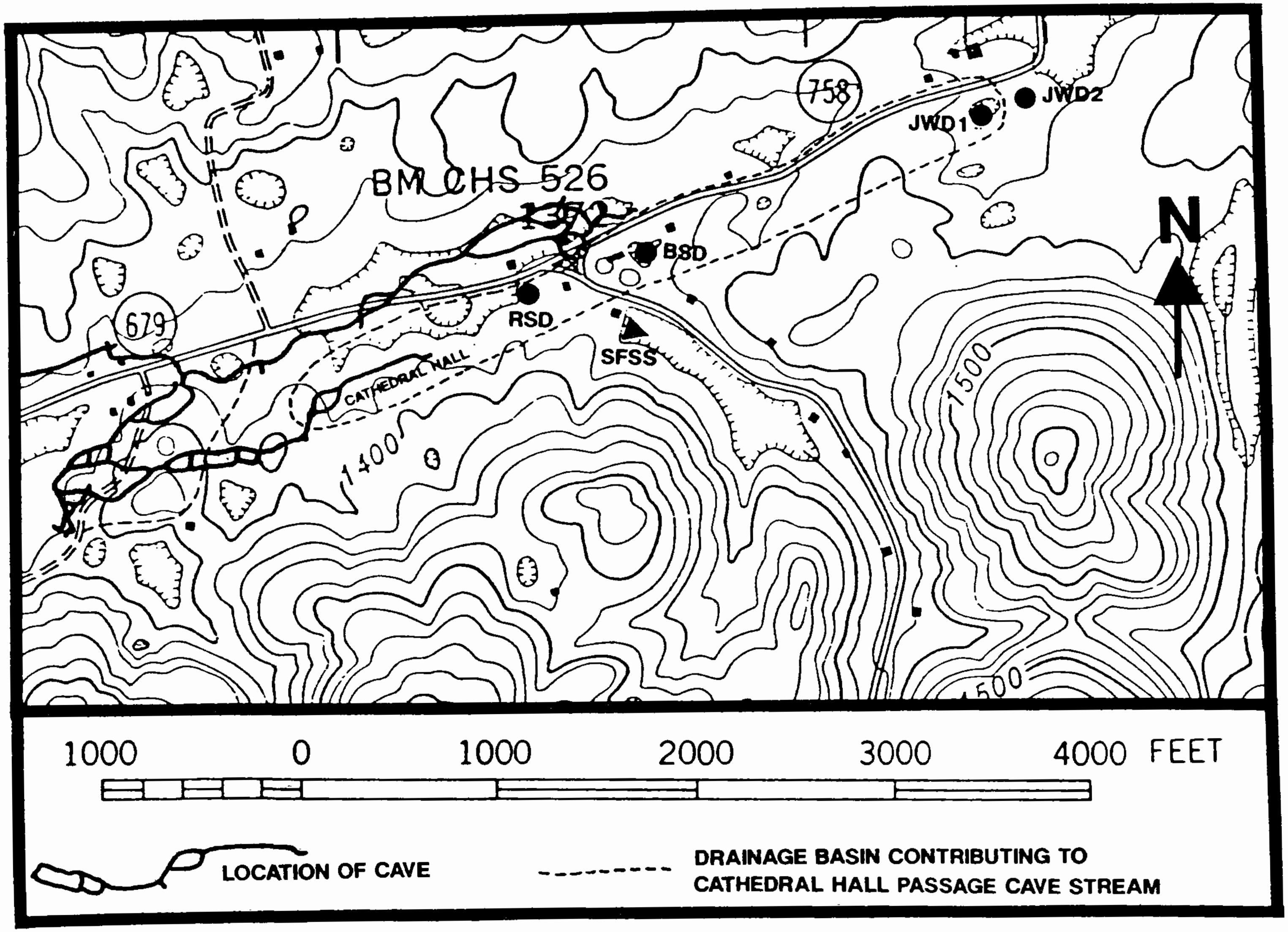

Figure 7. Map showing the location of dye insertion sites and the basin that contributes to the Cathedral Hall Cave Stream.

and the Cathedral Hall Passage. Dye introduced at this point was detected in the Cathedral Hall Passage and at dye monitoring sites further down stream.

CONCLUSIONS

The simultaneous response of all parameters at the onset of a storm event establishes the existence of a quick flow component to the flow of the Cathedral Hall Cave Stream. The lack of delay in response of conductivity and temperature relative to stage response confirms the absence of a significant phreatic flow path for this quick flow contribution.

Variations of conductivity and temperature shortly after the initial response from base level reflect the arrival of fresh input quick flow waters with flushed stores from more distant or restricted flow sources in the aquifer. Flushing of these stores or overwhelming of their signature by the increased volume of fresh input is considered to be the reason for the cessation of these occurrences. The ordinary absence of high conductance waters to be flushed from storage at the onset of storm events suggests that the routes of quick flow are normally drained of extended residence time waters.

Return of the waters conductance to base flow values more rapidly than temperature, reflects the absence of concentrated allogenic inputs and/or conduit adjacent storage of fresh input waters. Concentrated allogenic sources and conduit adjacent storage of fresh input water would have low conductance and compete with the high conductance restricted flow waters that maintain the base flow. The absence of these sources in the flow at Unthanks Cave allows conductance to return to base flow conditions before temperature.

The cave stream stops flowing during dry periods but the sinking streams flowing from Wallen Ridge, for a portion of this dry period, continued to flow and sink. There flow must not be related to the cave stream under these conditions.

Tracing of both allogenic sinking streams in the area shows that "HFSS" does not contribute to any portion of Unthanks Cave and that "SFSS" contributes to portions of the cave down stream from the Cathedral Hall Passage. Tracing of sinkholes in the area has shown the location of the drainage basin and confirmed the source of the water is exclusively autogenic.

The dye tracing has supported the conclusions of the nature of the drainage basin and the continuous monitoring has revealed more detail about the transfer mechanisms than the qualitative dye tracing could do.

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BIOGRAPHICAL SKETCHES

Peter Idstein received his Bachelor of Science degree in Geology at Eastern Illinois University. He is completing his Master of Science degree in Geology at Eastern Kentucky University. Mr. Idstein has spent one year working at the Florida Sinkhole Research Institute conducting studies on conduit dominated groundwater flow. He has also spent a year working for Ewers Water Consultants conducting dye tracing studies and continuous electronic monitoring studies in many karst dominated and non-karst terranes.

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Deducing Karst Aquifer Recharge, Storage, and Transfer Mechanisms Through Continuous Electronic Monitoring - A Confirmation With Tracers

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Richmond, Kentucky

Question:
Are there temperature lag-times, and are they the same for low flow and high flow conditions?

Response:
The study at Unthanks Cave showed no temperature lag-times, at the onset of the storm events during the period that the system was monitored. Work at Mammoth Cave National Park by Meiman et.al. 1988 did show significant temperature lag-times at the onset of storm events.

There were temperature lag-times in the return to base flow values. Both stage and conductivity returned to base flow conditions before temperature. The difference between low flow and high flow conditions does not seem to be as significant as the variability in temperature of the fresh input waters.

PETROLEUM HYDROCARBON REMEDIATION OF THE SUBCUTANEOUS ZONE OF A KARST AQUIFER, LEXINGTON, KENTUCKY

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Delta Environmental Consultants, Inc.
Charlotte, North Carolina

Investigation of karst aquifers using dye tracing techniques has reached widespread acceptance in the hydrogeologic community. These techniques have only recently been applied to tracing the release of contamination from a point source. These studies fail, however to provide suggestions for the remediation of the karst aquifer affected by the release.

Following the release of 1300 gallons of gasoline from an Underground Storage Tank an interceptor trench was installed and recovered approximately 800 gallons of the lost product. A subsequent tracer study from the loss location indicated flow through the subcutaneous zone at 400 ft/day and discharge to the ultimate resurgence, 1.75 miles away, in 67 days. Ground water samples collected at the spring resurgence indicated non-detectable levels of petroleum hydrocarbons. Residual soil and ground water contamination still remained in the subcutaneous zone at the release location. This residual contamination acted as a continuing source of ground water contamination into the karst conduit drainage network and resulted in gasoline vapors in local businesses.

A remediation system was designed to remove the residual gasoline contamination through the use of soil vapor extraction, in-situ soil washing and standard pump & treat technology. The system was put on line during April 1991. The recovery and treatment system continues to remove gasoline contamination from the subcutaneous zone and reintroduce treated ground water--mimicking constant precipitation event.

INTRODUCTION

Tracer investigation in karst regimes has become an exact and well defined technique to determine conduit drainage network boundaries and conduit flow paths. Tracer techniques have recently been used to trace the transport fate of contaminants from a point source release location within the conduit drainage network. Studies of this type have been used to estimate the residence times of the contaminants in the aquifer but fail to address remedial options for the affected portions of the aquifer. Releases of contaminants are often times to the overburden and subcutaneous zone of the aquifer rather than directly to the well developed conduit drainage.

Remedial options for karst aquifers are often not considered due to the fact that they are applied to the conduit system and fail to address contamination of the subcutaneous zone. This study presents information collected using classical contaminant investigative techniques for unconsolidated overburden materials and using dye tracing techniques and how is is applied applied to design and implement a remediation plan of the subcutaneous zone for the case study presented.

The subject area is located at the corner of South Limestone Street and Gazette Avenue in Lexington, Kentucky (Figure 1 & 2). The site lies in the Inner Blue Grass Physiographic Province of central Kentucky. North of the site are office buildings and houses, the University of Kentucky is to the east, office buildings and a residence to the west, apartment buildings to the southwest, and a bank, convenience store, and restaurant to the south.

SITE HISTORY

The site has been a gasoline station and convenience store for at least 20 years. A map showing the former, Shop-N-Go (Super America station), surrounding businesses and residences is included as Figure 2. On April 12, 1986, the Shop-N-Go reported a sudden loss of 1,300 gallons of gasoline from an underground storage tank. On April 13, 1986 gasoline was detected in the basement of Mr. Gatti's Pizza Restaurant, located approximately 400 feet to the south. Because the detection of gasoline occurred approximately 24 hours after the loss, the gasoline apparently originated at the Shop-N-Go site.

Several remedial activities were undertaken in response to this loss. Activities included excavation and removal of leaking underground storage tanks, removal of contaminated soils, and removal of liquid phase hydrocarbons (LPH) from the tank basin at the Shop-N-Go site. A recovery trench system was installed adjacent to Mr. Gatti's Pizza Restaurant and has been in operation since April 1986. Initial remedial activities resulted in the recovery of nearly 800 gallons of gasoline and the excavation of 350 cubic yards of contaminated soil from the site. Additional gasoline was reported to have discharged to the storm water sewers at the site. Approximately 800 gallons of the gasoline was recovered during the first few weeks after the reported spill. No LPH has been detected in the recovery sump since December 1986; however, contaminated ground water has been recovered since remediation was initiated.

GEOLOGY AND HYDROGEOLOGY

GEOLOGY

The site is situated within the Inner Bluegrass Physiographic Province of central Kentucky. This area is characterized by flat lying to very gently dipping limestones, dolostones and shales of Ordovician Age. Strata beneath the site consists of limestones and thin shales of the Lexington Limestone. Members of this formation show an intertonguing relationship throughout the Lexington area. The members underlying the site are, in descending order, the

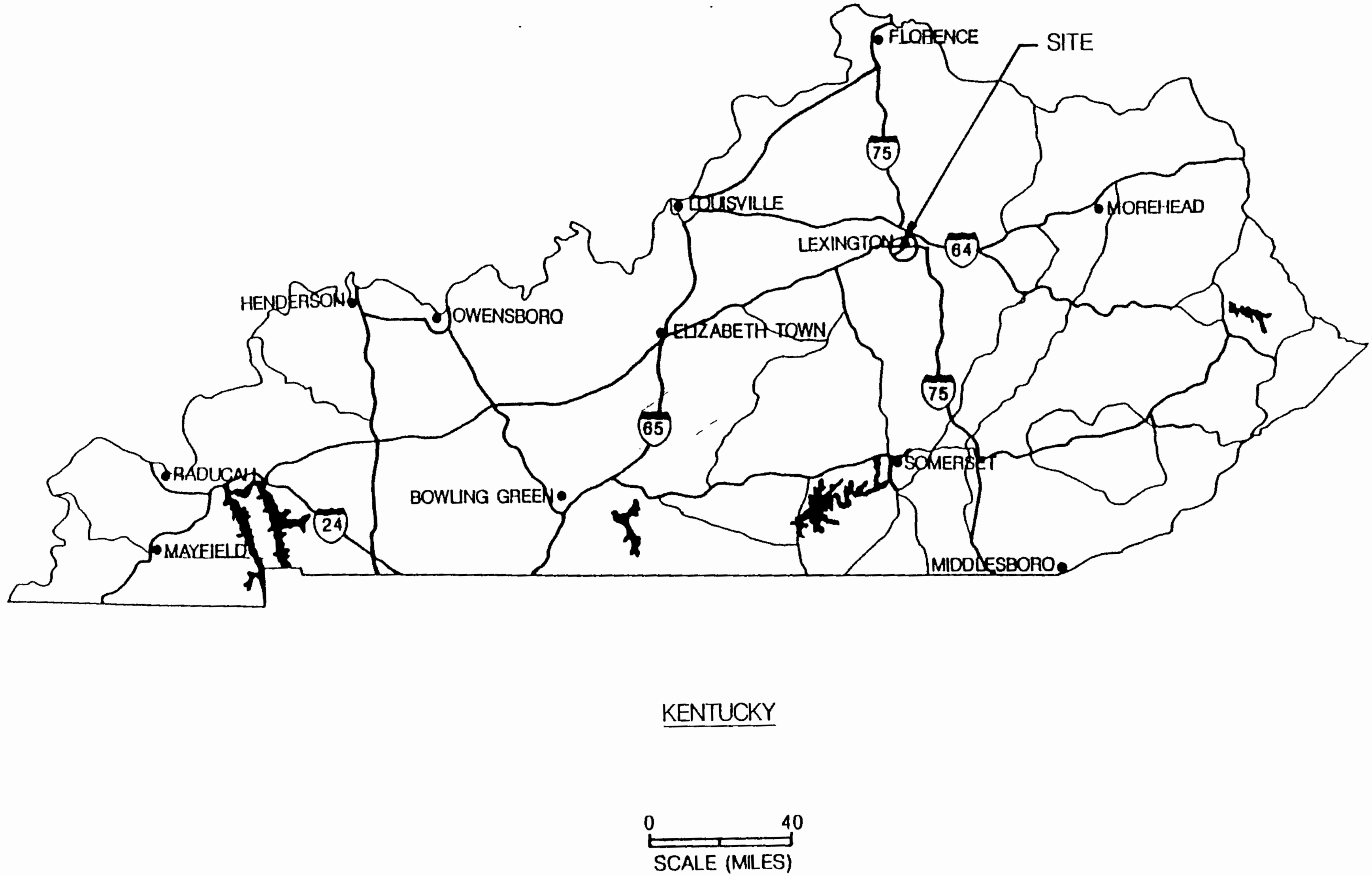


Figure 1. Site Location

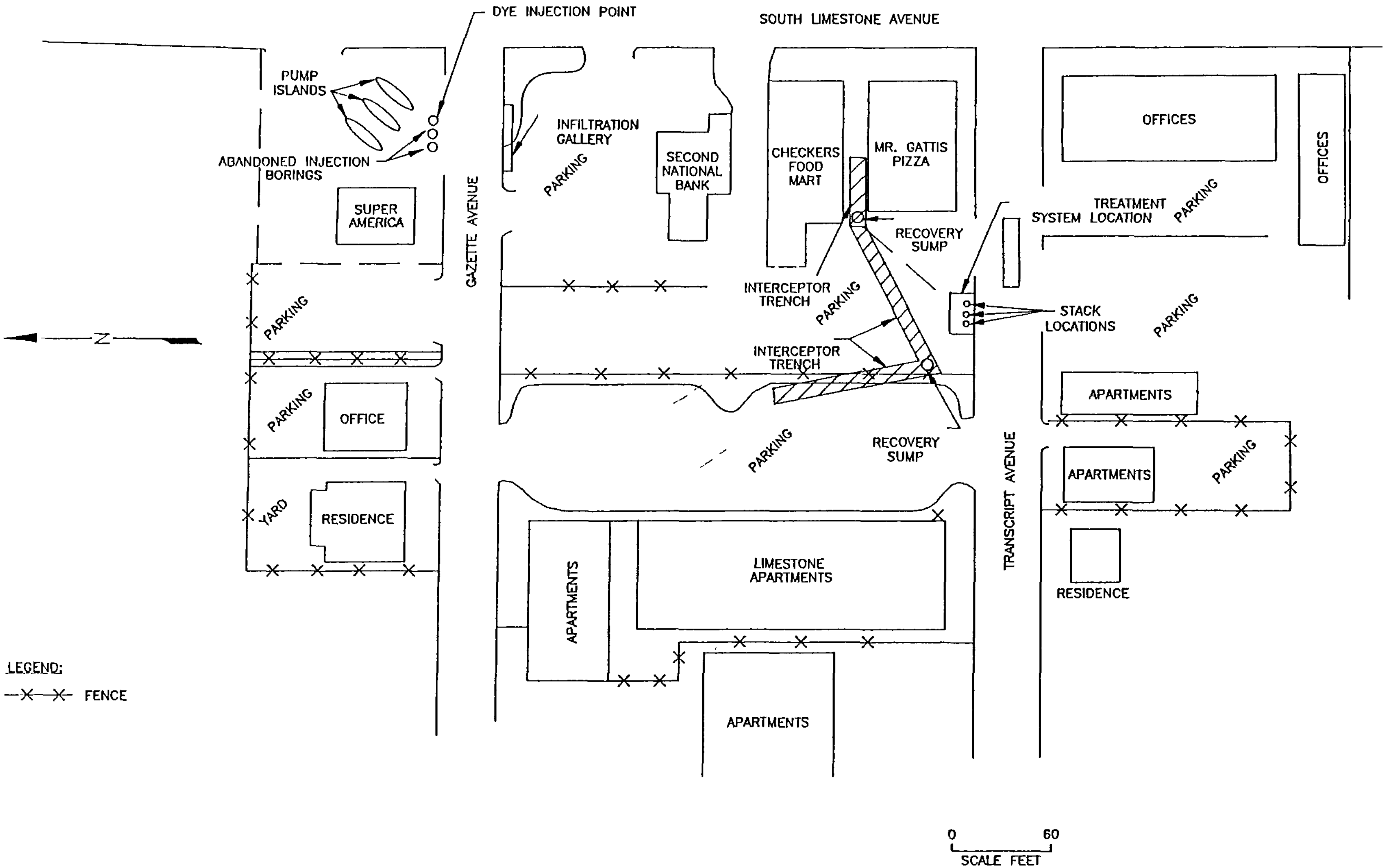


Figure 2. Site Map

Tanglewood, Brannon and Grier Members of the Lexington Limestone. The Tanglewood and Grier Members are characterized by thinly bedded argillaceous limestones with very minor and isolated shale partings. These limestones are subject to solutional modification by ground water flow as evidenced by solutional features along road cuts as well as abundant closed surface depressions in the area. The Brannon Member is characterized as argillaceous limestone containing shale and chert beds and solutional modification is uncommon (Miller, 1967).

A total of 36 test borings were drilled during November 1987 and twenty three additional borings were advanced by Delta in February 1988 and May 1990. The boring locations are shown in Figures 3 and 4. Material beneath the site consists of lean to fat unconsolidated overburden with minor amounts of chert and limestone rock fragments increasing with depth. The depth to the Tanglewood Member ranges from 8.5 to 16 feet below grade.

Soil was screened with a PID and soil samples were collected from the 14 borings advanced in May 1990.

Soil screening and analytical results indicate that elevated levels of petroleum hydrocarbons are concentrated at the soil/bedrock interface over a large area. Levels appear to be higher in a small area west of Mr. Gatti's Pizza Restaurant. The approximate extent of elevated hydrocarbon levels in soil is shown in Figure 5.

HYDROGEOLOGY

Ground water was not encountered during the boring programs, therefore, there have been no static measurements of ground water at this location. Flow mainly occurs along the soil bedrock interface through tiny channels and solutionally enhanced joints and bedding planes known as the subcutaneous or epikarstic zone.

Ground Water Tracer Study

In order to determine the ultimate pat for ground water and, hence, contaminant movement, it was necessary to conduct a dye trace from the areA of the gasoline loss.

After initial reconnaissance on April 15, 1989, the following observations were made:

1) Directly underlying the site is the Tanglewood Member of the Lexington Limestone. The Tanglewood is karstifiable, as evident by many area sinkholes.

2) Most area springs are discharged from the Grier Member of the Lexington Limestone.

3) Between the Tanglewood and Grier is the Brannon Member which, although contains some shale beds and chert, is considered a semi-permeable unit.

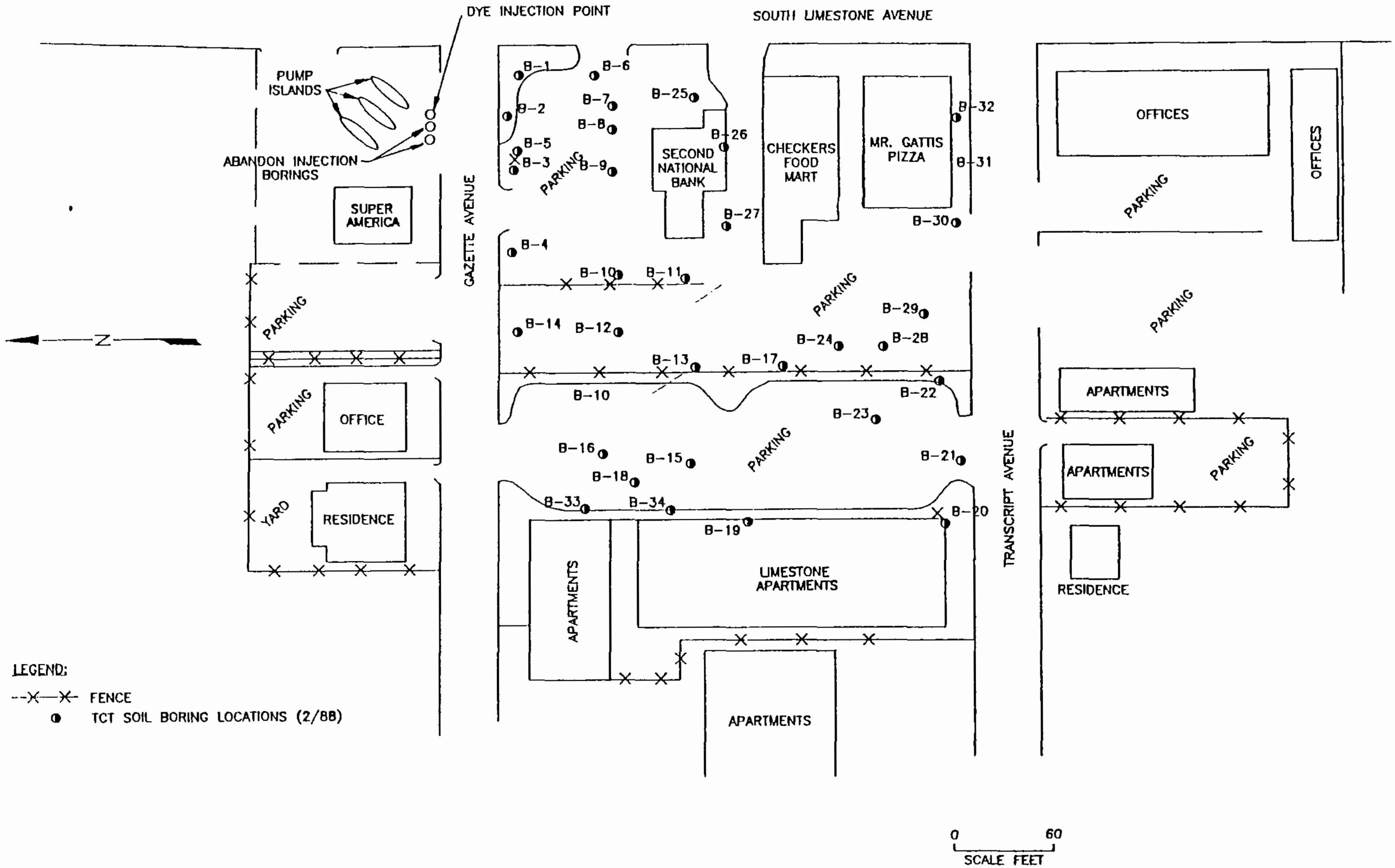


Figure 3. Borings advanced during 1987

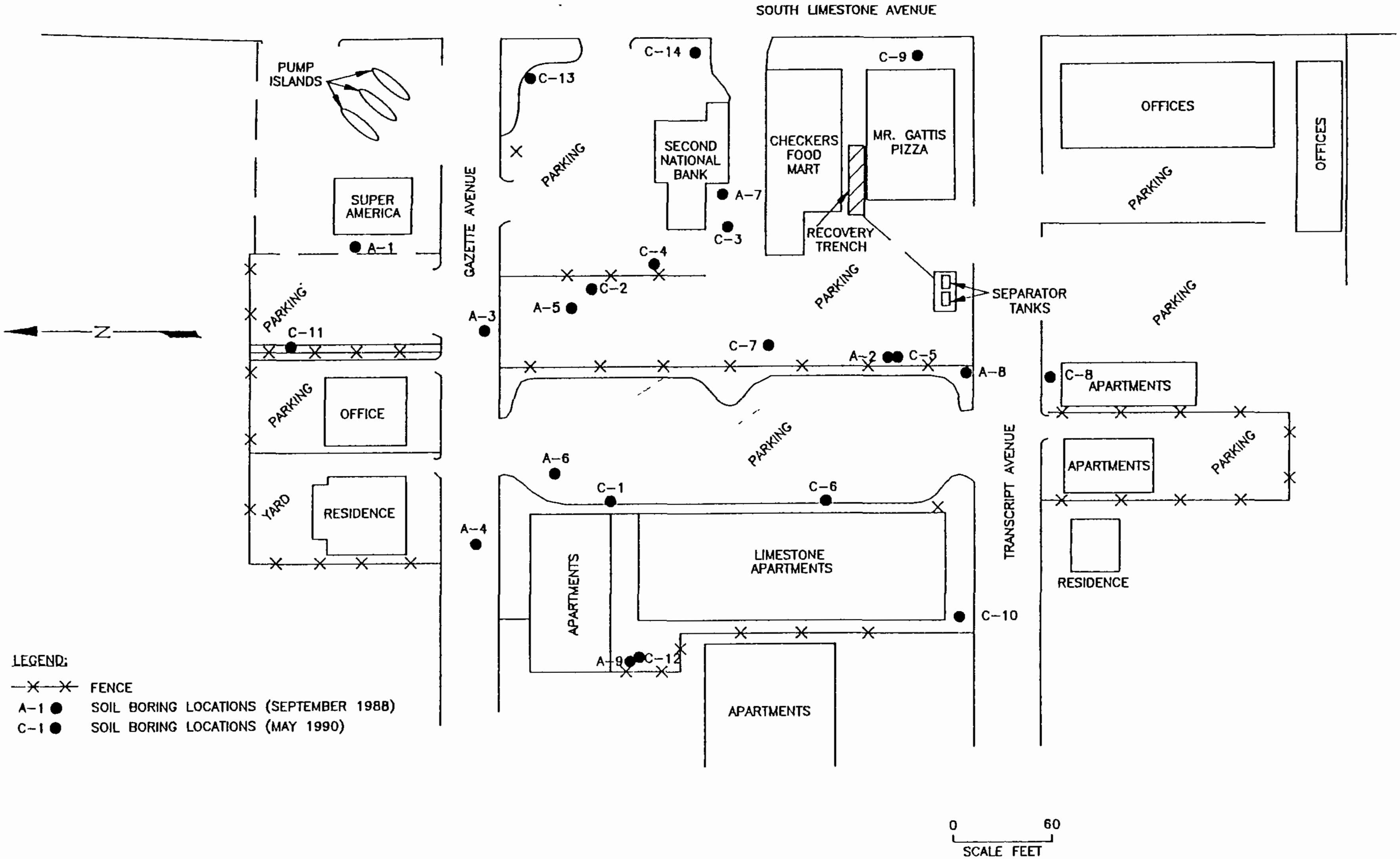


Figure 4. Borings advanced during 1988 and 1990.

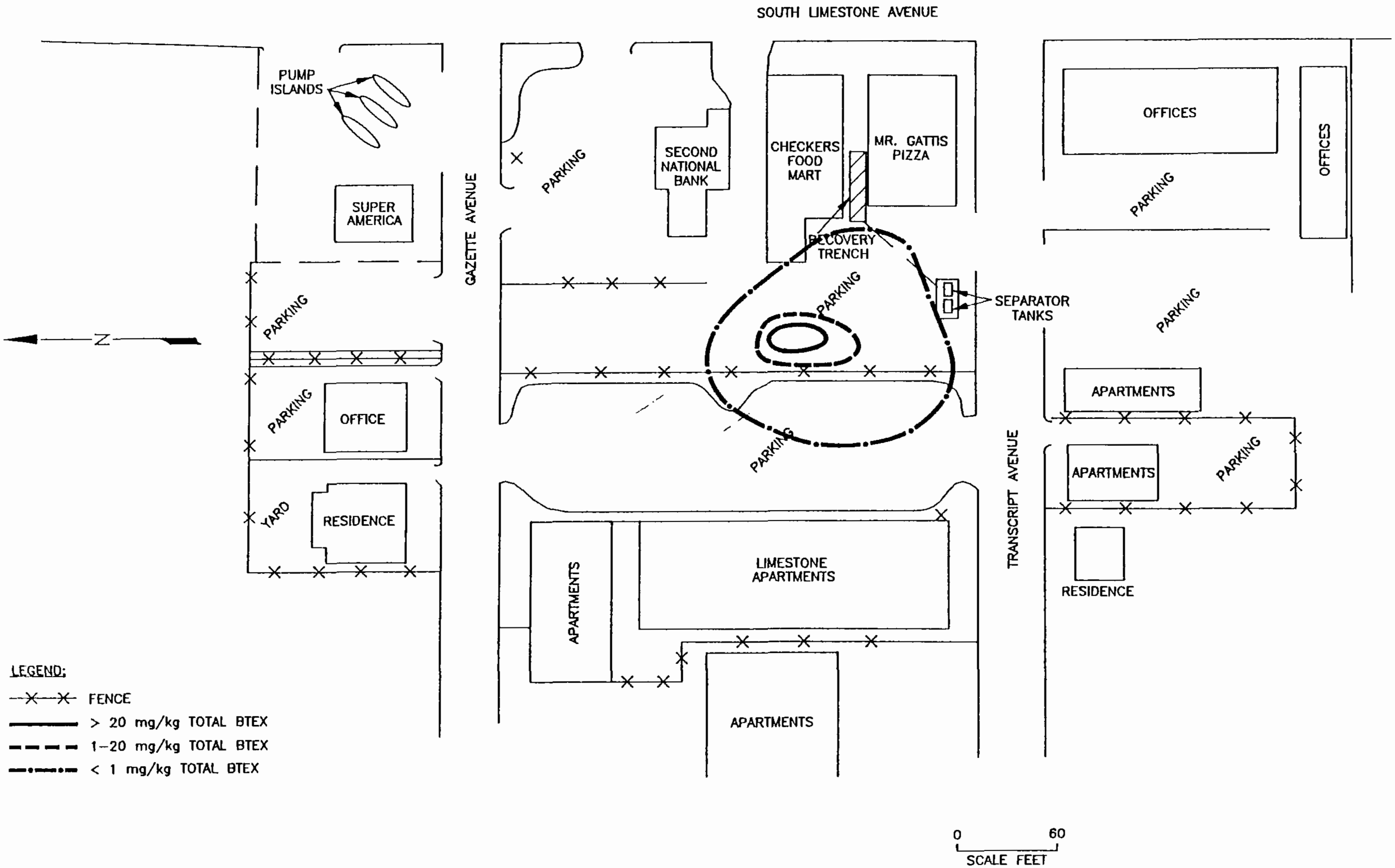


Figure 5. Estimated extent of soil contamination based on soil boring information.

Table 1

Detector Locations	Location Codes
Recovery Trench	RT
Mr. Gatti's Sump	MG
Storm Sewer	SS
Storm Sewer at Railroad Culvert	RR
Big Elm Country Club	BE
Wolf Run	WR
Vaughn's Branch	VB
Town Branch	TB
Hickman Creek	HC
McConnell Spring	MS
Preston's Cave Spring	PS

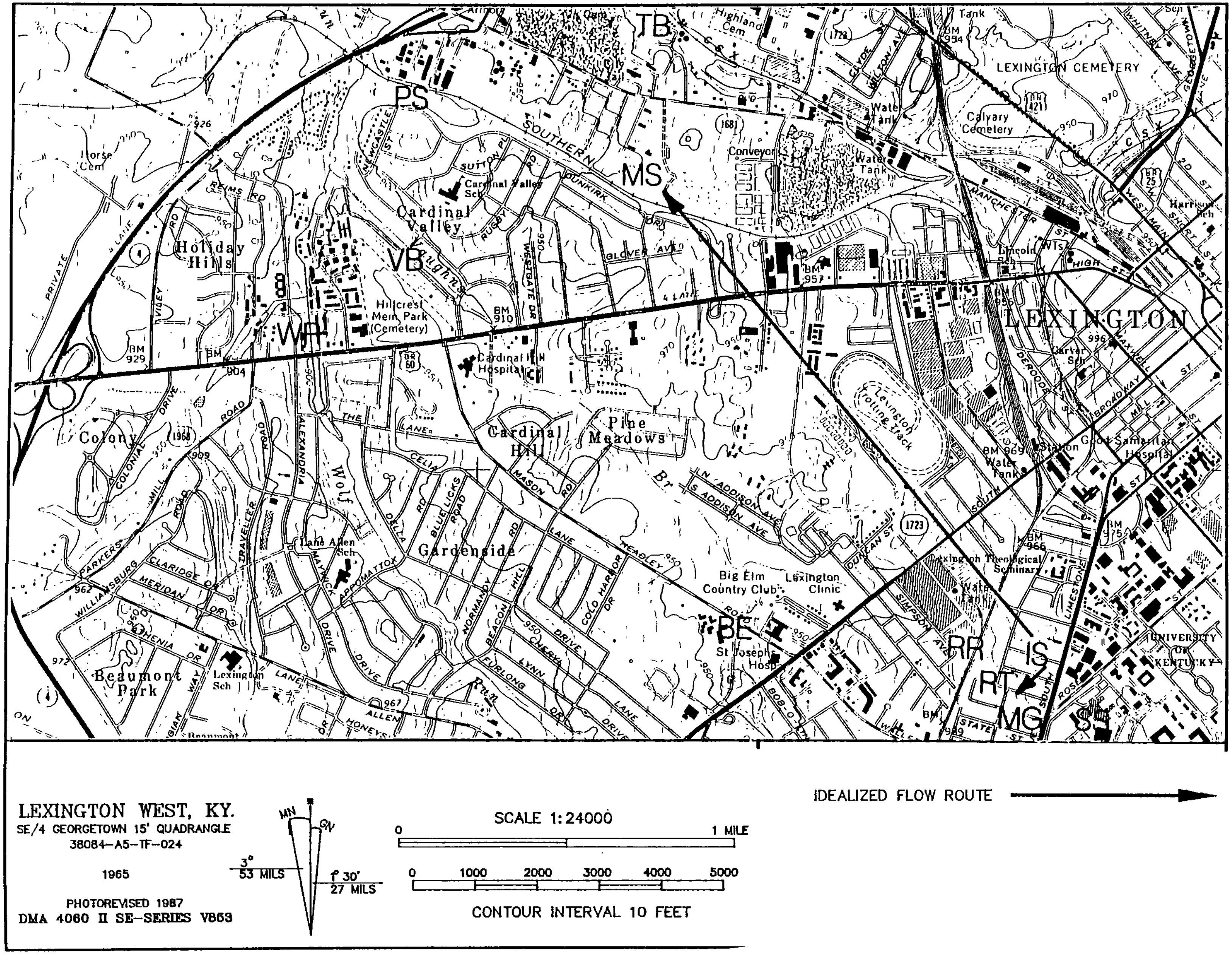


Figure 6. Dye detector locations and tracer results.

4) Local dip is to the northwest at approximately 20 feet per mile. There are no mapped faults in the immediate area of the site.

5) The storm sewer (later used as a dye detector site) which passes the site and receives water from the separator tank, ultimately discharges in Vaughn Branch (also later used as a detector site).

6) Dye detector locations chosen are included in Table 1 and shown in Figure 6.

Dye Background

Prior to dye tracer studies, an analysis for the presence of background levels of fluorescent dyes is necessary. Dye detector sites were chosen (Table 1 and Figure 6). Background detectors were deployed and analyzed on two occasions. These detectors, collected on July 14, 1989 and July 21, 1989 indicated the presence of extremely weak concentrations of fluorescein dye in the storm sewer and McConnell Spring sites. The dye was determined to be of insignificant concentrations to negate the use of fluorescein for the tracer study.

Dye Injection

On July 21, 1989 at approximately 11:00 a.m., fluorescein dye (acid yellow 73) was injected into a fresh 10 inch diameter soil boring at the Shop-N-Go site. The location of the boring is shown in Figure 2. The boring encountered bedrock at approximately 12 feet. Prior to the boring of the injection well, two additional borings were drilled approximately 3 and 6 feet from the injection well (Figure 2). These wells have a total depth of 9 and 10 feet, respectively, and would not effectively take a slug of water. The injection well readily accepted over 100 gallons in a period of less than 10 minutes. After the initial slug of water, 3 pounds of fluorescein mixed with 2 gallons of water was injected into the well. The dye was then followed with approximately 200 gallons of water.

Dye Recovery

Dye detectors were retrieved at weekly intervals throughout the course of the study. Dye detectors were eluted in a solution of potassium hydroxide and isopropyl alcohol and analyzed on a spectrofluorophotometer. Detector analysis results can be found in Table 2.

Dye Trace Results

The first sampling point to yield a confirmed positive was the sump in the basement of Mr. Gatti's. The dye was present on the detector collected July 27, 1989. The second round of detector

Table 2

SITE	7/14	7/21	7/27	8/3	8/9	8/16	8/24	9/1	9/9	9/15	9/26	10/10	10/19	10/27	11/5
RT	-	-	-	+V	+VB	+VB	X	X	X	X	X	X	X	X	X
MG	-	-	+	+VB	+VB	+VB	X	X	X	X	X	X	X	X	X
SS	VW	-	-	-	-	-	LS	+	+	+	+	+	+	-	+
RR	-	-	-	-	-	W	-	-	-	-	W	-	-	-	W
BE	-	-	-	-	-	+	+	-	-	+	-	-	-	-	-
WR	-	-	-	-	-	-	LS	-	-	-	-	-	LS	LS	-
VB	-	-	-	-	-	-	-	LS	-	-	LS	+	LS	LS	-
TB	-	-	-	-	-	+	-	LS	LS	-	LS	+	-	-	+
HC	-	-	-	-	-	-	LS	-	-	-	-	-	-	-	-
MS	-	VW	-	-	+	-	+	+	-	-	+	+	+	+	+
PC	-	-	-	-	-	-	-	-	+	-	+	W	+	+	+

DETECTOR RESULTS

-	=	dye not detected
VW	=	very weak
W	=	possible dye detected, weak
+	=	dye detected
+VB	=	dye detected, very strong
X	=	detector not collected as past + confirmed
LS	=	detector lost or stolen
DATE	=	detector collection date

NOTE: Test was conducted 7/14/89 to 11/15/89 collection on August 3, 1989 showed positive at Mr. Gatti's and in the recovery trench. The rapid movement of the dye to these two nearby sites demonstrates the relatively high ground water flow velocity associated with the solutionally enhanced upper bedrock (the epikarstic zone), which was the horizon of injection.

In the weeks following the initial confirmed positive at Mr. Gatti's and the recovery trench, there were a few "false positives". These false positives of fluorescein dye can be expected when tracing in urban areas since fluorescein in the coloring agent in many popular anti-freezes. These false positives are not a problem in a tracer study of this design. Since a large amount of fluorescein was injected upon the epikarst, one may expect a rather extended dispersal period over a large lateral area. Additionally, the detectors were retrieved at weekly intervals and a positive was not assumed until dye was detected for at least four continuous weeks at any particular site. Thus, false positives can be detected and noted accordingly.

There were several false positives during the course of this study. For example, detectors at Big Elm, Town Branch, and McConnell Spring showed false positives on several occasions. All false positives in the above sites were followed with negative readings in subsequent weeks. The storm sewer also displayed false positives throughout much of the study period. This site was adjacent to a road with heavy traffic, and undoubtedly encountered many urban-induced fluorescein spills. These spills were not great enough to cause positive readings at the railroad storm sewer a quarter mile directly downstream.

One focus of this study was to determine the ultimate resurgence of the injected dye. The detectors collected on September 26, 1989 marked the arrival of the first confirmed positive of the dye at McConnell Spring and Preston's Cave Spring sites. This first confirmed positive occurred some 67 days following the dye injection. The connection of McConnell Spring to Preston's Cave Spring was first documented by James Rebmann of the Lexington-Fayette Urban County Government in January 1988.

Dye introduced at the soil-bedrock interface moved very rapidly through the epikarst toward Mr. Gatti's and the recovery trench. The epikarst is noted for its rapid ground water flow velocity and wide lateral dispersion of recharge. After passing through the epikarst, it appears that the dye moved very slowly through a poorly developed conduit system toward McConnell Spring. After passing through McConnell Spring, the dye resurged at Preston's Cave Spring.

Ground Water Contamination

Ground water contamination, as detected at Mr. Gatti's and in the recovery sump has decreased from the presence of free product in April 1986 to dissolved contamination in the low parts per million range in December 1989. Fluctuations in contaminant levels have been documented during this time period. Increases can be attributed to heavy periods of recharge where ground water moving along poorly developed conduits in the epikarstic zone leach residual soil contamination. Decreases are marked by dry periods where precipitation has little effect in leaching residual soil contamination.

REMEDIATION DESIGN AND IMPLEMENTATION

In order to properly remediate the subcutaneous zone of this location, an approach different from classical pump and treat was needed. The design had to be specifically matched with the geological and hydrogeological conditions of the subcutaneous zone, satisfy state requirements of remeditaion to background, and satisfy the third party landowner that every possible avenue to remediation was being attempted using the fastest technology available.

When designing this recovery system we had to account for:

- contaminated soil above the soil/bedrock interface that was contributing to contamination;

- inaccessible contaminated soil in solutionally enlarged pore space of the subcutaneous zone;

- quickflow of contaminated ground water in poorly developed conduits of the subcutaneous zone;

- observed increases in BTEX concentration in ground water with an increase in recharge;

- available space and property boundaries; and,

- the possibility of volatile organic vapors in buildings.

SELECTED TECHNOLOGIES AND APPLICATION

Soil Excavation

Elevated levels of gasoline constituents were identified in an area of soils west of Mr. Gatti's Pizza Restaurant as outlined in Figure 5. Approximately 1960 tons of this soil was excavated from the area and transported to the Lexington-Fayette County Municipal Landfill.

Soil Flushing

Soil flushing was selected for this site for several reasons. The nature of flow along the soil bedrock interface is likely quick flow through poorly developed open and soil filled solution channels in the subcutaneous zone. Levels of dissolved gasoline components in ground water have shown a sharp rise following periods of heavy recharge from precipitation. Reinfiltration of treated ground water was designed to mimic a constant recharge to this zone, thereby continually flushing contaminants from the soil in the epikarstic zone. Soil flushing is also very cost effective, provides an option for the fate of the treated ground water and can increase and enhance ground water flow to the interceptor trench.

An infiltration gallery was designed to accept and reinfiltrate the volume of discharge generated by the ground water recovery and treatment system. At this site discharge will be the result of ground water recovered from a maximum of two continuously operational recovery sumps.

The purpose of the infiltration gallery is two fold. First reinfiltrated water serves to recharge the perched surficial epikarst and move any ground water contamination to the interceptor trench more quickly. Second, reinfiltrated ground water flushes residual soil contamination at the soil bedrock interface causing the leached material to become mobile in ground water for removal at the interceptor trench.

During the tracer study conducted July 1989, a small conduit was discovered at the soil bedrock interface at a location shown on Figure 2. This conduit was easily able to accept at least 100 gallons of water instantaneously. This conduit was also traced by fluorescent dye to the location of the present recovery trench in less than 24 hours. The infiltration gallery intersects the conduit in the epikarst therefore providing constant recharge to the surficial flow system and constant flushing of residual contamination from soil.

The gallery is 40 feet long, 3 feet wide and 7 feet deep. The discharge water is introduced by means of 4" diameter perforated pipe to allow drainage across the entire trench. The trench is backfilled with washed gravel to allow for even percolation of the treated ground water. Two clean-out ports have been placed on the perforated pipe to remedy any sediment, clogs or biofouling. Two piezometers were installed at opposite ends of the infiltration gallery to monitor its efficiency in reintroducing ground water. Emergency high level floats keep the infiltration gallery from over flowing if its capacity is exceeded. A schematic of the infiltration gallery is shown in Figure 7. The gallery was installed during the second week of January 1991. To ensure the gallery was hydraulically "connected" to water flow in the epikarstic zone approximately 1000 gallons of water was introduced to the open hole after it was excavated. The water quickly flowed from the gallery and into the subcutaneous zone.

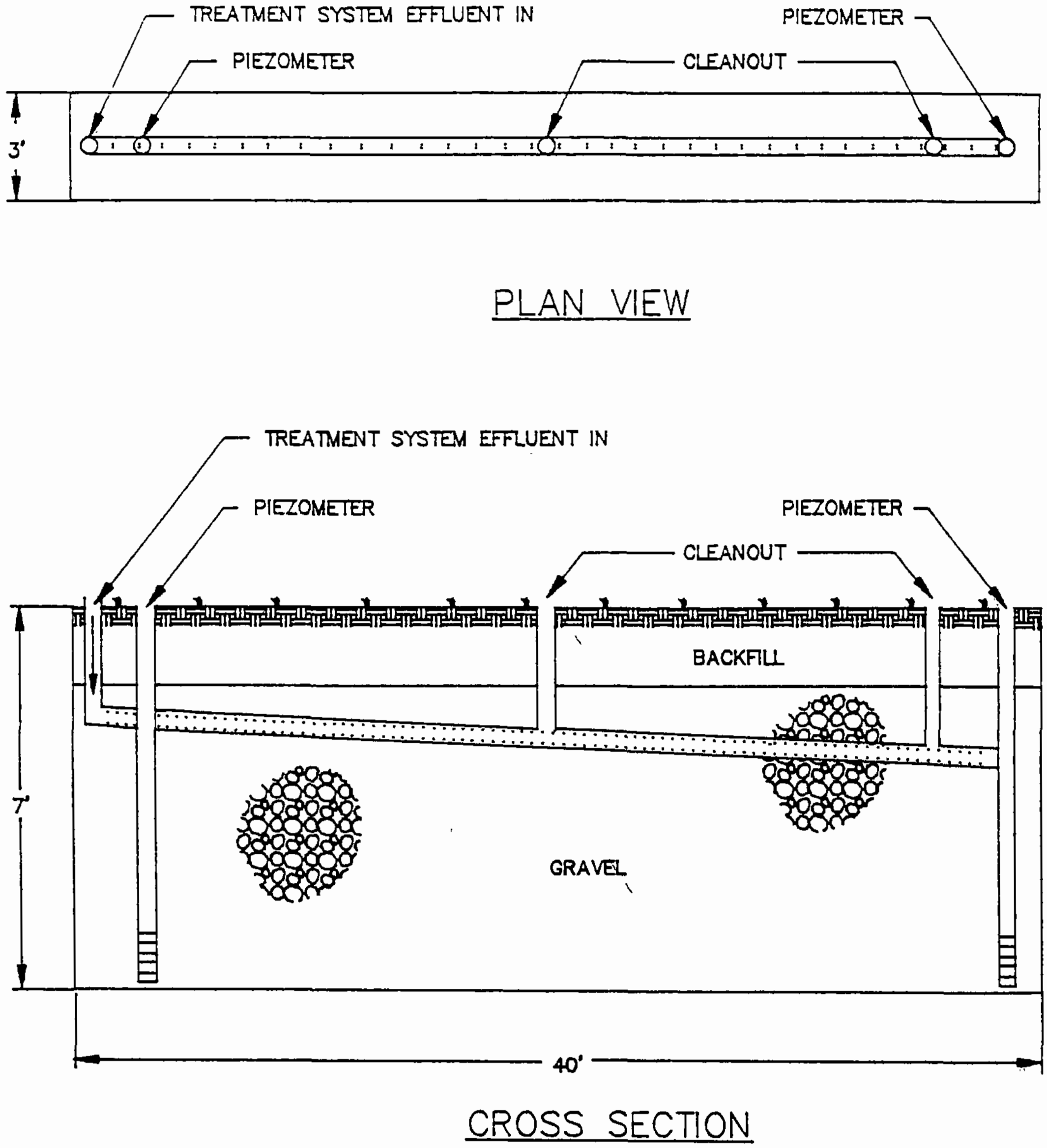


Figure 7. Infiltration gallery design.

Soil Venting

Soil venting was also a selected technology for implementation at this site for this site for several reasons. Soil venting allows for remediation of the residual soil contaminants with minimal disturbance of surface structures and roads. Soil venting is a cost-effective and field proven technology for remediating soil contaminated with petroleum hydrocarbons. Application of a vacuum to the interceptor trench should also increase the hydraulic yield of the trench and enhance the rate of ground water remediation. The primary reason, through, was to remove hydrocarbons from soil is the epikarst that could not be removed by excavation.

The soil vapor extraction system consists of a regenerative blower connected via 2-inch schedule 40 PVC piping to the ground water recovery/vapor extraction trench. The trench is screened through the vadose zone to accommodate the vapor removal. Depression of the potentiometric surface due to pumping of the ground water also increases the volume of contaminated soil exposed for remediation by venting.

Soil Remediation System Equipment

Figure 8 presents the layout for the equipment of the soil venting system which consists of a regenerative blower connected to the recovery well via the previously described PVC piping, an inlet filter to remove particulates from the air stream entering the blower, and a coalescing unit to remove moisture from the air stream entering the blower. The effluent from the blower will be discharged to the atmosphere via a 2-inch schedule 40 PVC pipe. As the liquid level in the coalescing unit reservoir increases, it activates a sensor which turns off the blower and opens a solenoid valve in the coalescer, thereby allowing the condensate to gravity drain into the diffused aeration tanks. A timer in the control panel reactivates the blower and closes the solenoid valve after all condensate has drained.

GROUND WATER RECOVERY AND TREATMENT

Ground Water Recovery Equipment

The ground water recovery system installed in 1986 at the Lexington site consisted of a ground water interceptor trench and one recovery sump. The former interceptor trench was extended and an additional recovery sump be added as shown in Figure 9. The ground water recovery system consists of two total fluids recovery pumps installed in the recovery sumps of the enlarged interceptor trench. The pumps intercept and pull contaminated ground water towards the interceptor trench. A cross section of the trench construction is shown in Figure 10.

The static depth to ground water in the recovery sump installed in 1986 was approximately 8 feet below grade. Both recovery pumps are positioned at a depth of 11 feet. The depression pump on/off floats are positioned at 11 feet and 10 feet below grade respectively in order to establish a maintained drawdown of approximately 2 feet in the interceptor trench. The operation of the total fluids pump is controlled by high and low level on/off sensors in order to facilitate the efficient removal of contaminated ground water from the soil bedrock interface.

Ground Water Treatment Equipment

The ground water treatment technology implemented at this site is a three step process:

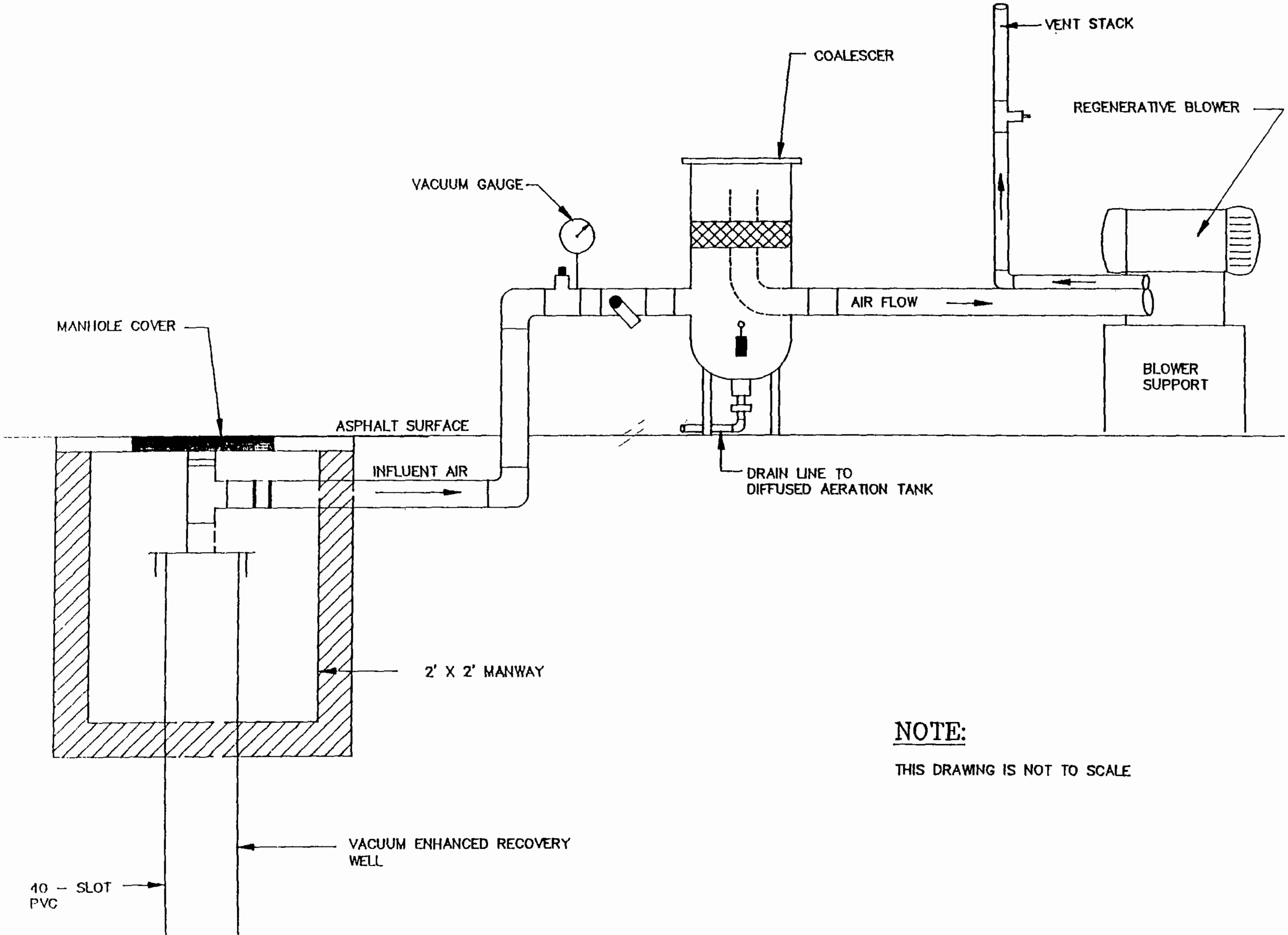


Figure 8. Soil venting system schematic.

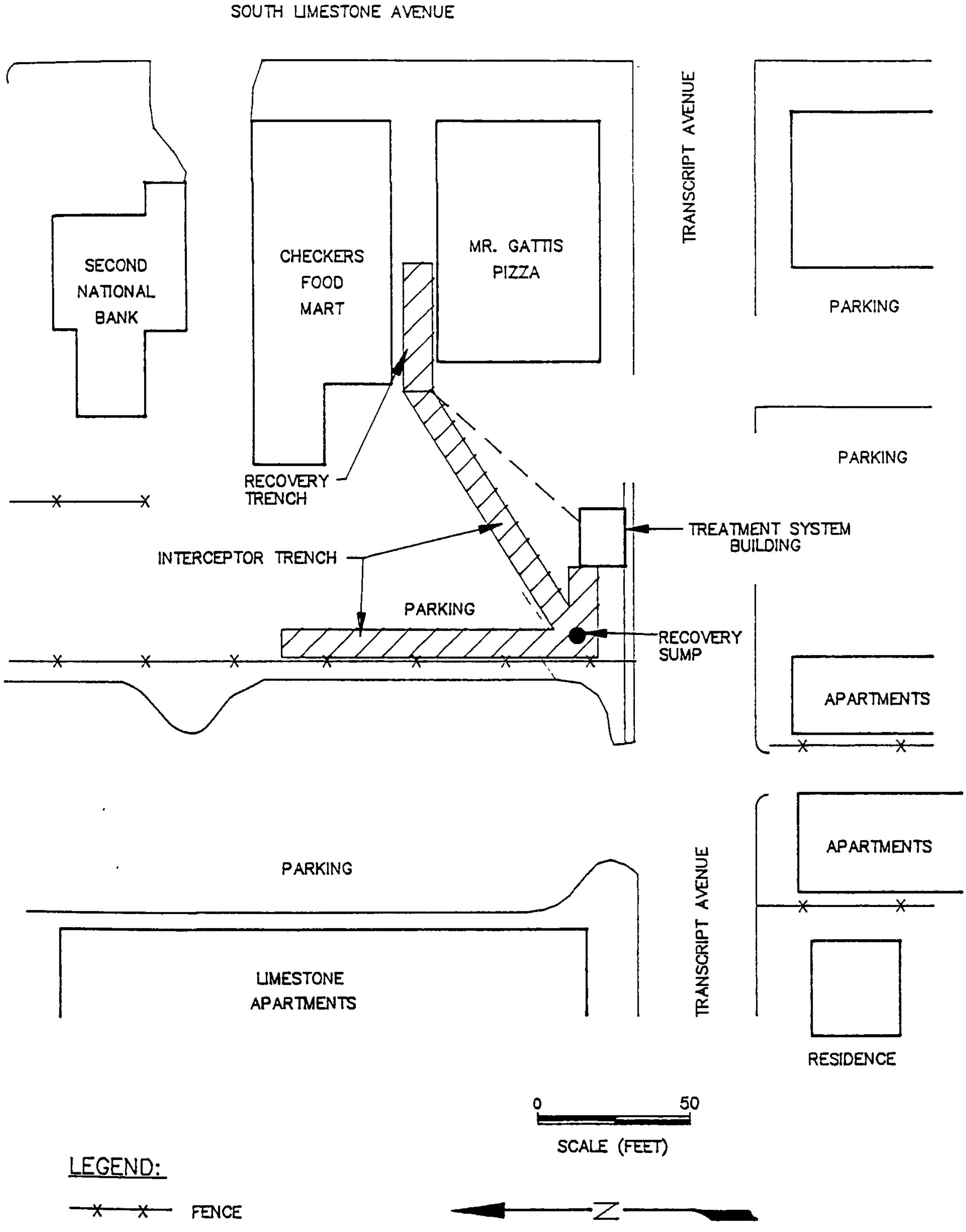
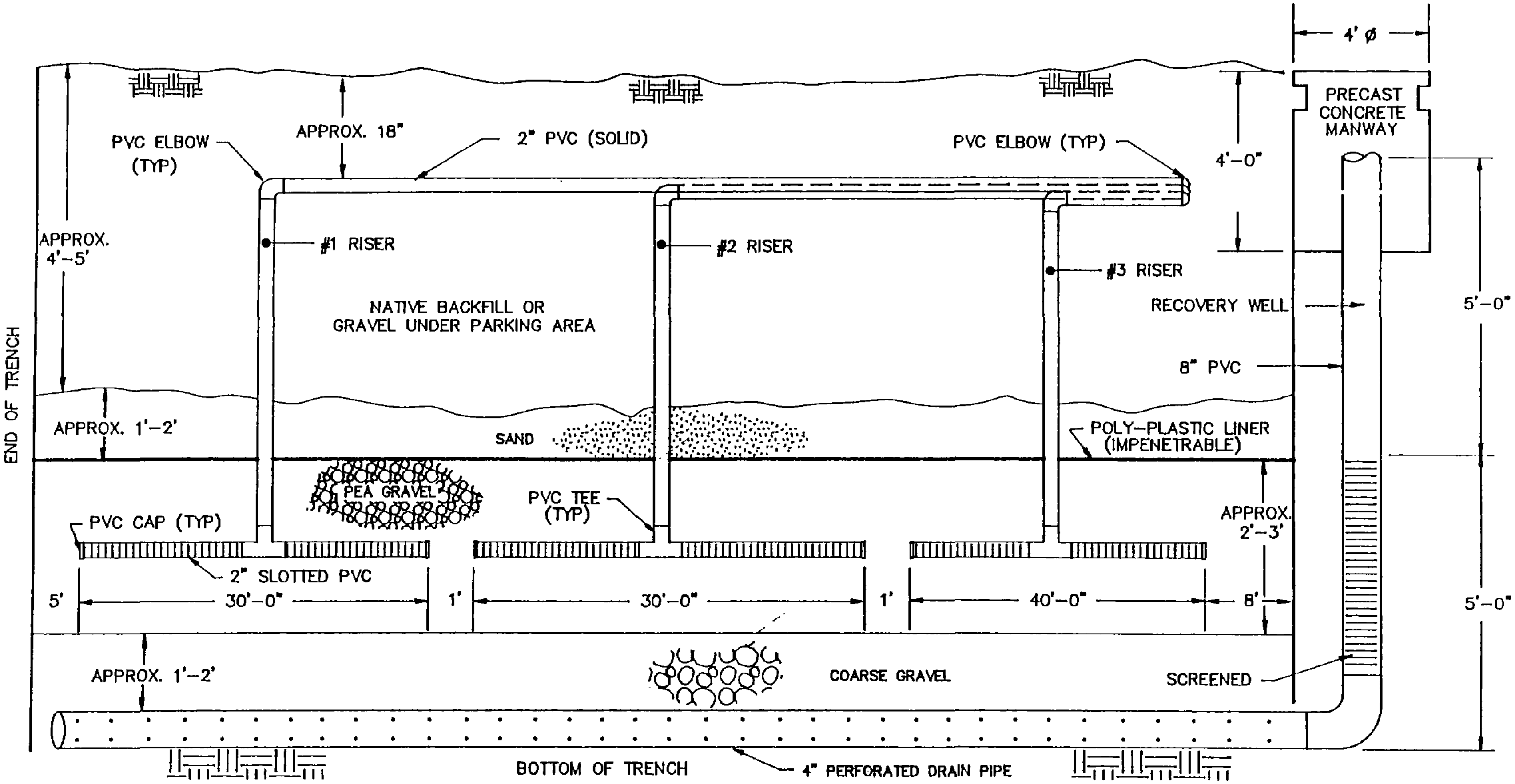


Figure 9. Recovery trench and treatment system location.



NOTE: DRAWING IS NOT TO SCALE.
ALL PVC IS SCHEDULE 40

Figure 10. Recovery trench construction details.

1) particulate filtration as a pretreatment step to remove suspended solids;

2) LPH/water gravity separation as a pretreatment step to provide for the flotation separation of LPH and flow equalization; and

3) diffused aeration to remove the dissolved volatile organic compounds.

Pretreatment Process

The ground water pretreatment system consists of an in-line particulate filter followed by an LPH/water separator tank. Figure 11 presents the treatment system lay out.

The ground water recovery pumps transfers ground water through an in-line filter to a 1000 gallon LPH/water separator tank to allow for flotation separation of the LPH from the ground water and particulate settling. A totalizing flow meter installed at the intake of the separator tank continuously records the amount of ground water and LPH that has been pumped.

Effluent from the separator tank flows by gravity through a contact chlorine chamber containing solid calcium hypochlorite tablets. This process reduces biological build up in the treatment system. The chlorination step inhibits the growth of biological microorganisms that could potentially cause the lines in the aeration tanks to become clogged.

Diffused Aeration Equipment

Effluent from the contact chlorine chamber will flow by gravity into a channeled diffused aeration tank. This tank will be four feet wide by six feet long by two feet deep. A 2 hp, 230V, single phase pressure blower will introduce air into the aeration tanks through perforated PVC pipes to facilitate stripping of the dissolved volatile organics.

Effluent from the diffused aeration tanks flows by gravity into a 120 gallon transfer tank. This transfer tank is equipped with a submersible electric sump pump with a nominal operating flow rate of 10 gpm. The sump pump moves the treated ground water from the transfer tank to an infiltration gallery located upgradient from the interceptor trench.

Soil Venting and Ground Water Remediation System Security

The soil venting and ground water remediation equipment are housed inside a pre-fabricated building located at a location shown in Figure 9.

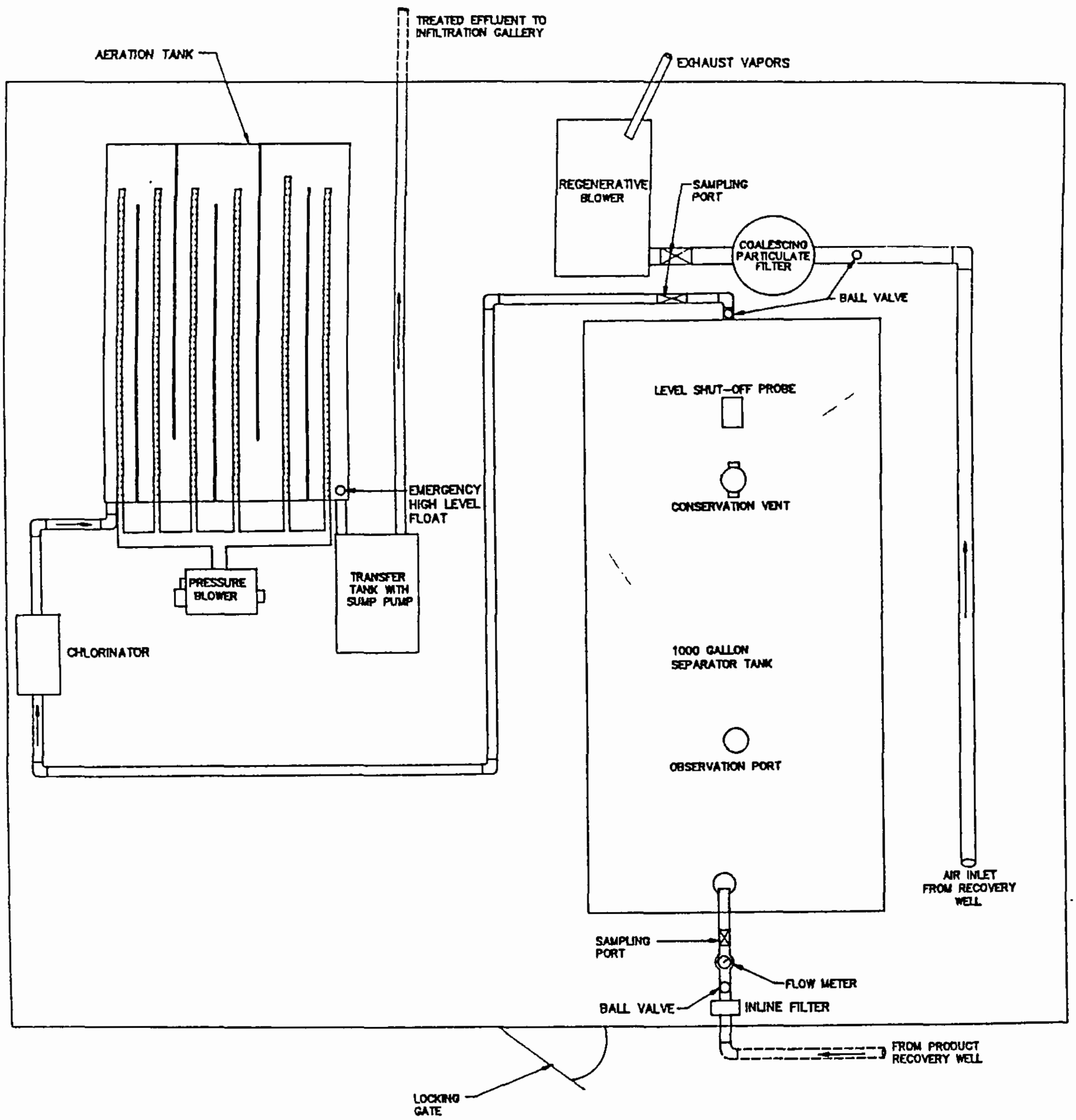


Figure 11. Treatment system layout.

RESULTS TO DATE

Analytical results from water samples collected from the former emergency sump and the current ground water recovery system are plotted against time in Figure 12, illustrating a wide variation in contaminant concentrations since recovery operations have begun. Maximum dissolved concentrations of approximately 15,000 ug/L total benzene, toluene, ethylbenzene and total xylenes (BTEX) have been recorded. Although flow readings do not exist for the early recovery system, a general correlation between recharge events and increasing contaminant concentratons has been made. The emergency recovery sump was taken off line from July 1989 through February 1990 to conduct the dye tracer study.

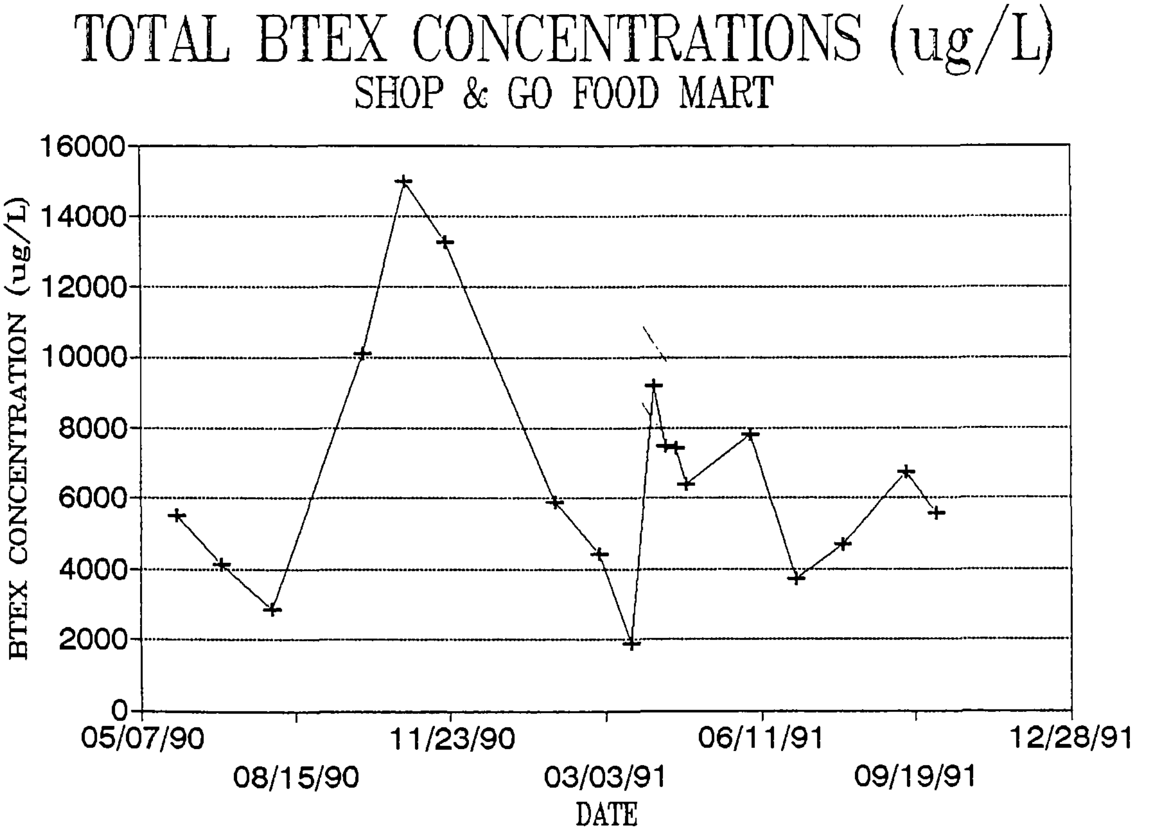


Figure 12. Total BTEX concentrations in treatment system influent since May, 1990.

The current ground water remediation system was monitored monthly to verify that the system is operating properly. Sample ports are provided at several locations along the treatment system. Water from the treatment system influent and effluent sampling ports are collected monthly, and analyzed for volatile organic constituents.

from the treatment system influent and effluent sampling ports are collected monthly, and analyzed for volatile organic constituents. A totalizing flow meter is installed to document the total volume of ground water being pumped from each recovery sump.

Plots of flow volume over time from both recovery sumps are presented as Figures 13 and 14. These illustrations verify the flashy nature of flow in the subcutaneous zone. Flow rates vary from recovery well 1 from almost negligible to approximately 5500 gallons per day. Flow variations from recovery well 2 are even more dramatic ranging from 3000 to almost 25,000 gallons per day. This information also correlates with periods of observed heavy rainfall events through that time period.

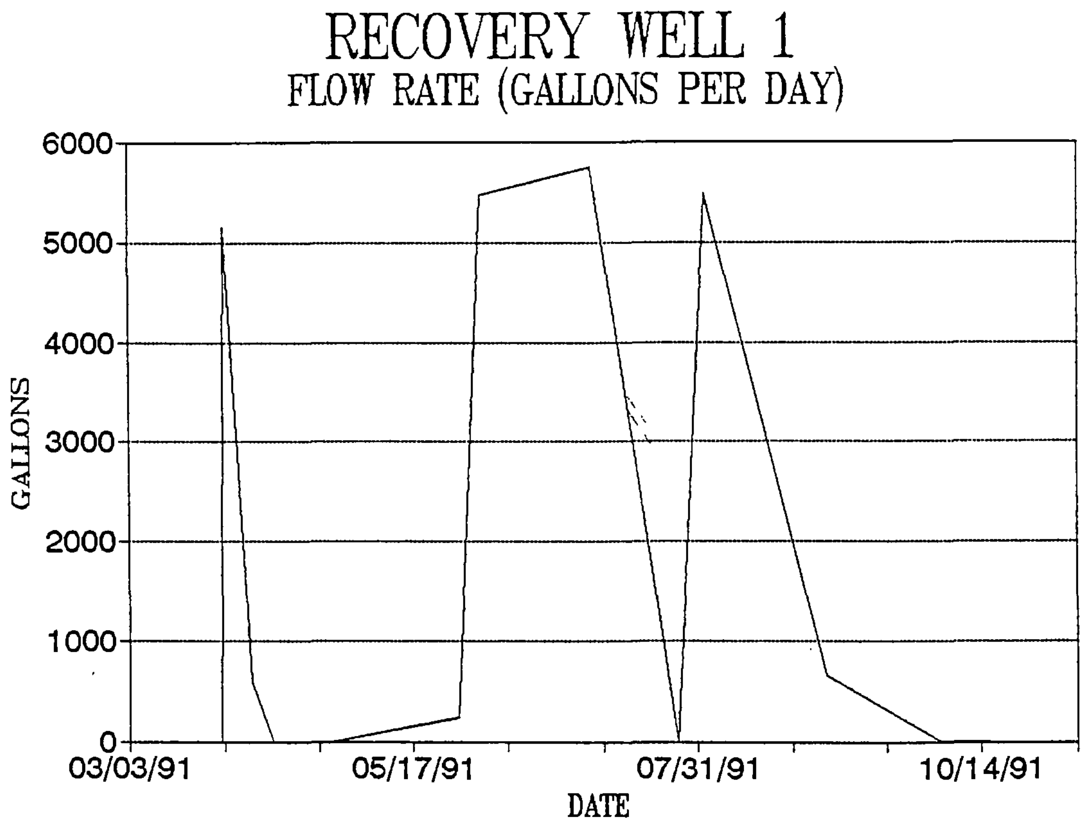


Figure 13. Flow in gallons per day from Recovery Well 1 since system start-up.

A plot of BTEX concentration in the treatment system influent since the currently system was put on line included as Figure 15. Figure 15 shows that the concentration of total BTEX varies widely but the maximum measured levels have deceased since remediation began. The peak concentrations are also roughly similar to maximum times of flow or maximum recharge. This may suggests that more contamination is being leached from soils, mobilized and removed during times of heavy recharge and that the constant recharge from the infiltration gallery may aid in expediting this removal process.

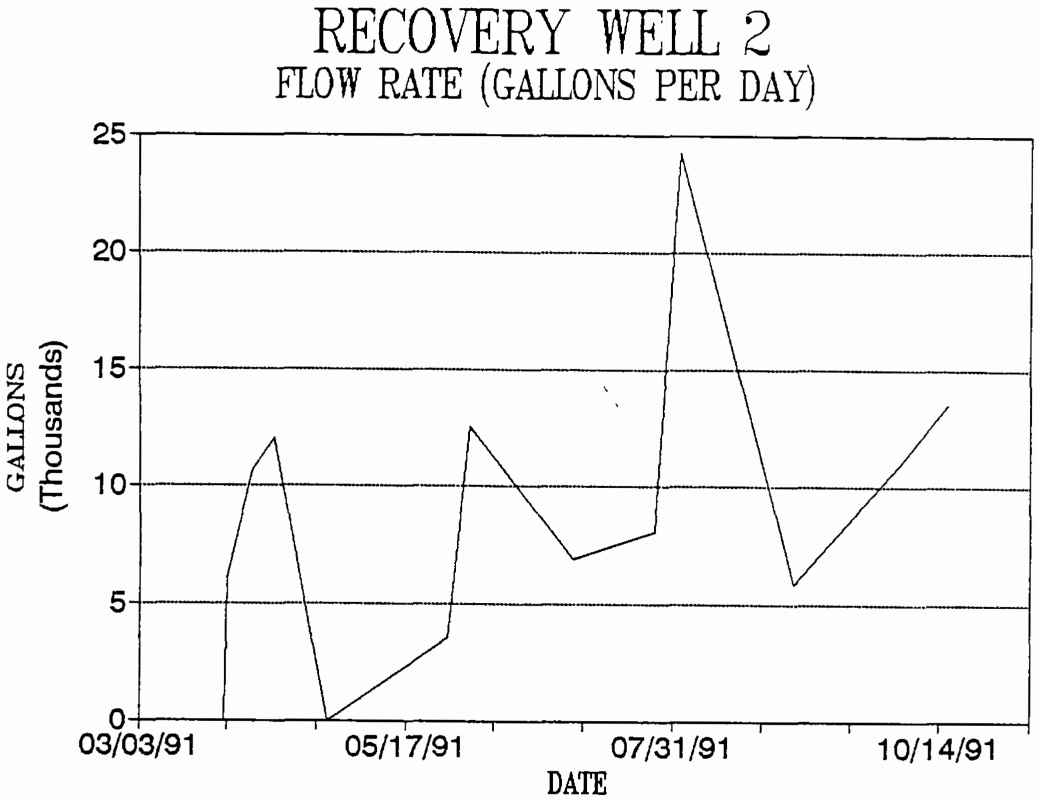


Figure 14. Flow in gallons per day from recovery well 2 since system start-up.

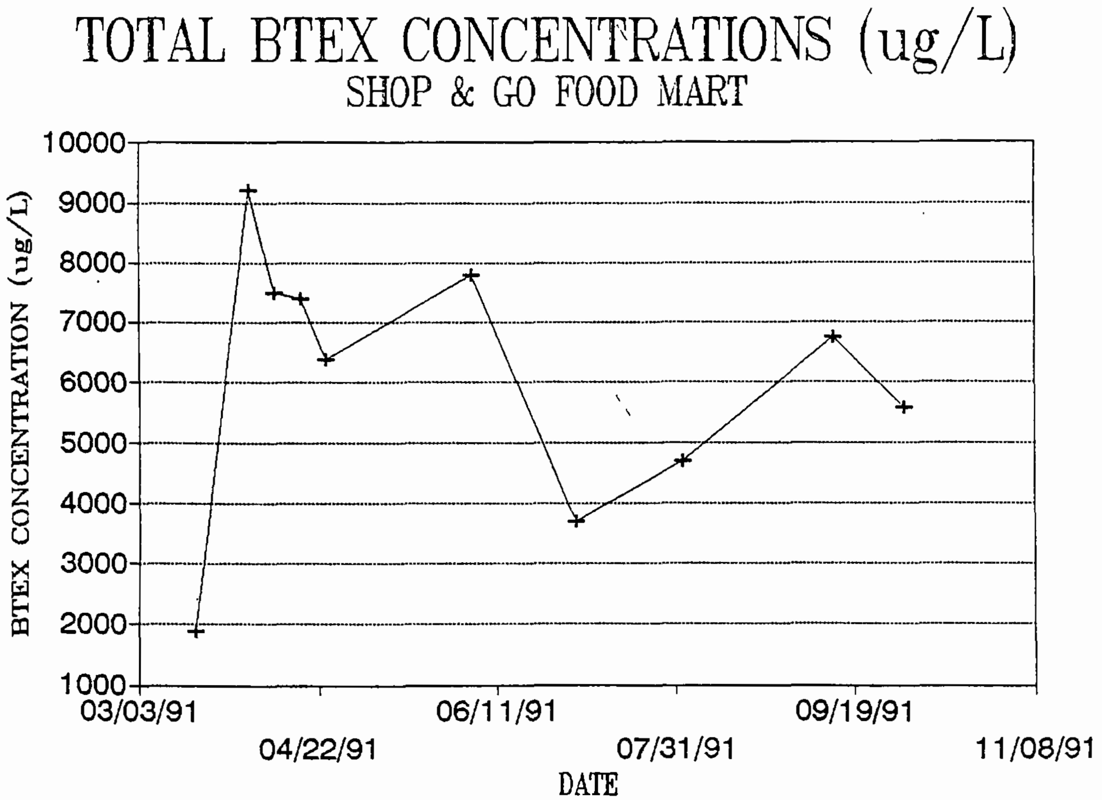


Figure 15. Total BTEX concentration in treatment system influentsince system start-up.

The performance of the soil venting system is illustrated in Figure 16 as a plot of soil vapor concentration since the system was put on line. Soil vapor concentrations were at a maximum when the vacuum blower was first started and have decreased to non-detectable since that time. The reason for the quick decrease is believed to be removal of BTEX laden soil vapor from the immediate area of the vapor extraction equipment. The system has since been turned off to allow equilibration of soil vapor with BTEX on soil particles. The system will then be put back on line and a decreased but similar pattern of soil vapor removal is expected.

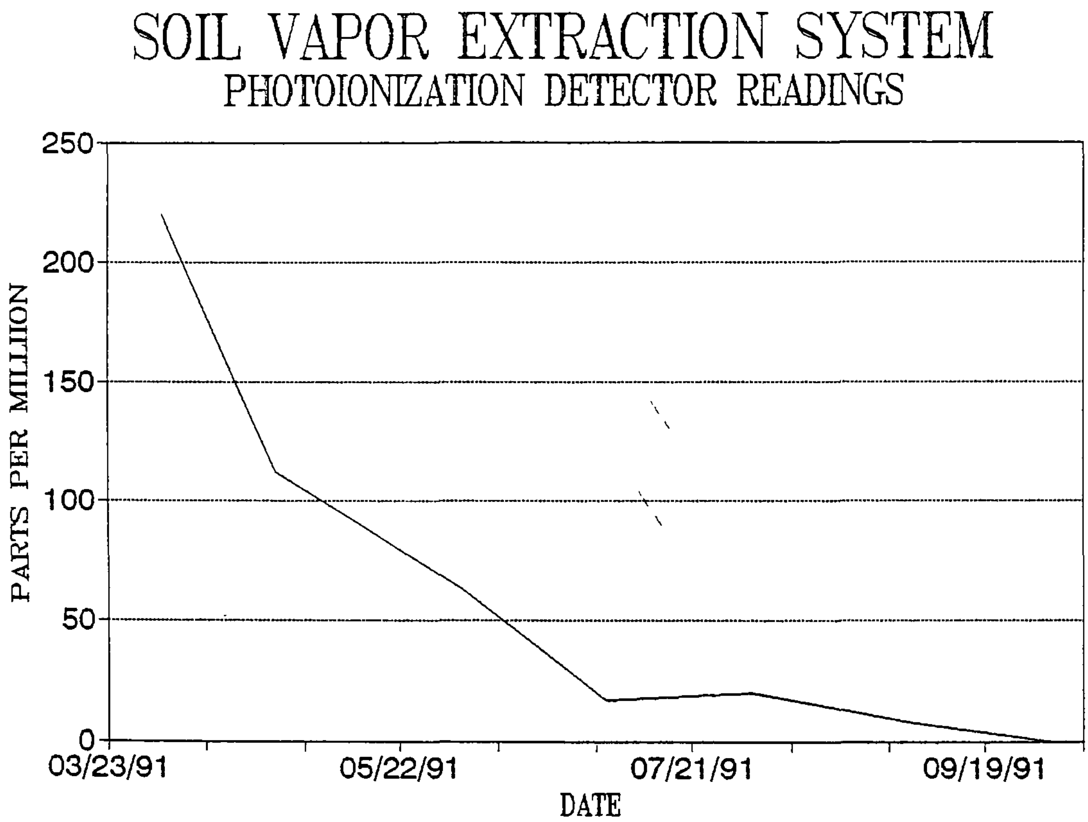


Figure 16. Photoionization detector readings of the soil vapor extraction emissions since initiation.

CONCLUSIONS

The likelihood of a release of contaminants directly into the conduit network of a karst aquifers is low. Most releases occur within the overburden and move into the subcutaneous zone of the aquifer thereby decreasing the quickflow components and increasing residence times in the intergranular overburden porosity and the poorly developed conduit system of the subcutaneous zone. The results of this effort have been presented as a site specific approach to remediate the subcutaneous zone of a karst aquifer. The remediation design was not arrived at using classical pump and treat, intergranular flow manner approach to the problem and set aside the notion that this karst aquifer was not "remediateable" as if it were conduit network contamination. A remediation system was designed around the specific characteristics of the surficial subcutaneous karst aquifer to be effective as a remediation tool in this setting. The use of excavation, soil flushing, soil venting and pump and treat has been shown effective at removing soil and ground water contamination from the s ubcutaneous zone as a decrease in maximum contaminant concentration over time. The system also appears to be working as designed by mimicking a constant recharge event thereby leaching residual soil contamination from the subcutaneous zone for removal by the interceptor trench. The system is currently operating as designed and data being collected for a future submittal concerning its efficiency and performance.

REFERENCES

Miller, R.D., 1967, Geologic Map of the Lexington West Quadrangle, Fayette and Scott Counties, Kentucky; United States Geological Survey Geological Quadrangle Map - GQ - 600.

BIOGRAPHICAL SKETCH

Scott A. Recker, a native of Cincinnati, completed a Bachelors Degree in geology at the University of Cincinnati in 1985 and received a Master of Science in hydrogeology from Eastern Kentudy University in 1990. Scott is a member of the National Speleological Society and his research centers around ground water flow in the subcutaneous zone of karst aquifers. He is currently residing in Charlotte, North Carolina and is employed as a consulting hydrogeollogist for Delta Environmental Consultants, Inc.

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PETROLEUM HYDROCARBON REMEDIATION OF THE SUBCUTANEOUS ZONE OF A KARST AQUIFER, LEXINGTON, KENTUCKY

Scott A. Recker
Delta Environmental Consultants, Inc.
Charlotte, North Carolina

1. Has the remediation effort been evaluated at the spring?

No, the spring showed non-detctable amounts of petroleum hydrocarbons during two sampling events. The spring has also been documented as the ulimate resurgence for a ground water basin which encompasses half of the City of Lexington. Remedial evaluations at the spring would therefore be unsuccessful due to the large amount of dilution and the presence of other potential petroleum hydrocarbon constributors.

CORRECTION OF BACKGROUND INTERFERENCE AND CROSS-FLUORESCENCE IN FILTER FLUOROMETRIC ANALYSIS OF WATER-TRACER DYES

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ABSTRACT

Filter fluorometry is a widespread analytical technique for the quantitative analysis of fluorescent tracer dyes in hydrogeology. However, mixtures of tracers and fluorescent background materials may result in ambiguous results unless steps are taken to isolate the tracer materials. Analytically it may be possible to chemically suppress or mask interfering substances, or to select filter combinations capable of excluding those wave-lengths frustrating analysis.

Post-analytically, an algebraic solution is possible, providing care has been taken to run calibration standards ({y}=1,2,3) through all filter arrays ({x}=A,B,C) to obtain a matrix of fluorescence readings F^x_y. The matrix of cross- fluorescence calibration coefficients $\{k^x_y\} = F^x_y/k^z_y$ can be determined, where z= A, B, C for y=1,2,3 respectively. By definition, the positive diagonal k^x_y values will always equal one. Off-diagonal k^x_y may be zero (indicating absence of cross-fluorescence), or non-zero (indicating cross-fluorescence and the need for correction). Note that exact values of k^x_y require empirical determination, because they depend on characteristics of dye stock, individual filters, and light sources.

Table of Cross-Fluorescence Calibration Coefficients $\{k_y^x\}$		Dye (y) 1	2	3
Filter Set (x)	A	k_1^A	k_2^A	k_3^A
	B	k_1^B	k_2^B	k_3^B
	C	k_1^C	k_2^C	k_3^C

$k_1^A=k_2^B=k_3^C=1$

If no cross-fluorescence $k_y^x=0$

If tracer dyes 1, 2 and 3 are matched to filter sets A, B and C respectively and calibration allows $k_1^A=k_2^B=k_3^C=1$, then an approximate but general solution to the cross-fluorescence problem can be derived. The actual fluorescence of dye 1 in a sample (F_1^A) can be expressed in terms of measured fluorescence of that sample in all three filter sets (F^A,F^B,F^C), and the coefficients $\{k_y^x\}$:

$$F_1^A = \frac{F^A - F^B(k_2^A - k_3^A k_2^C) - F^C(k_3^A - k_2^A k_3^B)}{1 - k_2^A k_1^B - k_3^A k_1^C} \tag{10a}$$

The relationship can be further simplified for two-component mixtures, to produce the common correction equation given by Käss (1967):

$$F_1^A = \frac{F^A - k_2^A F^B}{1 - k_2^A k_1^B} \tag{11}$$

Where cross-fluorescence coefficients are small numbers, further simplification is possible:

$$F_1^A = F^A - k_2^A F^B \tag{12}$$

The model will be robust for all fluorescent materials with a consistent spectral signature. This may allow correction for background interference from organic materials, provided that background variations are due to fluctuations in spectral amplitude rather than shifts in spectral character. The background "standard" used to determine k_y^x must be entirely free of tracer dye, but can be of arbitrary concentration.

The technique will not allow cross-fluorescence correction in waters containing fluorescent contaminants closely mimicking dyes, nor accommodate dramatic shifts in background signature. These problems may be particularly acute in elutant from activated charcoal, which must remain a semi-quantitative technique.

Where precise interpretation is required, a spectrofluorometer remains the instrument of choice. The same method may be applied to correct for cross-fluorescence.

INTRODUCTION

Fluorescent dyes are an invaluable tracing agent, widely used in surface and groundwater studies. They are inexpensive, benign (Smart 1982) and readily detectable at concentrations below the parts per billion (ppb) level. It is also possible to use several dyes simultaneously, allowing several components of a hydrological cycle to be monitored under a given set of flow conditions. However, in practice, multi-tracing can be complex, expensive and possibly ambiguous, especially if tracers can not be completely separated from one another (cross-fluorescence) or from natural background materials. In practical applications, this will sometimes be seen as exact, but highly attenuated mimicking of a primary breakthrough curve in a neighbouring spectral trace. Where the two tracers have been injected simultaneously, it may be of considerable importance to determine if the attenuated trace is real,or simply the result of cross-fluorescence.

The problem is illustrated in Figure 2 where cross fluorescence from Dye 1 has caused an increase in the apparent fluorescence of Dye 2 (broken line). The exact magnitude of the problem will depend on the overlap of the spectra and the relative concentrations of the two tracers. Also note that cross-fluorescence is "non-commutative"; despite the interference of dye 1 with measurements of dye 2, measurements of dye 1 are little influenced by concentrations of dye 2.

The problem of two-component cross-fluorescence in spectrofluorometry has received some attention in the European literature. The present paper first reviews a correction methodology and then provides an initial attempt to formalise the problem for multi-component cross-fluorescence, with particular emphasis on filter fluorometers.

There are two strategies to solving cross-fluorescence problems. Pre-analytical treatments are designed to mask or suppress fluorescence of interfering tracers in mixed samples without altering the fluorescence of a selected tracer. Behrens (e.g. 1982) has derived an array of such treatments such as filtration and pH control allowing spectacular segregation of dyes prior to analysis. Further segregation may be gained using chromatographic separation techniques (e.g. Rochat et al. 1975). In general, these techniques require relatively sophisticated processing and although automated techniques may be adopted, remain prohibitively expensive for most environmental investigations.

The alternative strategy where moderate cross-fluorescence occurs in mixtures of two or three tracers is post analytical. It involves application of an algebraic correction (Käss 1967) based on the theory of mixtures. This requires very simple cross-calibration of the fluorometer using single dye standards. Cross-calibration is best performed at the time of analysis, but in emergencies can be done some time subsequent to the analysis if necessary. The method is straightforward and inexpensive, and should be adopted in any multi-tracer test.

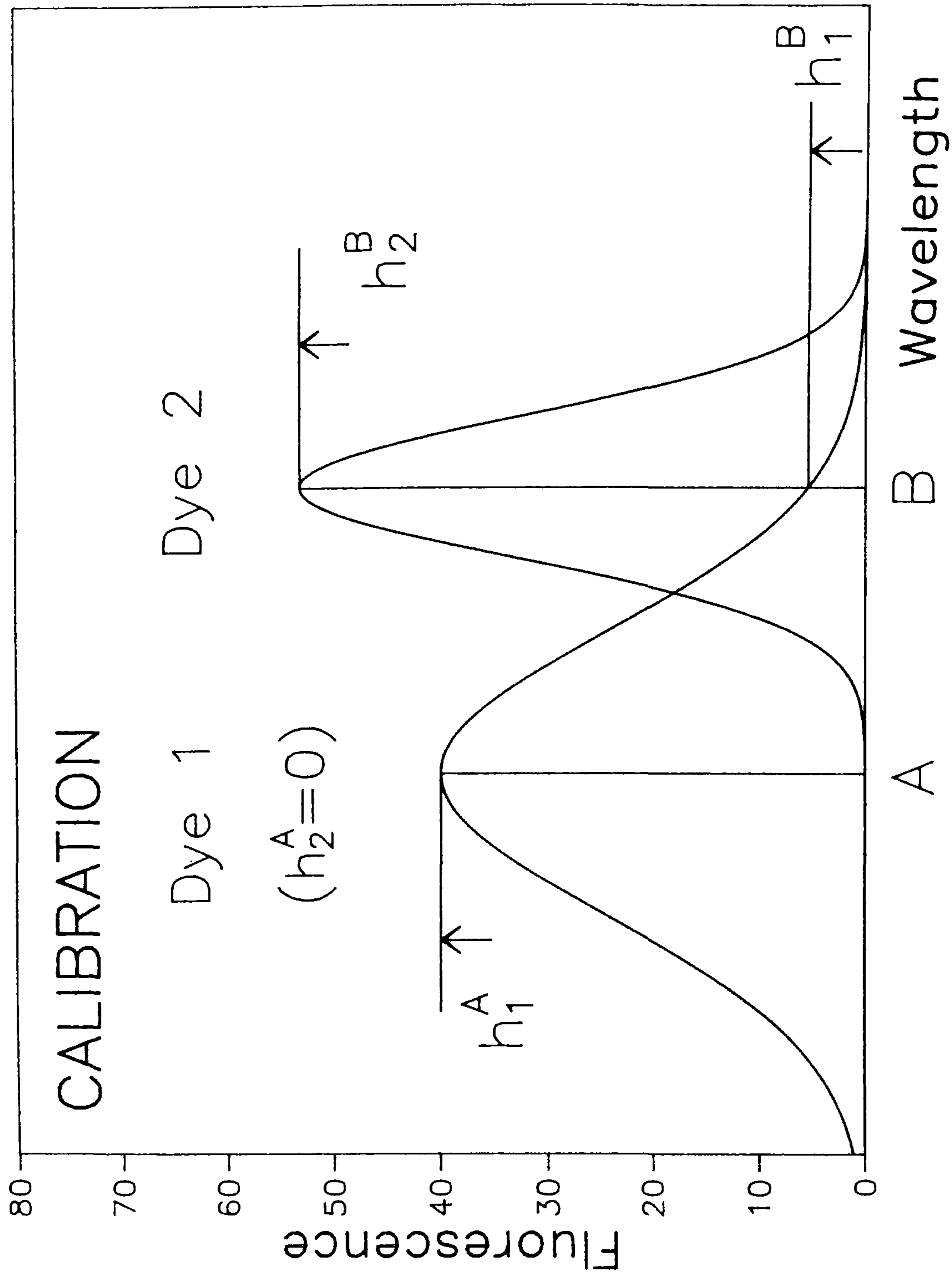

Fig. 1. Schematic diagram of two component cross-fluorescence in a spectro-fluorometric analysis. The h_y^x heights collected from single dye standards are the calibration data needed to correct for cross-fluorescence.

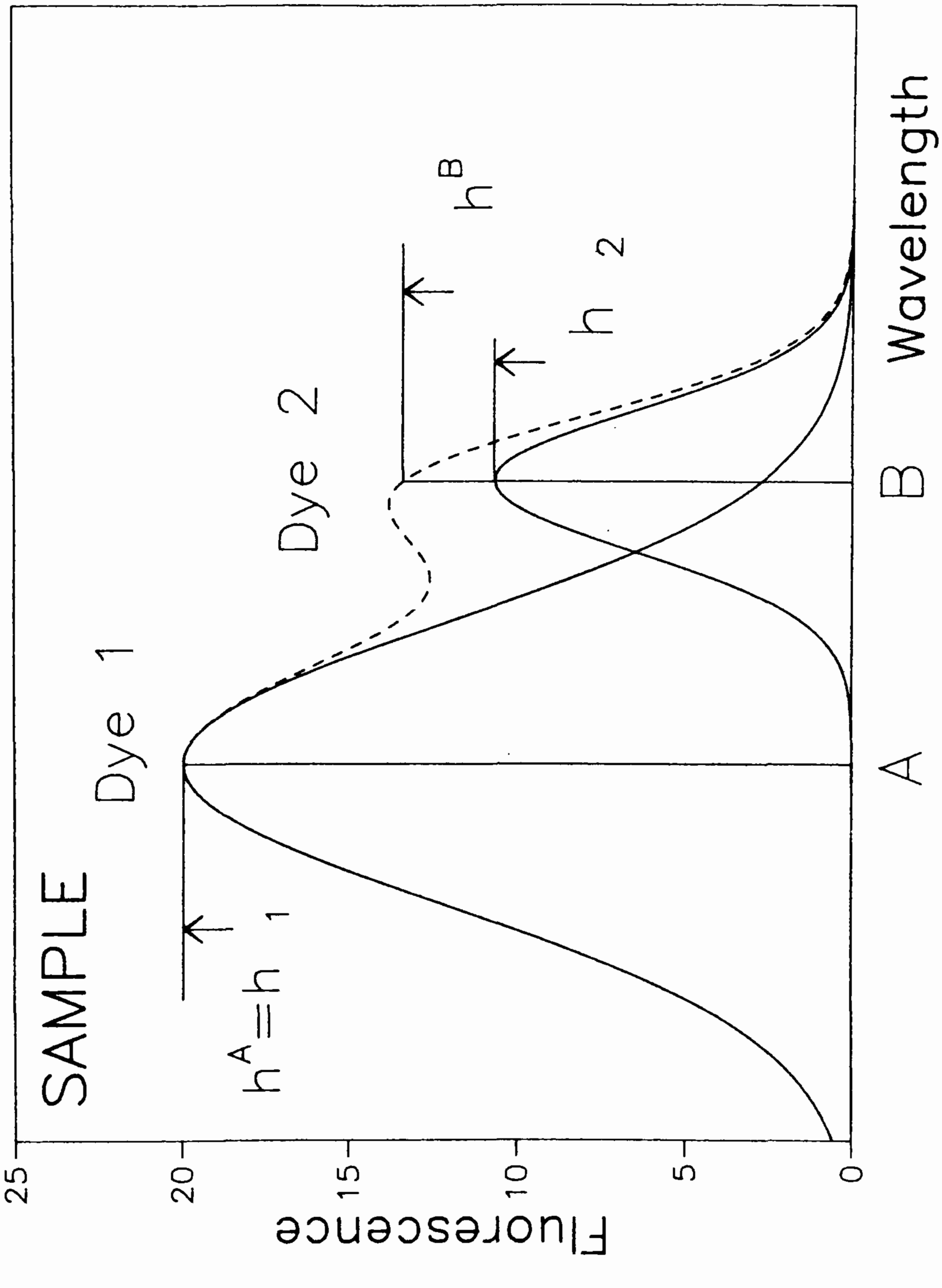

Fig. 2. Schematic diagram of fluorescence spectrum obtained from spectrofluorometric analysis of a water sample containing two dyes. The broken curve indicates the measured trace, the solid curves the underlying spectra of the two dyes. The h^x values are the measured spectral heights, the $\hbar_y$ values are the desired correct spectral heights to obtain true readings of the two components.

CROSS-FLUORESCENCE: SPECTROFLUOROMETERS

A fluorescent material absorbs light over a characteristic waveband and re-emits light at a characteristically lower waveband. In fluorometric analysis a sample is irradiated (excited) with light of wavelength within the absorption spectrum of interest, and emittance measured at an exclusive longer wavelength. The key to effective measurement is provision of suitable, but mutually exclusive excitation and emission wavelengths. In spectrofluorometers, this is achieved by very precise control of the wavelengths using diffraction gratings.

A spectrofluorometric analysis is usually undertaken by scanning with excitation and emission wavelengths a fixed distance apart (typically ~ μm). This produces a typical spectral trace of overlapping peaks where peak position is characteristic of particular fluorescent materials and peak height its concentration. Unfortunately, fluorescence is additive so that if the tail of one spectral peak underlie the peak of another, the peak height is no longer a simple measure of concentration of a single material. This is the problem of cross-fluorescence. In worse cases, the overlap may result in complete masking of smaller peaks in the flanks of more dominant ones. The problem becomes more acute at shorter wavelengths when the very broad fluorescence of naturally occurring organic materials may underlie tracer peaks.

In spectrofluorometry, the problem of cross-fluorescence is usually readily apparent in a compound form to the spectral trace. For two component mixtures, it may be corrected using the method of Käss (1967). As there is no formal statement readily available on the correction procedure, it is summarised as follows. First, single tracer standards are run, noting the height of the peak and shoulder at the characteristic peak wavelengths for both dyes. A matrix of trace height readings is thus obtained for each dye (1,2) at each diagnostic wavelength (A,B). For example, Figure 1 gives an array as follows:

Calibration Measurements		Dye	
		1	2
Wavelength	A	h_1^A=39.9	h_2^A=0.00
	B	h_1^B=5.40	h_2^B=53.2

The convention introduced here is that the superscript (A,B...) indicates the wavelength(s) used, and the subscript (1,2,...) the dye of interest.

Cross-fluorescence correction factors are the ratios of spectral readings to peak height. They are thus independent of concentration, and are obtained by dividing each column by the measurement obtained for the matched wavelength-dye pair (Dye 1-

wavelength A; dye 2-wavelength B). Thus,

Cross Fluorescence Calibration Coefficients $\{k_y^x\}$		Dye(y) 1	Dye(y) 2
Wavelength (x)	A	$k_1^A=1.00$	$k_2^A=0.00$
	B	$k_1^B=0.135$	$k_2^B=1.00$

The cross-fluorescence calibration factors are generalised as $\{k_y^x\}$ where x indicates the wavelength (normally the spectral peak for a dye), and y indicates the dye of interest. Note that $0 \leq k_y^x \leq 1$, the positive diagonal members will always be unity. Off-diagonal values will be zero where there is no cross-fluorescence (eg. k_2^A above), and non-zero (usually small numbers) when cross-fluorescence occurs. Käss (1967) and others (e.g Behrens 18982) employ α and ß to indicate these correction factors, but their derivation may not be clear and their meaning can be ambiguous.

Figure 2 provides an example of a sample analysis, where h^A and h^B are measured peak heights from a sample and other terms are defined above. The following estimates of actual peak heights $\hat{h}_{1,2}$ may be obtained by

$$\hat{h}_1 = \frac{h^A - k_2^A h^B}{1-k_2^A k_1^B} = \frac{20.0 - 0.0 \times 13.4}{1 - 0.00 \times 0.102} = 20.0 \tag{1}$$

i.e there is no cross-fluorescent interference with dye 1. For dye 2

$$\hat{h}_2 = \frac{h^B - K_1^B h^A}{1-k_2^A k_1^B} = \frac{13.4 - 0.135 \times 20.0}{1 - 0.00 \times 0.135} = 10.7 \tag{2}$$

In this case a 20% correction has been applied to the raw data. A formal derivation of this simple case is developed below.

CROSS FLUORESCENCE: FILTER FLUOROMETERS

In contrast to spectrofluorometers, filter fluorometers employ glass and gelatine colour filters and light sources (collectively referred to as a "filter set") which are carefully selected to optimise fluorescent sensitivity, although the spectral width of both absorption and emission wavelengths are much wider and less discriminating. The problem of cross fluorescence is thus more likely to occur, and will be <u>far less evident to the casual analyst</u>. For this reason, it will <u>always</u> be worthwhile obtaining cross-fluorescence calibration coefficients if more than one fluorescent tracer is likely to be encountered in a tracer test.

Measurement of fluorescence using a filter fluorometer is shown in Figure 3 which may be contrasted to Figure 1. The filter cut-off is shown to be unrealistically sharp, this simplification in no way invalidates the analysis which follows. A filter fluorometer integrates across a broad spectral band to obtain fluorescence. It can be seen that Dye 2 now has a small overlap with spectral band A, in contrast to the complete exclusion possible with a spectrofluorometer.

Despite the increase in cross-fluorescence, the calibration and analytical procedure is identical to that developed above. First, calibration standards (of arbitrary concentration) are run through both filter sets to get a matrix of fluorescence readings $\{F^x_y\}$:

Calibration Measurements $\{F^x_y\}$		Dye	
		1	2
Filter Set	A	F^A_1=241.	F^A_2=1.1
	B	F^B_1=45.9	F^B_2=260

The calibration measurements are divided by the column matched pair (F^A_1 and F^B_2) to generate the correction factor matrix $\{k^x_y\}$.

Cross Fluorescence Calibration Coefficients $\{k^x_y\}$		Dye(y)	
		1	2
Filter Set (x)	A	k^A_1=1.00	k^A_2=0.004
	B	k^B_1=0.191	k^B_2=1.00

These factors are then employed on sample analyses to correct for any cross-fluorescence. Figure 4 provides an example where F^A and F^B are 313 and 196 respectively. Substituting F^A and F^B for h^A and h^B in equations 4 and 5 above gives estimated sample F^A_1 and F^B_2 of 312 and 136 respectively, with -0.2% and -30% change from the measured fluorescence. Fluorescence measurements may be converted to concentrations in the usual manner.

Colour filters, light sources and electronics are specific to each filter fluorometer. The same idiosyncracy applies to dye stocks, even from a single batch if poorly homogenised. It is not possible or desirable to transfer calibration data between machines, components, dyes or over time. A final caution concerns the high temperature coefficient of rhodamine dyes (Smart and Laidlaw 1977). Where sample temperatures differ from the temperature of the standard used in calibration, temperature measurements will be necessary on both A and B analyses, and the

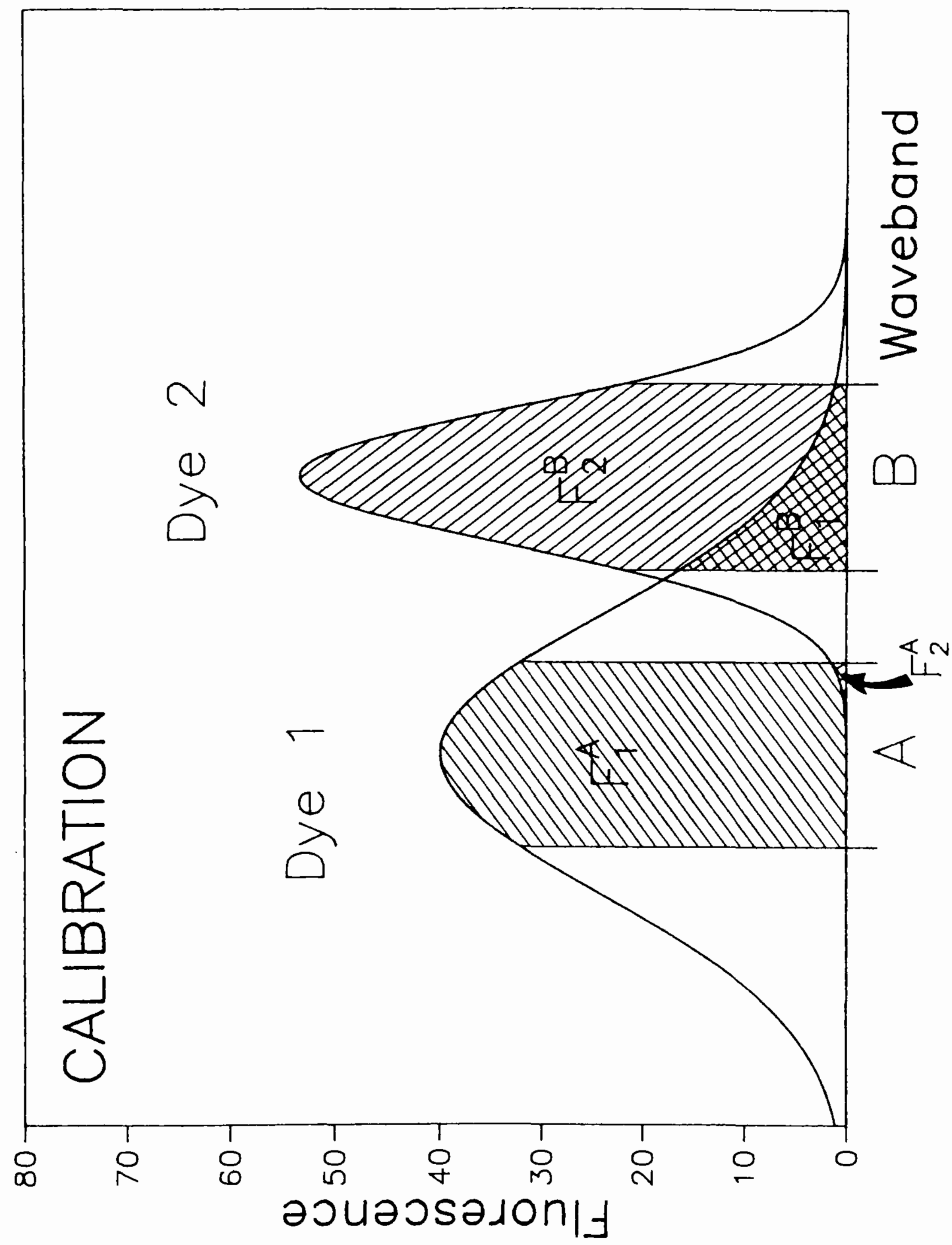

Fig. 3. Schematic diagram of two component cross-fluorescence in a filter-fluorometric analysis. The lack of specificity makes for stronger cross-fluorescence. The areas F_y^x indicate fluorescent readings on dyes 1 and 2 in filter sets A and B. See text for details of the calibration.

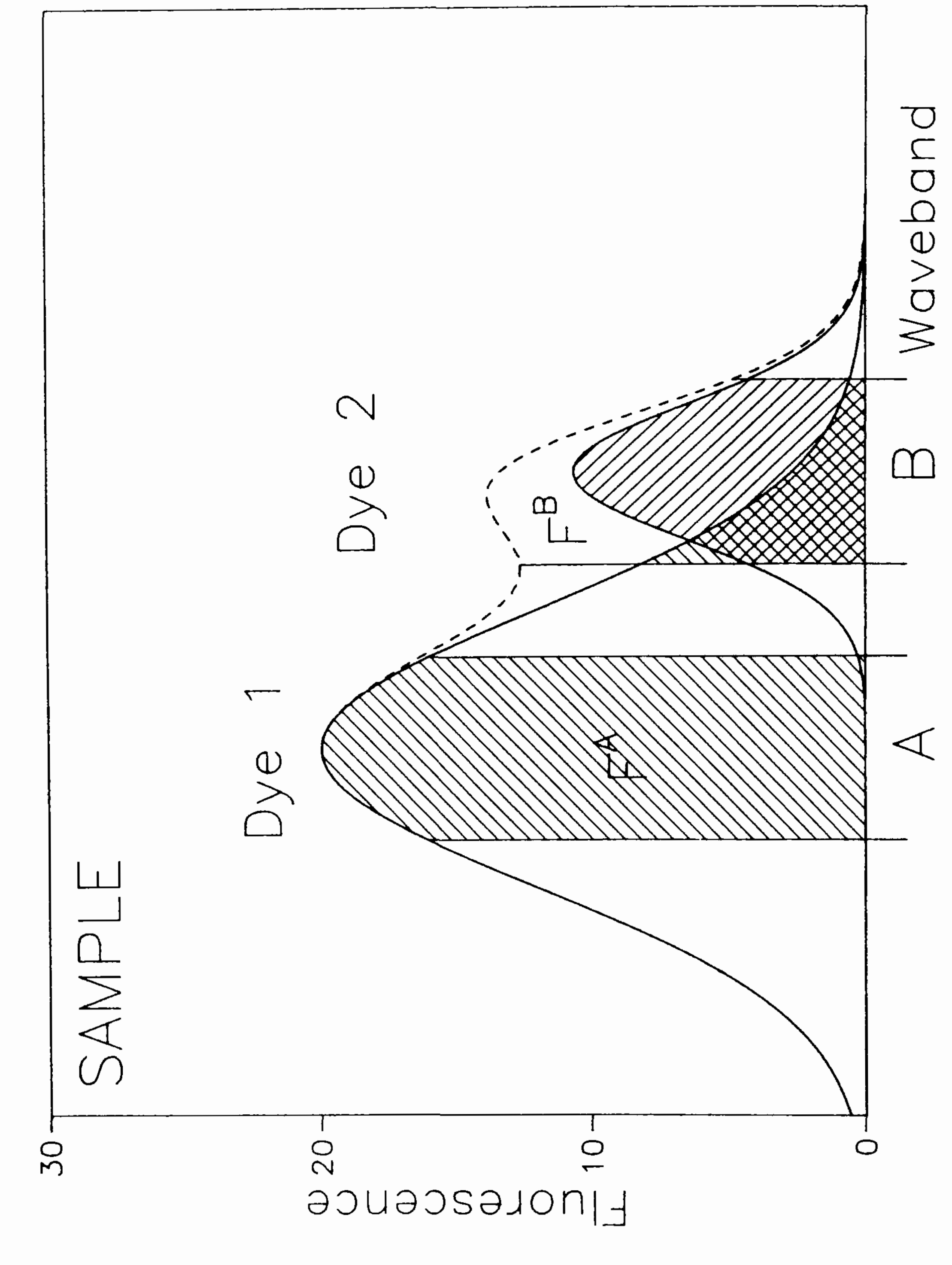

Fig. 4. Schematic diagram of the fluorometric analysis of a water sample containing dyes 1 and 2. Measurement only reveals F^A and F^B. The underlying cross-fluorescence requires correction to obtain true fluorescent readings.

correction factor applied to the estimated fluorescence of the rhodamine dye.

THEORY OF CROSS FLUORESCENT CORRECTION

The methods above have a theoretical basis which can be expanded to deal with a more general case of three dyes (1,2,3) and filter sets (A,B,C).

Calibration of three dyes through three filter sets will give a 3X3 matrix of fluorescence readings, which divided by the matched dye-filter pair for the column will yield the following general matrix of correction factors $\{k_y^x\}$:

Table of Cross-Fluorescence Calibration Coefficients $\{k_y^x\}$		Dye (y) 1	2	3
Filter Set (x)	A	k_1^A	k_2^A	k_3^A
	B	k_1^B	k_2^B	k_3^B
	C	k_1^C	k_2^C	k_3^C

$k_1^A=k_2^B=k_3^C=1$

If no cross-fluorescence $k_y^x=0$

The total (i.e. measurable) fluorescence of a mixture of three cross-fluorescent dyes in a particular filter set is the sum of the matched dye fluorescence, plus the other dye fluorescences in their matched filter set, corrected with the appropriate cross-fluorescence calibration factor. Thus,

$$F^A = F_1^A + k_2^A F_2^B + k_3^A F_3^C \tag{3}$$

$$F^B = F_2^B + k_1^B F_1^A + k_3^B F_3^C \tag{4}$$

$$F^C = F_3^C + k_1^C F_1^A + k_2^C F_2^B \tag{5}$$

The quantity of interest is the positive diagonal F_y^x, the actual fluorescence due to a particular dye in its matched filter set. Re-expressing 3-5 gives

$$F_1^A = F^A - k_2^A F_2^B - k_3^A F_3^C \tag{6}$$

$$F_2^B = F^B - k_1^B F_1^A - k_3^B F_3^C \tag{7}$$

$$F_3^C = F^C - k_1^C F_1^A - k_2^C F_2^B \tag{8}$$

It is necessary to remove unknown terms from these expressions by substitution. Taking (6) as an example, substituting (7) for F_2^B and (8) for F_3^C, collecting terms and then resubstituting gives the following general expression

$$F_1^A = \frac{F^A - F^B(k_2^A - k_3^A k_2^C) - F^C(k_3^A - k_2^A k_3^B)}{1 - k_2^A k_1^B - k_3^A k_1^C} \cdots$$

$$\cdots \frac{- k_2^A k_3^B k_2^C F_2^B - k_3^A k_2^C k_3^B F_3^C}{+ k_2^A k_3^B k_1^C + k_3^A k_2^C k_1^B} \tag{9}$$

The product of three off-diagonal k_y^x will be negligible. Ignoring such terms (collected in the lower line of (9)) not only simplifies the expression, but leaves the required fluorescence expressed in measurable and known terms.

$$F_1^A = \frac{F^A - F^B(k_2^A - k_3^A k_2^C) - F^C(k_3^A - k_2^A k_3^B)}{1 - k_2^A k_1^B - k_3^A k_1^C} \tag{10a}$$

The complementary equations to solve for the true fluorescence of the other two dyes are as follows:

$$F_2^B = \frac{F^B - F^A(k_1^B - k_3^B k_1^C) - F^C(k_3^B - k_1^B k_3^A)}{1 - k_1^B k_2^A - k_3^B k_2^C} \tag{10b}$$

$$F_3^C = \frac{F^C - F^A(k_1^C - k_2^C k_1^B) - F^B(k_2^C - k_1^C k_2^A)}{1 - k_1^C k_3^A - k_2^C k_3^B} \tag{10c}$$

In two component mixtures (e.g. dyes 1 and 2) all terms with superscript C or subscript 3 will equal zero, giving
In many cases the product of two k_y^x will be negligible, thus allowing a very simple cross-fluorescence correction.

$$F_1^A = \frac{F^A - k_2^A F^B}{1 - k_2^A k_1^B} \tag{11}$$

It is unlikely that a full three-component cross-fluorescence correction will be necessary where widely different dyes and filter sets are employed (e.g. the Red, Green, Blue suite outlined by Smart and Laidlaw, 1977). It may be possible to separate dyes previously rejected because of excessive cross-fluorescence, although no experiments have been tried as yet. The most common application will be for Rhodamine WT interference with fluorescein sodium (i.e. Uranine, CI 45350, Acid Yellow 73). Very high rhodamine concentrations may mask a weak fluorescein peak. Alexander (1991, pers comm.) reports just such an occurrence which was only detected following correction for cross-fluorescence.

BACKGROUND CORRECTION

The single greatest problem in contemporary fluorometric tracing is that of natural background fluorescence. Although seldom a serious problem with "red" dyes, it is a common difficulty with green dyes, and especially acute with blue dyes. This consistent wavelength dependent sensitivity to background interference indicates that background has a characteristic spectral signature (e.g. Smart et al. 1976). Smart and Laidlaw (1977) and Smart and Friederich (1982) have employed this property to establish the ratio of background fluorescence in dye-free water. Significant departure from this line is an indication of a positive sample. A critical displacement of two standard deviations from the line was suggested, although a statistical confidence band might provide a more objective criterion. The technique will fail if the tracer dyes are present in a similar ratio to the background fluorescences.

The viability of the ratio technique suggests that it is possible to correct for background interference with red and green dyes by treating the background as a third dye component. A third filter set, sensitive to background (an ultra-violet:blue combination will be most suitable) constitutes the C waveband (Figure 5). A dye-free background sample is essential to calibrate the method, but the concentration for all cross-fluorescence calibrations is arbitrary, so composition is less important than stability and representativity. Fluorescence readings are taken using the three standards in all three filter sets to give a 3x3 matrix $\{F_y^x\}$.

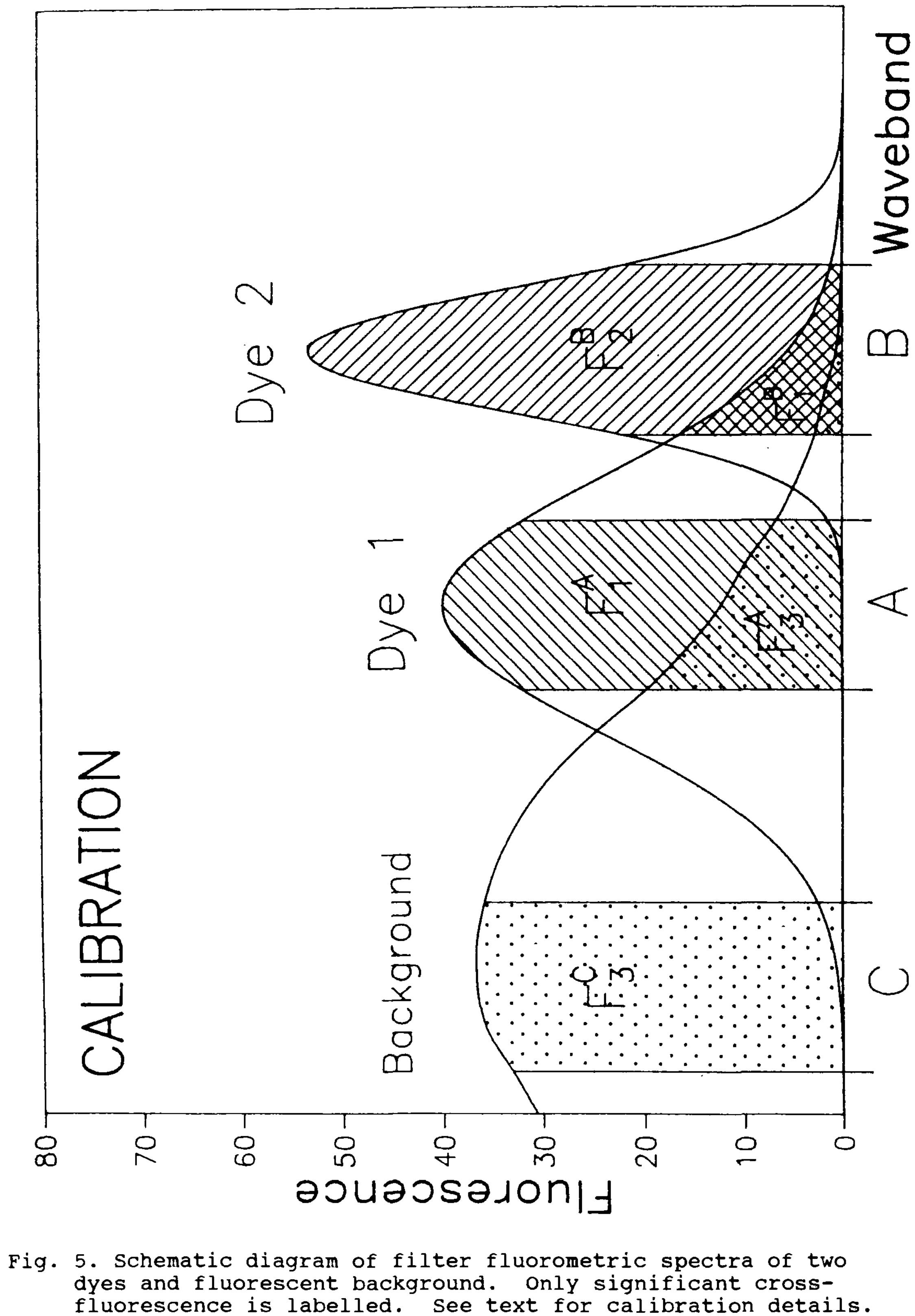

Fig. 5. Schematic diagram of filter fluorometric spectra of two dyes and fluorescent background. Only significant cross-fluorescence is labelled. See text for calibration details.

An example based on idealised data can be drawn from figure 5 to give

Table of Calibration Measurements $\{F_y^x\}$		Dye (y)		
			2	3
Filter Set (x)	A	243	1.50	85.2
	B	44.1	269	6.53
	C	4.75	0.00	238

Division of each row by its positive diagonal element will give the $\{k_y^x\}$ matrix.

Table of Cross-Fluorescence Calibration Coefficients $\{k_y^x\}$		Dye (y)		
		1	2	3
Filter Set (x)	A	1	0.0056	0.358
	B	0.182	1	0.0274
	C	0.0195	0.0	1

Taking an imaginary example (Figure 6) of F^A=292, F^B=117 and F^C=340 and applying these values with $\{k_y^x\}$ into equations 10a, 10b, and 10c gives F_1^A=169, F_2^B=76.9, and F_3^C=337, corrections of −42, −34 and −0.9 percent respectively. It is clear that the magnitude of the background problem can be severe and probably accounts for many cases of excess dye recovery estimations.

The cross-fluorescence method of background correction is the best developed to date, although it requires considerable analytical commitment to provide all the necessary data. The method is currently only theoretical and will require considerable testing and verification before its reliability can be assured. The simplest empirical indication of an error in application is the estimation of negative fluorescence. The key assumption is that the spectral signature of the background is steady and changes only in amplitude over time. The influence of spectral shift is unknown, but may not be too significant if a sufficiently broad wave band is employed in characterising the background signature. There is very little information available on the pattern of background variation between springs and with time at a single site. Quinlan (1991, pers. comm.) suggests that variations between springs are greater than variations at a single spring. This means it would be necessary to establish a $\{k_y^x\}$ matrix for each spring if local stability were assured.

Tracer samples eluted from activated charcoal detectors often exhibit the most serious background problem. Unfortunately, the concentration and exchange of organic molecules on tracer

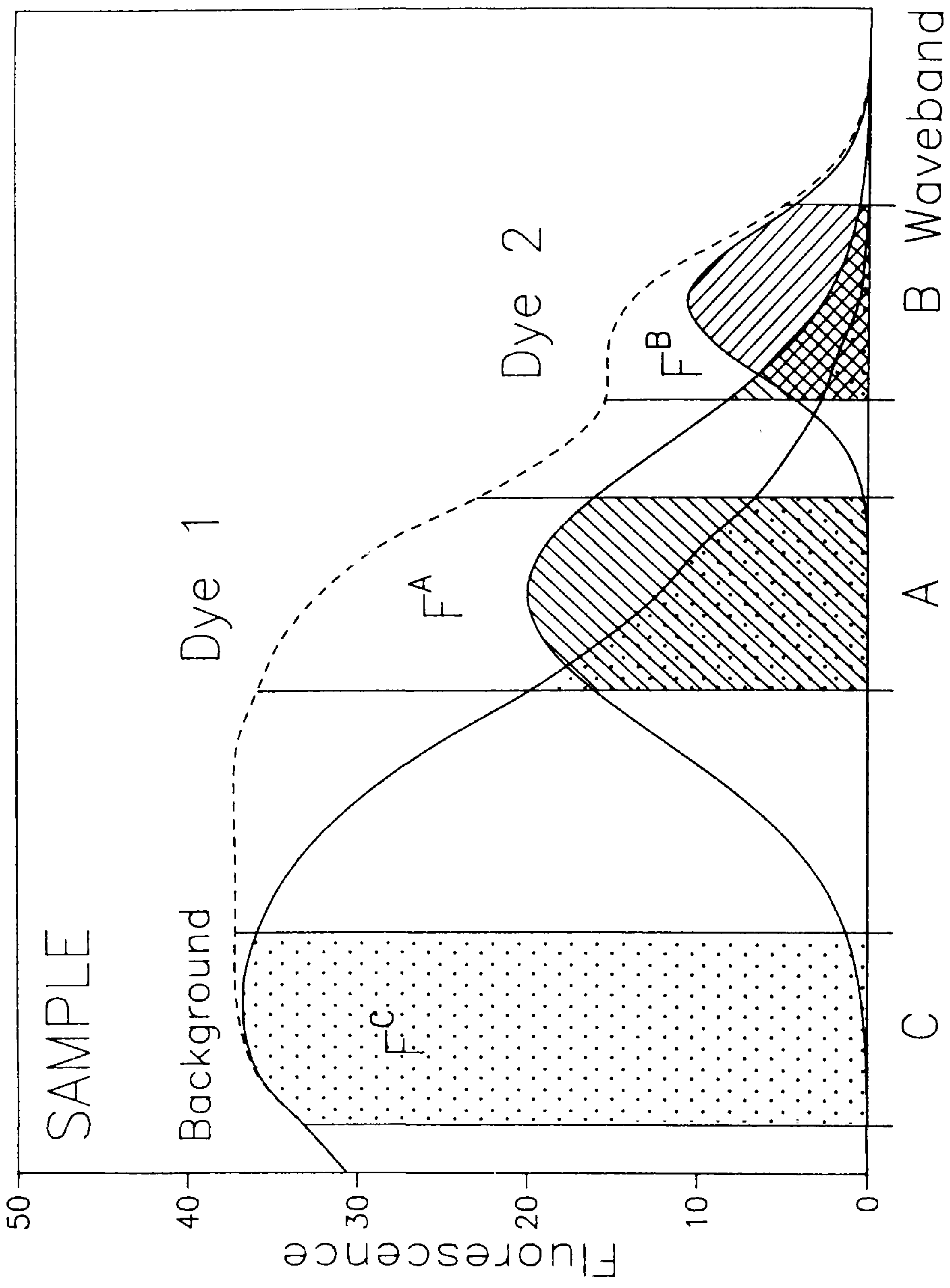

Fig. 6. Schematic diagram of the fluorometric analysis of a water sample containing two dye and a background component. Only the F^x components can be determined, but it is possible to use the cross-fluorescent model to correct for such severe background interference.

samples eluted from charcoal detectors creates dramatic variations in background composition and resulting signature (Quinlan pers comm.), so that the cross-fluorescence method will not be applicable to this particular problem.

Experiments are currently under way to test and evaluate the cross-fluorescence methodology. The results and conclusions will be reported following this completion of this work.

CONCLUSIONS

Cross-fluorescence is a potentially serious problem in the analysis of fluorescent tracers, especially with filter fluorometers. Cross-calibration should always be undertaken during analysis to determine the extent of the problem and to allow post-analytical correction if necessary.

The three-component general theory of cross-fluorescence as presented can be extended to more complex mixes with considerable spectral overlap, providing care is taken to validate the approach. The method promises to provide a useful operational correction for background interference, but substantial field experiments will be necessary to assess the reliability of the method.

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Session VI:

Ecology of Caves and Karst Terranes

ASSESSING GROUNDWATER QUALITY IN CAVES USING INDICES OF BIOLOGICAL INTEGRITY

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ABSTRACT

The states of populations and communities give an integrated record of past and ongoing pollution whereas toxicity bioassays and chemical analyses may detect pollution occurring only at the moment of sampling. Of the population and community biocriteria that are derived from field data, indicator species, diversity indices, and community similarity metrics are either totally flawed or have more disadvantages than advantages. The preferred biocriterion is an Index of Biological Integrity (IBI) which combines habitat data with population and community data. In caves there are few species and so most must be considered. For fish, salamanders, and crayfish all individuals are located on a map, measured, and examined for reproductive condition and lesions. The smaller species and habitat characteristics are sampled at random stations. Since surface waters enter caves by different routes, I sample communities from diffuse input formation areas, from mixed input areas like terminal breakdowns, and from direct inputs associated with sinking streams, vertical shafts, and backflooding from surface rivers. Through 30 years of study, mainly in the Mammoth Cave Region, I have documented or inferred the non-lethal effects on populations and communities due to homogenizing of habitat structure by siltation, due to favoring of facultative over obligate cave species by excess food input, and due to compromising reproduction and growth of very long-lived species by biomagnification of toxins along food chains.

BIOLOGICAL MONITORING IS CHEAPER THAN CHEMICAL MONITORING AND GIVES A MORE INTEGRATED RECORD OF PAST AND ONGOING POLLUTION

Chemical toxicity tests have now been flawed as a single approach because they have been done on a minority of suspected toxicants, can not assess effects of bioaccumulation or resulting indirect ecological effects, use only a few test species, and toxicity of one chemical changes as the overall chemistry of water changes. Increasing use of whole effluent ("out of the pipe") toxicity tests has only partly mitigated these problems. Neither chemical-specific criteria nor whole effluent tests assess impacts of habitat modification, such as water level, or non-toxic pollutants, such as of sediments and nutrients. Furthermore both depend on sampling at the moment of pollution input, unlike the case for field biological criteria.

The composition of ecological communities integrates ongoing and past pollution over space and time whereas toxicity tests usually only detect pollution occurring at the instant of water sampling. Yearly biological surveys often suffice to demonstrate ongoing sublethal pollution or past pulses of lethal pollution (as with "midnight dumping") whereas these impacts are usually missed by monthly or even weekly sampling of water for chemical and toxicity tests. Thus biological surveys should be used as an inexpensive early warning system to detect the presence of a problem. Expensive chemical analysis and toxicity testing should be reserved to pinpoint the exact nature of a problem and to develop discharge limits for particular pollutants. This combination of biological surveys and chemical tests provides a complementary balance in the protective and regulatory process.

In karst regions monitoring water quality is more difficult than in surface waters because of problems of inaccessibility and complications of diffuse flow. Quinlan and Alexander (1987) and Quinlan (1990) show that nearly continuous sampling is needed to detect organic pollutant pulses (e.g. the herbicide atrazine) coincident with hydrograph peaks during conduit flow in streams. It may be that headwater point inputs of organic pollutants via sinking streams and open sinkholes on the Sinkhole Plain will be diluted before they reach Green River Springs but diffuse input of pollutants can linger and cause insidious bioaccumulation in cave organisms (Barr 1976). Diffuse input of pollutants through the soil and epikarst into wells and eventually into streams is slow and complex as Quinlan and Alexander (1987) demonstrate for an in organic fertilizer (nitrate).

THE NEED FOR A COMMUNITY AND HABITAT BASED INDEX OF BIOLOGICAL INTEGRITY

Of the biocriteria available toxicity bioassays, indicator species, diversity indices, and community similarity metrics are either totally flawed or have many more disadvantages than advantages. The preferred biocriterion is an index of biological integrity (IBI) which combines habitat data with population and community data (Karr 1989).

Single Species Toxicity Bioassays

EPA's acute and chronic bioassays on *Ceriodaphnia* and fathead minnows have the advantage that they have been standardized and are rigorously analyzed statistically but they are fatally flawed as single best biocriteria for assessing impacts of pollution (Cairns 1986, Karr et al. 1986). Cairns points out that the central flaw in toxicity testing is the "myth of the most sensitive species". One corollary of this myth is the unwarranted assumption that the responses of the test species correspond to those of the many species not tested. Another flawed assumption is that there are no responses that are more sensitive than the 50% lethal dose usually measured. For example even the standard test organism may have sublethal responses more sensitive than LC50, such as depression of respiratory rate. Such sublethal responses in nature could change behavior of predators or competitors and so have reverberating efffects throughout the community. Still another flawed assumption is that the cost savings using single species tests more than compensates for the cost of failing to act when the single species test does not predict negative effects in nature. In addition, Karr points out (1988) that even in situ field toxicity tests are not sensitive to environmental degradation due to non-toxic pollutants like sediments and nutrients. In situ toxicity tests *are* more reliable at detecting intermittent pollution than laboratory tests using water collected at one instant in time but the costs of in situ tests and their practical difficulties are great. The most important practical difficulty is inacessibility of the caged test organisms during flood peaks when we expect the greatest concentrations of many organic pollutants (e.g. Quinlan 1990).

I believe that there is little hope of developing a "white mouse" toxicity test using a cave species because of the complexity of culturing cave organisms and the conflicting toxicological results using field collected cavernicoles. Reliable culturing of a troglophilic amphipod has been difficult (Dan Fong personal communication) and nobody has successfully gotten an aquatic trogobitic species through even one generation in the laboratory. The one toxicity test using EPA protocols on field-collected animals (Bosnak and Morgan 1981) had the counterintuitive result that the cadmium LC50 for a troglobitic isopod was higher than for a troglophilic isopod. My explanation for this result is that the troglobite would be less sensitive to acute exposure to a toxin if it had a lower metabolic rate than the troglophile, as is consistent with the troglobite having a 16 fold lower body concentration of cadmium with a 15 fold higher LC50 than for the troglophile. Barr's acute toxicity studies at Mammoth Cave (1976) are not very sensitive because he used only 1000 ppM of several toxins and his results are equivocal. Given the complexities in his own data and the conservation problems of using so many individuals for tests, Barr recommended to NPS that no more toxicity tests be done.

Problems of Using a Single Indicator Species in the Field

Use of a single indicator species or even a single genus as a field indicator of adverse impacts has fewer advantages and some of the same disadvantages as just discussed for single species toxicity testing. Presence or absence of an indicator is a simple criterion but there are many problems of interpretation. In addition to the same problems as with the "myth of a most sensitive species" there is the problem that a species may be absent due either to pollution or for natural reasons. One natural constraint is inability to live in some habitats. For example shrimp do not live at upper levels or even at the headwaters of shaft master drains (Leitheuser et al. 1982) for reasons that are not clear. Another natural constraint is of predation or predation mediated competition. For example the isopod *Caecidotea stygia* does not occur in lower level habitats possibly because of predation by crayfish and/or fish but this does not appear to be a problem for *Caecidotea bicrenata* (Lewis and Lewis 1980). In other situations food and predation may interact to affect the distribution and abundance of a species. An example is of isopods in three unimpacted master shaft drains under Flint Ridge (Poulson 1968). Isopods are most abundant where there is a combination of gravel and rock cover, that mitigates the impact of fish and crayfish predation, and a lot of fine particulate organic matter, that serves as food.

Interpretation of Species Diversity is Difficult Because Not All Species are Equal

The number of species present, species richness, is easy to understand and the the diversity of species, H, is easy to calculate but both have critical problems with measurement and/or interpretation.

As a single biocriterion species richness is especially sensitive to the recurring problem of standardizing sampling effort and efficiency. Just how long should one sample over how large aan area to be sure that one has found all the species present?

The information theory index of diversity H (see Poulson and Culver 1969 for a detailed description) is not so sensitive to sampling effort but its interpretation is made difficult because it confounds the separate effects of numbers of species and their relative abundances. Because of this problem one should report both numbers of species and their relative abundances and not use the single index H.

Quantitatively both species richness and H are flawed by the implicit assumption that all species are equal in their effects in a community. In fact the potential impact of a species is measured by its absolute importance value (Poulson and Kane 1981). Importance Value of a species is the composite of its frequency, density, and impact per individual. A very frequent species, found in all habitats and at all times, is more likely to have an impact than a rare species. A species which is locally very dense could have a high local impact even if it is not frequent in space and time. And, a species which is large relative to others in its community could have a great impact even if it is infrequent and has a low density.

Qualitatively both species richness and H are flawed by the implicit assumption that communities with natural and exotic species are equally indicative of biological integrity. In the case of caves this is equivalent to saying that two communities are the same if one has many accidentals and troglophiles and the other has only troglobites, as long as both communities have the same number of species and the same H. An example shows that this is patent nonsense. Clearly the pollution stressed Hidden River Cave community dominated by tubificid sewage worms and several large protozoan species with high numbers of troglophilic crayfish and rare unspecialized troglobitic amphipods (*Crangonyx*) is very different than the unimpacted Eyeless Fish Trail under Flint Ridge with a regular troglobitic cave fish and crayfish, one or two common copepods, uncommon Isopods, and rare but specialized troglobitic amphipods (*Stygobromus*).

Measuring Statistical Similarity of Community Species Composition Has Advantages but it is Not A Good Single Biocriterion

The simplest measure of community similarity uses only presence-absence data and and is calculated by dividing the species in common by the sum of species in the two communities to get a percent similarity. This measure easily detects the differences between a badly impacted community such as in Hidden River Cave and an unimpacted one such as in Eyeless Fish Trail. However it does not give an early warning sign if there are slight changes in community composition and or changes in abundance of species without changes in species composition.

Multivariate statistical techniques, such as Principal Components Analysis, use both species identity and abundance data to group species along two or more axes. The axes often do not correspond to any simple environmental variable and so the species clusters can be difficult to interpret ecologically. None-the-less statistical changes in species position in a cluster or in location of a cluster, compared to the natural variability seen from time to time in baseline communities, could signal subtle changes and provide a good early warning sign before negative impacts proceed too far.

An Index of Biological Integrity Keeps the Advantages and Avoids the Disadvantages of Other Biocriteria and An IBI Has Unique Advantages

A maximum value for an IBI indicates that a sampled area has the "capability of supporting and maintaining a balanced, integrated, adaptive community having a species composition and functional organization comparable to that of the natural (unimpaired, original) habitat in the area" (Karr et al. 1986). The EPA now mandates use of both a fish and a benthic macroinvertebrate IBI in all surface stream surveys. An IBI includes many different kinds of criteria that are ecologically meaningful and so it is sensitive to a wide array of differing insults to the natural environment. An IBI does a particularly good job of integrating, over space and time, the effects of past and ongoing negative impacts. An IBIs detailed interpretation depends on the professional judgement of an ecologist because the use of a variety of qualitative and quantitative measures does not allow for statistical treatment. However the methodology for measuring each metric used is clearly enough explained that anyone can be trained to use the IBI.

The multiplicity of criteria used to calculate an IBI include major categories of habitat heterogeneity, energy source, numbers of species and guilds, and well-being of each species. Available IBIs and my proposed cave IBI give positive points to sensitive natural species like cave fish and give negative points to troglophiles, exotic species, and pollution tolerant species not normally found in the community. The two scales of habitat heterogeneity measured are also applicable to caves. At a large scale one measures characteristics of the stream itself such as size metrics, meander length, presence of undercut banks, and the proportions of pool, raceway, and riffle habitats. At a finer scale one measures the sediment size distribution on the bottom from bedrock and cobble, through gravels and sand, to silt and clay. In a cave there is no plant photosynthesis and so the relevant measure of energy input is the amount of coarse and fine particulate organic matter and the amount of non-toxic dissolved organic matter. Most of the energy will be on or in the substrate and so I have used weight loss on ignition at 550C as an index of total organic matter in the sediments (Poulson 1968). Simple indices of well-being can be applied to most cave species. A species is doing well if it is abundant and has a size distribution that includes the small individuals indicative of recent reproductive success. In fish a rounded belly and subcutaneous fat deposits are good signs and in translucent crayfish the presence of yolked eggs in the ovary is a good sign.

A CAVE IBI WILL INCLUDE MOST KINDS OF AQUATIC SPECIES AND SHORELINE TERRESTRIAL SPECIES

Cave ecosystems have few species and so currently EPA approved IBIs for fish (Karr et al. 1986) and for benthic macroinvertebrates (Ohio EPA 1987) cannot be used or even simply modified for use in caves. The major reason for low species diversity in caves is that lack of photosynthesis means there is a small energy base for food webs. The problems of low food supply mean that most cave species must be generalists in both feeding and habitat use and this in turn makes it difficult for new species to invade unimpacted cave communities. For example cave crayfish feed as detritivores, scavengers, and predators and cave isopods occur in fast and quiet water and on all kinds of substrates.

All Kinds of Species Will Be Sampled

Because there are few species, a cave IBI must include all kinds of easily sampled organisms. Field census without collection is easily done for fish, crayfish, shrimp, and flatworms and can be done for isopods and amphipods if species identification is not critical (Poulson 1968). Censuses of the larger shoreline terrestrial species is also easy in the field. Baiting and trapping techniques will be used where organisms are rare, where much of the habitat is inaccessible, and where turbidity and high water preclude visual observation (Cooper and Poulson 1978, Cooper 1975, Poulson and Culver 1969). A high intensity diving light will greatly increase efficiency of census any time that waters are slightly turbid and/or deep.

Detailed Data Taken On All Individuals Of Large Species

For extremely long-lived species with slow growth and infrequent reproduction (fish Poulson 1963, crayfish Cooper 1975, and shrimp Leitheuser et al. 1982) the entire accessible stream will be searched and all individuals will be measured. Size distributions are a good index of current and past reproductive success, as explained earlier. To check for the presence of fin erosion, sores or tumors, and ectoparasites the fish must be examined in a translucent lucite box for a few minutes. Even more detailed data will be taken on crayfish because their hard exoskeleton protects them during handling. It is easy to sex them and assess male reproductive condition (form I vs form II). In addition the translucent exoskeleton of the trogobitic species allows detailed assessment of female reproductive status even by observing them in the stream (Jegla and Poulson 1970). Unlike crayfish, shrimp are too delicate to handle but they are transparent and so status of ovarian or hatched eggs is observable in the stream with appropriate snorkel or SCUBA gear (Leitheuser et al. 1982).

Stratified Random Sampling To Census Smaller Organisms and Habitat Characteristics

Smaller organisms have shorter generation times than fish, salamanders, or crayfish and so are better indicators of short term changes in water quality. Their more cryptic habits and higher densities require that random station be sampled [Permanent stations are useful for short term studies of life cycles (Lewis 1982) but can be misleading when there are shifts of organism microdistribution as locations of substrates or food shifts after periods of high stream flow.] At each random station aquatic macroinvertebrates are censused stream habitat characteristics measured, abundance of surface particulate organic matter scored, large shoreline terrestrial species censused, and tiny terrestrial species trapped. Both large aquatic and terrestrial species will be censused using a combination of timed and areal searches to allow comparison between areas with abundant and sparse populations. All microhabitats will be included in the censuses. If there is a stream current then mini-surber samples will be taken using a small silk aquarium net (in my experience this picks up copepods as well as some large protozoa and some bacteria). Habitat characteristics will be assessed just as discussed earlier for Karr's fish IBI but with special attention to refuges from floods such as undercut banks, breakdown, and rocks and gravel in the streambed. Cores of the stream and/or shoreline sediments may be taken for weight loss on ignition and to assess the presence of lenses of particulate organic matter that could be exposed by flooding. Often there is no input of new leached organic matter for years and so redistribution of the low quality old particulate matter may be important.

BASELINE IBIS MUST BE DEVELOPED AT ECOREGION, SUBREGION, DRAINAGE BASIN, AND LOCAL WATERSHED SCALES

For karst, contrasting ecoregions are the Valley-Ridge of the central Appalachians vs the Interior Low Plateau in the Mammoth Cave area. Cave streams have different characters in these two ecoregions and so there are different baseline faunas (e.g. Culver 1982). Within the Mammoth Cave ecoregion southeast of Green River there are two subregions, the Sinkhole Plain and the Mammoth Cave Plateau. These subregions differ greatly in geology, hydrology, aand cave development (as reviewed by many authors in White and White 1989). Each of these areas has its characteristic communities (Barr and Kuehne 1971, Lewis and Lewis 1980, Lewis 1981, Poulson 1968, Poulson and Culver 1969). In addition there is a community that includes shrimp. It appears to be restricted to the interface of downstream flows and backflow, during times of flood from the Green River, through alluviated springs (Lisowski and Poulson 1979, Duchon and Lisowski 1981, Leitheuser et al. 1982). The still different Hilly Country to the northwest of Green River is now being studied hydrologically but has scarcely begun to be studied biologically.

DIFFERENT COMMUNITY RESPONSES TO NATURAL FOOD INPUTS AND DIFFERENT CATEGORIES OF POLLUTANTS: A SERIES OF NATURAL EXPERIMENTS

In this final section I review what baseline data we have so far to use in developing IBIs and discuss what natural experiments can tell us about community change with categories of pollutants that include sediments, human and animal wastes, pesticides, heavy metals, and a potpourri associated with oil and gas exploration. More detail is provided in the unpublished Proceedings of the 1990 Mammoth Cave National Park Karst Conference.

General kinds of responses of cave communities to different categories of disturbance have been considered elsewhere. Thus Poulson and Kane (1977) discuss the general impacts of removing single species, simplifying habitat structure by siltation, modification of the timing and extent of floods, favoring facultative cave species (troglophiles) by excess nutrient input, and stress on long-lived obligate cave species (troglobites) associated with toxins that are biomagnified along food chains.

Natural Variation Among Vertical Shaft Master Drain Streams Under Flint Ridge

Vertical shaft drains and master drains under the Mammoth Cave Plateau are almost certainly in a near natural state and base level streams under Flint and Mammoth Ridges are probably also natural since some receive little input from the Sinkhole Plain. Thus data on these streams (especially Poulson 1968) are a solid start for developing IBIs to use in assessing impacts on more impacted baselevel streams under Joppa Ridge.

The diversity of fauna among the streams is correlated with diversity of habitat and kinds of food supply. The stream with the highest diversity of aquatic species and the least census to census variation is the longest stream with both the greatest diversity of the food base and the greatest diversity of habitats. Thus the right habitat combination for each species is to be found somewhere at all times. In contrast the stream that is smallest in extent has a low diversity of habitats. Even though it has by far the greatest particulate organic matter it also has the greatest variation in aquatic organism density as the stream varies in extent among years. On the other hand the variation in the terrestrial fauna is the least of the four streams. The veneer of coarse particulate organic matter on the ceiling provides a virtually unvarying food supply that supports by far the richest terrestrial fauna of the four streams and the highest densities of troglobitic millipede (*Scoterpes copei*) and harvestman (*Phalangodes armata*) that I have censused anywhere in the Mammoth Cave Area.

Tradeoff of High Food Supply and Low Predation Pressure with Low Habitat Stability in Upper Level Seeps, Drip Pools, and Shaft Streamlets

In the caves under the Mammoth Cave Plateau the upper level aquatic habitats do not have a diverse fauna but an IBI should be developed for them because they are likely to be the first areas to be impacted by any local pollution on the plateau and their waters often reach the shaft stream complexes at lower levels. In general, whether impacted or not, these upper level habitats are dominated by amphipods and flatworms that have adaptations for surviving in the mud when pools or seeps dry up annually and can take advantage of high seasonal inputs of bacteria and fine particulaate organic matter. When there is permanent water isopods are added (*Caecidotea stygia*, Lewis 1982) and a troglophilic amphipod may occur if there is unnaturally high organic input (*Crangonyx packardi*, Lewis 1987).

Cave Base-Level Community Compromised By Historic Alterations Of Green River Discharge Patterns And Depth

The combined effects of three dams have compromised the base-level communities, of the Styx-Echo River areas of Mammoth Cave, through a combination of increased siltation and decreased food availability. This conclusion is based on historical records pre and post installation of Lock and Dam #6 downstream on Green River and on biosurveys of fish, crayfish, and shrimp during development of negative impacts of the upstream dam in the 60s and 70s (Barr and Kuehne 1971, Lisowski and Poulson 1979, Duchon and Lisowski 1981) and continuing to present. A main focus in the 70s was the apparent decline of the Kentucky cave shrimp, *Palaemonias ganteri*, in the Styx-Echo River area, a pattern that has now been borne out by detailed study of the shrimp's distribution and abundance (Leitheuser et al. 1982).

Short-lived Species May Be More Sensitive To Acute Exposure To Toxins And Long-Lived Species More Sensitive To Chronic Exposure

Cave communities will respond differently to acute and chronic exposure to heavy metal or organic toxins depending on whether the exposure is acute and/or chronic. Single or repeated acute exposures are likely to kill small, short-lived species with relatively high metabolic rate. This will decrease or eliminate the food supply for the larger, longer-lived and lower metabolic rate species of upper trophic levels. The early warning signs of this scenario are easy to detect from biosurveys and IBIs. More subtle and insidious are the effects of sublethal chronic exposure to toxins because the early warning signs are first seen in a further decrease in the normally infrequent reproductive success of fish and with no elimination of species. With chronic toxin pollution the cave fish will have increasing body loads of toxins due both to bioaccumulation associated with long life and to biomagnification along the food chain. If we are lucky there will also be obvious signs of problems such as with the hemorrhaging and impaired swimming of "broken-back syndrome" in cave fish (Keith and Poulson 1979). Cave crayfish are less likely than cave fish to be affected by chronic organic toxins because they store less fat and less likely to be killed by severely acute exposures because they can leave the stream and so decrease exposure as was observed to occur in summer 1979 with an acute hydrocarbon exposure in Hawkins River.

If there are changes in IBI or individual species well-being due to suspected toxin exposure then identification of the culprit(s) will depend on chemical testing since all toxins can kill both by acute exposure and by chronic bioccumulation and biomagnification during prolonged sublethal exposures.

Inorganic Fertilizers Affect Cave Communities Only If They Stimulate Primary Productivity Of Surface Waters Which Are Subsequently Diverted Into Caves

There is no photosynthesis in caves so the stimulating effect of inorganic fertilizers, especially phosphates and nitrates, on aquatic plant growth can only affect cave communities indirectly. Impacts will occur when algal choked sinkhole ponds overflow into caves or when the sinkhole pond disappears underground as its drain suddenly opens. A natural experiment that simulated this scenario was of a catastrophic leakage of the Job Corp sewage treatment lagoons into vertical shafts that feed Eyeless Fish Trail under Flint Ridge in 1967. This was a one time event and did not cause a detectable change in the distribution, abundance, or well-being of cave fish or crayfish (Poulson 1968 and unpublished) but it had a sudden depressing effect on the species diversity and abundance of the shoreline terrestrial community (Poulson and Culver 1969). This depressing effect was presumably due to toxic effects of the blue-green bacteria *Spirulina* that was left as a band along the shore as the pollution laden crest of water passed and receded. This influx of sewage lagoon waters caused the substrate organic content to increase from 1.05-1.23% to 2.44-3.06%.

Human And Animal Wastes Impact Cave Stream Communities Mainly By Favoring Species That Tolerate Low Oxygen And Are Reproductively Stimulated By Increased Food Supply

The destruction of the cave community in Hidden River Cave was due initially to a combination of past toxic effects of heavy metals pollution (Barr 1976) and the high biological oxygen demand of decomposing cheese whey and human wastes (Austin personal communication) but the current slow recovery seems due to residual effects of human wastes. My conclusion is based on natural experiments in other caves where the only pollution is by human or animal wastes. Holsinger (1966) and Lewis et al. (1982) provide qualitative observations, Weingartner (1977) provides detailed quantitative data, and I have preliminary quantitative data on different degrees of impact. At high impact levels, as directly under a feed lot or a large sewage treatment plant the high BOD kills all macroscopic organisms and leaves only strands of colonial sewage bacteria and associated protozoa. If the BOD does not completely remove oxygen then tubificid sewage worms become part of the community as has recently been the case in the south branch of Hidden River Cave (Lewis et al. 1982, Poulson unpublished). If the amount of wastes is not too great, as with septic field diffuse input or with dilution by less polluted water as in east branch of Hidden River, the sewage fauna drops out and the higher than normal food supply favors survival and reproduction of shorter-lived macroscopic cave fauna, especially troglophilic isopods, amphipods, and crayfish which may replace troglobitic species. As the input of waste becomes less accidental chironomid midges and troglophiles survive, but do not reproduce, and reproduction of short-lived troglobitic isopods and flatworms is stimulated. At still lower impact levels the reproduction of larger troglobites may also be stimulated. Far downstream from a presumed feed lot input into Black River of Roppel Cave there are incredibly dense populations of cave crayfish (Leitheuser personal communication) and by far the highest densities of meiofaunal nematodes (280/liter of sediment) and harpactacoid copepods

(858/liter of sediment) measured anywhere by Whitman (in Leitheuser et al. 1986). High meiofaunal densities may be an early warning sign of increasing human and animal waste pollution because normally the meiofaunal species diversity and density decreases downstream from vertical shaft inputs.

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I am indebted to a host of colleagues with whom I have discussed aquatic cave ecology over the past 30 years. My interpretations do not always match theirs but I could not have generated my ideas without their verbal input and their data. I cannot mention everyone who has helped to form the ideas expressed herein and/or has helped me in the streams in the Mammoth Cave Area but a few bear special mention. In alphabetical order they are Tom Aley, Bill Austin, Tom Barr, Nick Crawford, Dave Culver, John Cooper, Dave Griffith, Jack Hess, Tom Kane, Jim Keith, Terry Leitheuser, Jerry Lewis, Ed Lisowski, Eric Morgan, Rick Olson, Jim Quinlan, Stan Sides, and both Bette and Will White. Finally I could not have learned what I know without the continued cooperation of landowners and the help and encouragement of the staff at Mammoth Cave National Park and of the JVs and members of The Cave Research Foundation. I thank you one and all!

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BIOGRAPHICAL SKETCH

Tom Poulson got his B.S. at Cornell University and his Ph.D. at the University of Michigan. After a postdoctorate at U.C.L.A. he taught 10 years at Yale and 3 years at Notre Dame before coming to the University of Illinois at Chicago where he is now Professor of Biological Sciences. He is Chief Scientist of The Cave Research Foundation. He Studies morphology, physiology, life history, and behavior in ecological and evolutionary contexts. In caves he studies fish, crayfish, spiders, millipedes, springtails, flies, and beetles. Outside he has studied many communities including tropical forest dung beetles, salt marsh and desert birds, mountain and desert chipmunks, floating plants, dunes grasses, and forest trees.

USE OF BIOLOGICAL INDICATORS OF GROUNDWATER QUALITY
Thomas L. Poulson

Q. When using an Index of Biological Integrity (IBI) to assess groundwater contamination in a cave, how necessary is it to have a pre-contamination assessment? Will nearby cave streams be sufficient to provide a baseline?

A. It is sufficient to define maximum values for an IBI using the variation seen for the most pristine cave stream(s) in the local region. Only slight adjustments to expectations for non-impacted streams are made due to abiotic factors such as stream order and above or below average refuges from flooding and predation.

Q. How useful and reliable might the IBI approach be for karst springs where caves are not accessible?

A. The principles behind an IBI will hold for any aquatic community including karst springs. I would develop an IBI using spring basin faunas and floras for the region in question and, if possible, use a net to collect the organisms washed out of the spring over a week long period both at times of low and high flow. The problem with spring outwash data alone is that one cannot assess the underground habitat to determine what it can support and so it might be important to have pre contamination data on the outwash fauna for a particular spring system.

Q. Are the top species of the food chain sufficiently rapid in their responses to be useful bioindicators of pollution?

A. Fish and crayfish at the end of the food chain react quickly to some and slowly to other contaminants. Crayfish crawl out of the water when there is an acute input of a toxin at high concentrations. Increased proportions of females becoming fecund may indicate chronic input of human and animal waste at low concentrations. But neither long-lived fish nor crayfish may respond quickly to chronic input of toxins at low concentrations that take many years to bioaccumulate enough to cause visible tumors or obviously compromise reproduction. If toxin pollution is suspected then I would find out what is used in the watershed (which pesticides and herbicides) and monitor the most dangerous toxin in the water at peak flow when runoff will be greatest.

Q. Have you seen any significant changes in the aquatic fauna of Mammoth Cave since the 1950s and if so why have the changes occurred?

A. No species has been lost or gained but the diversity and abundance of the Echo-Styx River communities apparently declined since the early 1900s and has clearly declined since the 1950s when I first censused the area (Lisowski and Poulson 1979). At present cavefish, cave crayfish, and cave shrimp are either absent or very difficult to find and the upper level species of isopod may have replaced the usual river species. These changes were associated with changes in flow regime and water level starting with the construction of Lock and Dam #6 just downstream on Green River around 190 and then continued with the construction of the Nolin River Dam just downstream and the Green River Dam far upstream in the 1950s. I do not know the mechanism but have hypothesized that it is increased silt deposition in the cave. Siltation does decrease substrate heterogeneity and so may have decreased refuges of copepods, isopods, amphipods, and shrimp from floods and/or predation by fish and crayfish. Siltation may also cover or dilute particulate organic matter and so cause declines of the bacteria and protozoa which are eaten by copepods, isopods, amphipods, shrimp, and small crayfish.

THE USE OF BENTHIC MACROINVERTEBRATES FOR ASSESSING THE IMPACT OF CLASS V INJECTION WELLS ON CARBONATE GROUND WATERS

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ABSTRACT

The U.S. Environmental Protection Agency contracted the authors to investigate the impact of Class V injection wells on ground water in karst terranes to demonstrate the need for regulations over these shallow injection methods. Any sinkhole that has been modified to better accept drainage, including storm water runoff, was considered a Class V injection well. In addition, service station bay drains that lead to septic tanks, pits, or dry wells were a focus of the study.

The study was conducted in Cookeville and Johnson City, Tennessee, to compare contaminant transport in both flat-lying and folded rock. A major cave system underlies both cities. Ground water flow paths and velocities were determined using dye tracing methods. Water and benthic macroinvertebrate samples were taken where water entered Class V injection wells and at the springs influenced by these waters. As a control, samples were also gathered from a sinking stream and spring in a forested area. Concentrations of MTBE, BTEX, TPHC, ethylene glycol, and metals were below detection limit at most sites and, therefore, were not useful in demonstrating an impact. This emphasizes that grab samples in karst seldom indicate overall water quality conditions. In contrast, the Shannon Diversity, Shannon Evenness, and Family Biotic indices, coupled with a cluster analysis of the benthic data, were very useful in classifying sampling sites as to their degree of pollution where water chemical data did not indicate degradation.

INTRODUCTION

Part C of the Safe Drinking Water Act (Public Law 93-523) authorized the U.S. Environmental Protection Agency (EPA) to establish regulations to assure that potable ground water is not endangered by the underground injection of waste. Guidelines for underground injection and the classification of injection wells come under Part 146.04 of the Federal Underground Injection Control program (U.S. EPA, 1981). This program created five classes of underground injection wells. Classes I through IV include such categories as radioactive and hazardous waste injection and disposal of brines from the oil and gas industry.

Class V wells are generally defined as those which inject only nonhazardous fluids into or above strata that contain underground sources of drinking water (USDW). USDWs not only include aquifers which are currently serving as drinking water supply sources but also aquifers which are of acceptable quality for possible future use. Class V wells include any type of injection well not covered in the UIC definition of Classes I, II, III, or IV. EPA has classified Class V injection wells into six groups based in part on the expected quality of the injected fluid. The following is a listing of these groups as found in regulations for the State of Florida which is particularly sensitive to karst aquifers:

Group 1 - wells associated with thermal energy exchange processes, which include air conditioning return flow wells and cooling water return flow wells. Cooling water return flow wells may be part of a closed-loop system, with no hazardous additives;

Group 2 - recharge wells, saltwater intrusion barrier wells, connector wells, and subsidence control wells (associated with aquifer overpumping);

Group 3 - wells which are part of domestic waste treatment systems, swimming pool drainage wells, injection wells used in experimental technologies, wells used to inject spent brine into the same formation from which it was withdrawn after extraction of halogens or their salts;

Group 4 - nonhazardous industrial and commercial disposal wells, which include laundry waste, dry wells, sand backfill wells, and nuclear disposal wells used to inject radioactive wastes, provided the concentrations of the waste do not exceed drinking water standards contained in Chapters 17-22, FAC, and injection wells used for in-situ recovery of phosphate, uraniferous sandstone, clay, sand, and other minerals extracted by the borehole slurry mining method (lignite, tar sands, oil shale, coal);

Group 5 - lake level drainage and stormwater drainage wells; and

Group 6 - geothermal wells and "other" wells.

The U.S. EPA funded twenty-four projects nationwide under its Shallow Injection Well Initiatives Program to help evaluate the impact of Class V injection wells on ground water and to establish best management practices. The authors of this paper were chosen to help evaluate Class V injection well practices in the karst of Tennessee. Any sinkhole that had been modified to accept waste, including stormwater runoff, was considered a Class V injection well in the study.

OBJECTIVES

The primary goal of the proposed study was to determine if Class V injection wells have created a ground water pollution problem. Evaluating the effects of service station bay drains (Group 6) that lead to septic tanks, pits, or dry wells was a primary target of the investigation. Other high priority Class V injection wells that were investigated include: 1. Agricultural and municipal drainage into improved and unimproved sinkholes (Group 1), 2. Industrial drainage into sinkholes (Group 4), and 3. Domestic wastewater drainage into sinkholes (Group 5).

To fulfill these objectives, samples were collected from waters entering sinkholes and the springs hydrologically connected to them as determined by ground water tracing methods. A benthic macroinvertebrate study was then conducted at most of the sites to evaluate the impact of contaminants on stream biota. Since ground water chemistry in karst terranes changes rapidly during storm events (Ogden, 1988 and Quinlan, 1989), it was felt that the benthic survey might prove a more reliable indicator of long-term water quality.

HYDROGEOLOGY OF THE STUDY SITES

Two geologically different karst terranes in Tennessee were chosen so that the results would have applicability throughout much of the karst in the United States (Figure 1). Around

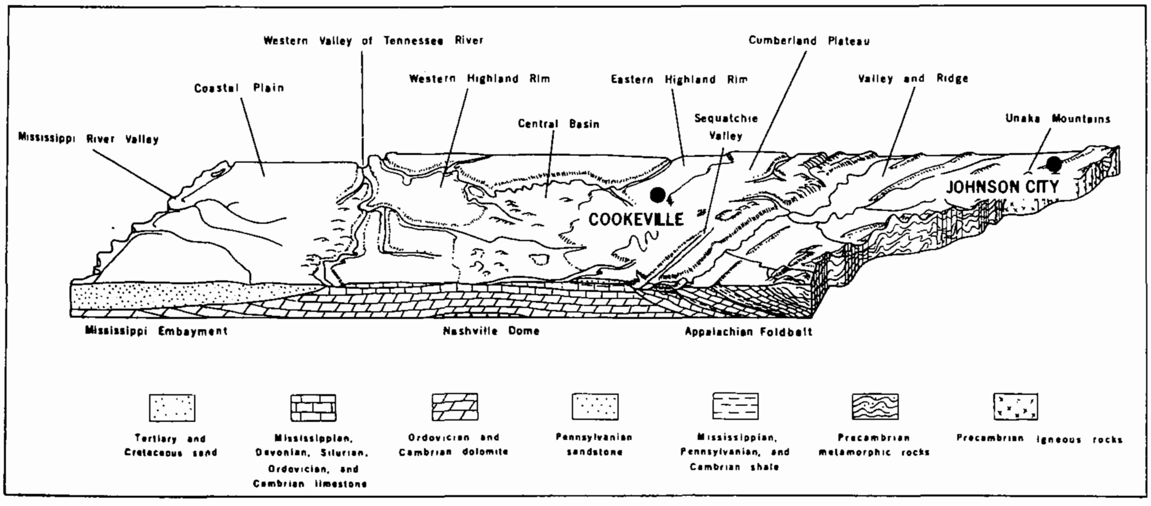


Figure 1. Location of study areas (adapted from Miller, 1974).

Cookeville (Putnam County), the Mississippian-aged carbonates of the Eastern Highland Rim Province are flat-lying allowing ground water to move along a wide range of orientations corresponding to joint and photo-lineament trends (Ogden et al., 1989). Streams originating on the Cumberland Plateau east of Cookeville flow over shales and sandstones until underlying limestone beds are intersected. Some streams sink when they reach the Bangor Limestone while others sink into the Monteagle or St. Louis limestones. Subterranean water moves through caves, pits, and solution-enlarged fractures until emerging as spring flow, commonly at the St. Louis-Upper Warsaw boundary. Within much of the city limits of Cookeville, surface runoff is perched on the upper Warsaw Formation which is a cross-bedded, reddish-brown, sandy limestone. This water then sinks into the middle limestone member. A four-mile-long cave that drains much of Cookeville's storm water runoff occurs within this unit. The lower Warsaw member is a relatively impermeable calcareous siltstone and shale with some argillaceous limestone beds. The largest springs around Cookeville are found perched above this lower member.

Ground water conditions in Putnam County have not been extensively studied. An early regional analysis of the aquifers and a listing of water wells was made by Smith (1962). A ground water resource analysis around Center Hill Lake, which included the Cookevile area, was made by Moore and Wilson (1972). A water quality survey of some water wells around town was performed by Collar and Ogden (1990). Faulkerson and Mills (1981), Pride et al. (1988), and Hannah et al. (1989) have performed some dye traces around Cookeville. Physiochemical degradation of the water within the four-mile-long cave beneath Cookeville has been documented by Smithson (1975), Faulkerson and Mills (1981), Wilson (1985), and Pride et al. (1988). A recent assessment of sinkhole flooding in Cookeville from storm water drainage has been conducted by Mills et al. (1991).

The second study area used for the investigation occurs within the city limits of Johnson City (Washington County) located within the Appalachian Valley and Ridge Province of eastern Tennessee (Figure 1). In the Valley and Ridge Province around Johnson City, the Ordovician-aged Knox Group carbonates are complexly folded and faulted, and ground water moves predominantly along stratigraphic strike within solution-enlarged bedding planes. The Knox Group is over 3,000 feet thick and has not been subdivided in the study area. Much of Johnson City is developed on a sinkhole plain. Many of the sinkholes were designated as Class V injection wells by this study due to storm water routing into them and/or the presence of discharging pipes (often of unknown origin) within the sinkhole basin.

Early reconnaissance ground water studies of the Johnson City area were performed by Maclay (1956) and DeBuchananne and

Richardson (1956). Many of the public water supplies near Johnson City utilize springs that emerge from cavernous limestones and dolomites. The vulnerability of these municipal water supplies prompted the First Tennessee Development District (Stanley Consultants, Inc., 1983; Matthews, 1986; and Brown, 1987) and TVA (Foxx, 1981) to make a survey of potential pollution sources within close proximity of the springs. Other springs in the area have shown probable contamination from septic tanks (Wilson et al., 1988). Many of the eastern Tennessee springs become turbid after storm events and show elevated levels of fecal coliform (Brown, 1987). This reflects the impact that natural and anthropogenic activities in the recharge area have on ground water quality. Ogden et al. (1991) recently completed a "wellhead" protection study of nine municipal-used springs in eastern Tennessee that involved ground water tracing and a time-series analysis of water quality. The springs within the city limits of Johnson City were not included in any of these studies.

No ground water tracing had been conducted in Johnson City until initiation of this project. Field investigations performed for this study showed that most of the storm drainage in north Johnson City flows through cave systems and then emerges at springs along Knob and Cobb creeks which flow into the Watauga River. These creeks are being studied by the U.S. Geological Survey as part of the nationwide effort to monitor urban storm water runoff quality (Johnson, personal communication).

METHODS

Samples were collected of waters entering Class V injection wells (sinkholes) and at the springs influenced by these waters during the 1991 wet and dry seasons. As a control, samples were also gathered from a sinking stream and a spring with a predominantly forested recharge area. Dissolved oxygen (DO), pH, conductivity, and temperature were measured in the field with a Hydrolab field monitor. Laboratory analyses used as indicators of contamination from agricultural activities and septic tanks included nitrate, chloride, fecal coliform bacteria, and fecal streptococcus bacteria. Zinc, chromium, lead, methyl tertiary butyl ether (MTBE, a gasoline additive), benzene-toluene-ethylene-xylene (BTEX), total petroleum hydrocarbon (TPHC), and ethylene glycol levels were measured as indicators of waste products and leaks from service stations and parking lot runoff. Seven springs and seven sinking streams were sampled around Cookeville; whereas, six springs and three sinking streams were sampled around Johnson City.

To enhance the interpretation of the impact of Class V injection wells on spring waters, benthic macroinvertebrate samples were collected at riffle areas and pools at most of the injection points and the springs. Samples were collected with a modified kick net and a Surber sampler (0.09 m^2). All organisms were preserved in a 10% formalin solution, enumerated, and identified to the genus level when possible. Statistical analysis of the benthic macroinvertebrate data were then conducted.

Qualitative ground water tracing was conducted using fluorescein dye and activated charcoal detectors. Optical brighteners and cotton detectors were also used for tracing. A control packet was placed at a spring known not to be hydrologically connected to the tracer input site during each test.

RESULTS

Water Quality Sampling

Both wet and dry season samples from nearly all of the twenty-three sites in the two study areas showed below detection limits for BTEX, MTBE, ethylene glycol, chromium, lead, and zinc. One site, the storm drain for Tennessee Technological University, showed below detection for BTEX in one sample and high levels in another (benzene - 1200 ppb, toluene - 220 ppb, xylene - 780 ppb). Fumes were reported in this drain which initiated its sampling. The drain receives runoff from the university's garage and lies next to several gasoline underground storage tanks. In addition, a nearby underground storage tank used to provide fuel for a background power supply in the basketball arena was recently pulled, and soil samples showed TPHC levels as high as 2088 ppm. No remediation or site characterization was performed at the site, and a new building has since been constructed over it. The storm drainage system beneath the university has been diverted into a large sinkhole which also receives storm water runoff from other areas of Cookeville. Raw sewage from the city lines is occasionally bypassed into the same sinkhole. Ground water tracing showed a hydrologic connection between the sinkhole and Big Spring.

MTBE was found in only two samples, and it was at low levels (< 5 ppb). TPHC was above the detection limit at only one site during the wet season with a level of 309 ppb. Sampling during the dry season showed detectable TPHC at only the Cookeville city garage drain site with a value of 6 ppm. Zinc was measured at levels above detection at five sites with concentrations being up to 73 ppm.

Chloride levels were significantly higher at springs receiving recharge from Class V injection wells compared to the control springs. The control springs had chloride levels between 1 and 3 ppm; whereas, the other sites ranged from 3 to 64 ppm with an average of 14 ppm. Nitrate concentrations were below 2 ppm at all but two sites. The highest measured value was 5 ppm. Fecal coliform and fecal streptococcus bacteria counts were dramatically different between the wet and dry sampling periods. The wet season samples were relatively free of these bacteria. Only five of the samples had fecal streptococcus bacteria with the highest count being only 21 colonies/100 ml. All but two of the sites had fecal coliform bacteria counts over fifty with the highest being 350 colonies/100 ml. In contrast, most of the dry season samples had fecal streptococcus levels that were too numerous to count (TNTC). Fecal coliform bacteria counts were greater than 500 colonies/100 ml at thirteen of the sites for the dry season samples and only one site had less than 25 colonies/100 ml. The storm drain at Tennessee Technological University and Big Spring had the highest fecal coliform bacteria counts in the Cookeville samples related to leaky sewers and bypassing of raw sewage (2900 colonies/100 ml and TNTC, respectively). The fecal coliform/fecal streptococcus ratios suggest that most of the other sites are being polluted from animal versus human wastes.

<u>Benthic Macroinvertebrate Sampling</u>

Benthic macroinvertebrate samples were collected from twelve sites in the two study areas. To date, identifications have been completed for six samples collected during the wet season. As a control, a spring with a forested recharge area was chosen for comparison to the springs with urban recharge areas. Macedonia Spring in Cookeville and Taylor Spring in Johnson City served as the control springs.

Shannon Diversity indices (Washington, 1984) were calculated on each spring and sinking stream (Table 1). Wilhm (1970) found values between 3 and 4 for the Shannon Diversity Index ($\bar{d}$) in unpolluted waters, but in badly polluted waters it was generally less than 1. All of the sites shown on Table 1 exhibited low Shannon Diversity (Base 2) values with Macedonia Spring having the highest (2.77) indicating only slight organic pollution. However, EPA biologists for the southeastern region have found $\bar{d}$ not to be sensitive enough and have used equitability e (Shannon Evenness) to better demonstrate degradation of stream waters (Weber, 1973). E is sensitive to slight levels of degradation. Polluted streams usually have values between 0.0 to 0.5; whereas, unpolluted streams fall between 0.5 to 1.0. In the Cookeville area, the Shannon Evenness Index (Table 1) clearly distinguishes the polluted urban springs (Big and Pigeon Roost) from the rural spring (Macedonia). In the Johnson City area, this distinction is not as apparent. This may be due to Taylor Spring's immediate confluence with a surface stream which may not allow time for the benthic environment to change from heterotrophic to autotrophic. According to Vannote et al. (1980), this change is necessary for increasing species diversity. Also, the substrate at Taylor Spring is almost completely composed of sand, silt, and clay with nearly no gravel or cobble-size material.

Table 1. Shannon Diversity, Shannon Evenness, and Family Biotic indices calculated for the wet season samples.

	Shannon Diversity (Base 2)		Shannon Evenness		Family Biotic Index	
Macedonia Spring	2.77	SP	0.69	UP	3.97	VG
Big Spring	0.88	P	0.31	P	5.84	FP
Pigeon Roost Spring	0.15	P	0.08	P	6.01	FP
Taylor Spring	2.14	SP	0.58	UP	6.08	FP
Shoe Spring	1.64	SP	0.55	UP	7.05	P
Car Wash Insurgence	2.15	SP	0.68	UP	5.50	F

P = polluted; SP = slightly polluted; UP= unpolluted; VG = very good; F = fair; FP = fairly poor; P = poor

Macedonia Spring, Big Spring, and Pigeon Roost Spring also emerge from caves but their substrates are composed primarily of gravel, cobble, and boulder sizes allowing for more diversity and greater numbers of species. Car Wash Insurgence and Shoe Spring have substrate compositions composed of gravel and cobble mixed with sand, silt, and clay.

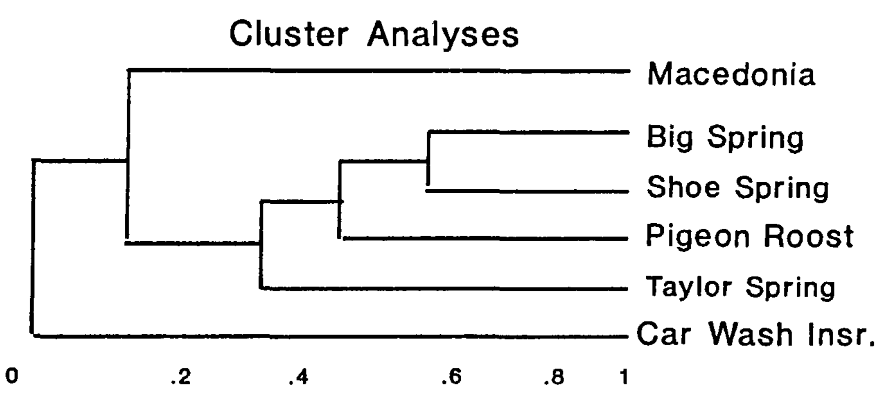
Hilsenhoff's (1988) Family-Level Biotic Index (FBI) was also calculated for each site to see how it compared to the Shannon Base 2 values and equitability e values (Table 1). In general, the higher the FBI, the greater is the amount of organic pollution with 10 being the highest number possible. In the Cookeville area, the rural spring was rated as very good while the urban springs were ranked as fairly polluted. In the Johnson City area, a distinction between the rural and urban sites was not seen, probably for the reasons discussed previously. Visual and olfactory evidence of pollution was found at the two urban springs in Johnson City, and water quality differences between the two groups were seen based on chloride and conductivity results (Table 2). This water quality data helps support the substrate and heterotrophic vs. autotrophic hypotheses just presented.

A cluster analysis of the benthic macroinvertebrate data was also performed for the study sites. The cluster analysis, together with the Shannon Diversity, Evenness, and the Family Biotic indices, gave further insight to which sites have threatened benthic communities and possible ground water degradation. Big, Shoe, and Pigeon Roost springs are polluted by city runoff into Class V injection wells and thus cluster together (Figure 2). Low number of genera at Taylor Spring, related to substrate composition, caused it to be grouped with

Table 2. Wet and dry season chloride and conductivity values.

	Chloride (Mg/L)		Conductivity (μmhos/cm)	
	Wet season	Dry season	Wet season	Dry season
Macedonia Spring	1.70	2.40	364	142
Big Spring	4.00	15.60	158	236
Pigeon Roost Spring	18.20	56.30	286	262
Taylor Spring	2.82	2.30	281	572
Shoe Spring	7.92	7.39	478	442
Car Wash Insurgence	10.20	11.80	475	863

the polluted urban springs. Macedonia Spring, which has relatively pristine water, was grouped alone. Car Wash Insurgence, a small sinking stream, was clustered separately due to large annual water temperature fluctuations which have a significant effect on the biologic community structure not seen at the springs.



Linkage	Clusters	Linked	Similarity	Prob
1	Big Spring	Shoe Spring	.50446	.29000
2	Big Spring	Pigeon Roost	.38494	.15000
3	Big Spring	Taylor Spring	.28472	.10000
4	Macedonia	Big Spring	.04615	.01000
5	Macedonia	Car Wash Ins.	.00098	.01000

Figure 2. Cluster analysis results.

CONCLUSIONS

The water quality tests performed for this study were not definitive in assessing the impact on karst ground waters of volatile chemicals entering Class V injection wells. The open conduit flow in the karst aquifers allows contaminants to be rapidly flushed through the ground water system. Nearly continuous water quality monitoring would be necessary to detect contaminants dumped or spilled into Class V injection wells such as from service station bay drains. In addition, turbid flow in the caves and sinking streams causes rapid degassing of the volatile components, thus explaining why some sample sites showing the presence of oil and grease had no measurable amounts of volatiles. Also, if volatiles were detected, it would be nearly impossible to determine if they were derived from bay drains or from leaky underground storage tanks. Fecal coliform and fecal streptococcus bacteria levels, on the other hand, were very high during the dry season demonstrating the impact on ground water from leaky sewers, sewage bypassing, and animal wastes entering Class V injection wells. Chloride and conductivity values were also useful in demonstrating ground water degradation.

Benthic macroinvertebrate analyses may be better for accessing the impact of Class V injection wells on ground water since these organisms can be very sensitive to daily water changes not detected by collecting occasional grab samples for chemical analysis. In this study, the Shannon Diversity, Shannon Evenness, and Family Biotic indices, in conjunction with the cluster analysis, proved useful in distinguishing spring waters that have been degraded by pollutants entering Class V injection wells.

ACKNOWLEDGEMENTS

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Albert E. Ogden received his B.S. degree in geology from the Pennsylvania State University and his Ph.D. in hydrogeology from West Virginia University. He was the hydrogeology professor for the University of Arkansas from 1976-1981. He then was assistant director and hydrogeologist for the Edwards Aquifer Research and Data Center in Texas from 1981-1985. Following this, Dr. Ogden was the senior hydrogeologist for the RCRA and Superfund programs within the Idaho Division of Environment from 1985-1987. He presently is an associate professor for the Center for the Management, Utilization and Protection of Water Resources, Box 5033, Tennessee Technological University, Cookeville, Tennessee 38505, (615) 372-3353. Dr. Ogden's research interests include site characterization of hazardous waste sites and solving water quality problems in karst terranes.

Ronald K. Redman received his B.S. degree in biology from Arkansas Tech University. He then worked as a manager for a fish hatchery where his duties included water quality monitoring. He now is working on his M.S. in biology at Tennessee Technological University, and this paper is a portion of his thesis research.

Teresa L. Brown received her B.S. degree in geology from James Madison University followed by graduate work in the Hydrology and Environmental Science Program at Indiana University. After college, she worked as a geologist and pollution control specialist for the Virginia Division of Mined Land Reclamation. Since 1986, she has been the environmental management coordinator for the First Tennessee Development District, 207 North Boone Street, Suite 800, Johnson City, TN 37604, (615) 928-0224. Ms. Brown has experience in siting landfills, assaying the impact of land application of sludge, conducting water quality surveys, and delineating wellhead protection areas.

The Use of Benthic Macroinvertebrates for Assessing the Impact of Class V Injection Wells on Carbonate Groundwaters

Albert E. Ogden, Ronald K. Redman, and Teresa L. Brown

Question 1. The Shannon-Wiener diversity indices are problematic because they:

a) confound the numbers of species and evenness of numbers of individuals among species; and

b) count tolerant species, e.g., surface chironomids, the same as obligate cave species (e.g., *Caecidotea* isopods). So, what were the species and their degrees of restriction to caves and groundwater in springs with outflows from control and contaminated basins?

Response: No samples were taken in caves.

Question 2. Most, if not all, of the insect orders of your control surface streams are not normally found in caves. What orders of Crustacea and what kinds of species (Facultative vs. obligate cave species) did you find in outflows from your control and contaminated basins?

Response: In the control springs, we found decapods, amphipods, and isopods. We also found them in some contaminated springs, but amphipods and/or isopods were usually absent.

Session VII:

Ground Water Monitoring

The Response of Landfill Monitoring Wells

In Limestone (Karst) Aquifers

To Point Sources and Non Point Sources of Contamination

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Abstract

The terms "point source" and "non-point source" in the context of groundwater contaminants are relative terms. Landfills can be considered non-point sources, relative to a monitoring well at the boundary of a typical landfill, if contaminants are released at hundreds of points, Conversely, the landfill is a point source relative to these wells if contaminants are released at one or two points.

This author and several of his colleagues have maintained that the convergent flow in karst aquifers should make monitoring wells unreliable. Implicit in these arguments was the assumption that the landfill was a point source of contamination.

In the non-point source case, virtually every joint and bedding plane would be exposed to contamination. Thus, any down-gradient monitoring will which intersects the groundwater should show contaminants, although they may be unrepresentative of leachate from the entire landfill.

In the point source case, the monitoring well may not intersect the conduit which carries the contaminant. This contention was confirmed at a western Kentucky landfill where rhodamine-WT was injected into an up-gradient well, simulating a point source of contamination. Two down-gradient monitoring wells did not show the dye. although the dye appeared at three off site springs approximately three miles distant.

Therefore, neither contaminated nor uncontaminated landfill monitoring wells in karst aquifers should give us confidence in their reliability.

Introduction

Landfills As Non Point Sources Of Contamination

When a monitoring well is placed down-gradient from a landfill an assumption is made that it will intercept a plume of contamination if the landfill leaks. If the leachate escapes uniformly over the entire liner or at hundreds of points, the traditional system of one monitoring well up gradient and three down-gradient will signal that this has occurred (Fig. 1). Relative to the monitoring wells, such a landfill is a non-point source of contamination.

Although the leakage in this case is homogenous the leachate may not be. Should the most hazardous waste be located in a single cell, and if that cell is located along a flow vector midway between two of the monitoring wells, dispersion may be insufficient to bring this substance to these monitoring points. The contamination detected by the monitoring system would not be representative of the chemical nature of the leachate as a whole. Traditional monitoring wells may not provide a representative sample of leachate unless the landfill waste is homogenous.

Landfills As Point Sources Of Contamination

If the leachate passes through the landfill liner at a small number of discrete points, the traditional monitoring well system may not signal that this has occurred or it may give an incomplete understanding of the problem (Fig. 2). Only favorably located points of leachate escape may develop plumes which intersect the wells. Relative to the traditional monitoring wells, such a landfill is a point source of contamination.

This scenario also presents problems of obtaining a representative sample. Should the landfill waste be less than homogenous, the most troublesome leachate may go undetected. This would be the case unless each point of release is sufficiently distant from the well so that dispersion has time to spread the contaminants, or unless each point of release is directly up-gradient from a monitoring well.

Landfills In Karst

An additional complication arises when the aquifer beneath the landfill is composed of carbonate rocks. These aquifers often possess very little primary (intergranular) porosity but they exhibit significant fracture porosity (secondary porosity) and tertiary conduit porosity. While contaminant plumes may develop in the primary and secondary porosity, the plumes may be truncated and their contaminants entrained by the conduit porosity (Fig. 3).

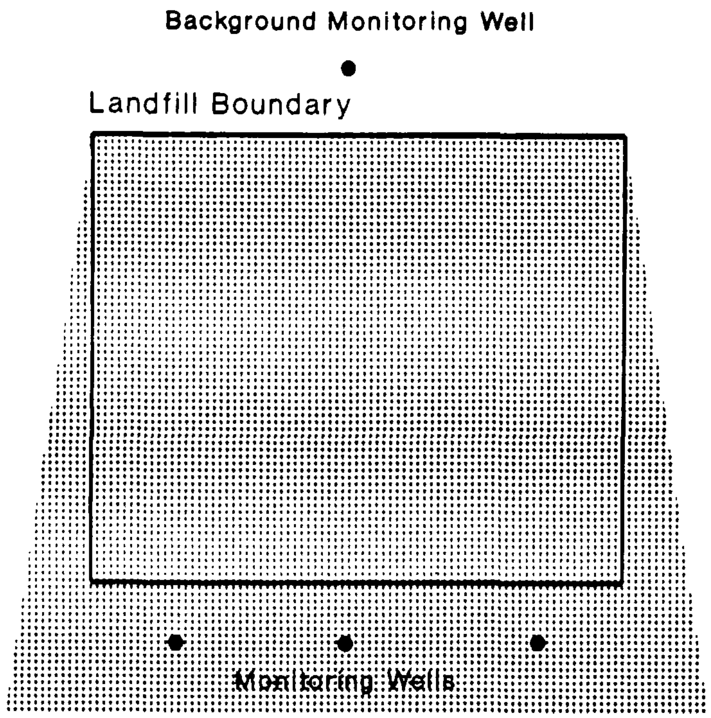

Figure 1. The contaminant plume at a hypothetical landfill which releases leachate uniformly over its entire liner. The stippled area indicates the plume.

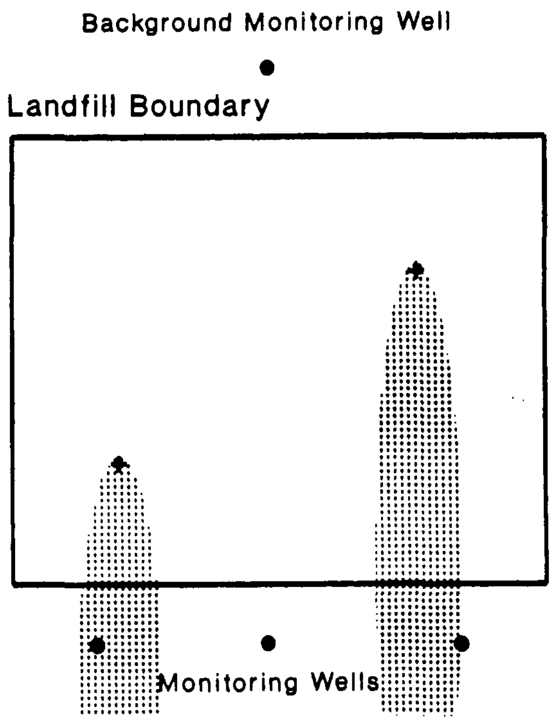

Figure 2. The contaminant plumes at a hypothetical landfill which releases leachate at two discrete points. The stippled area indicates the plume.

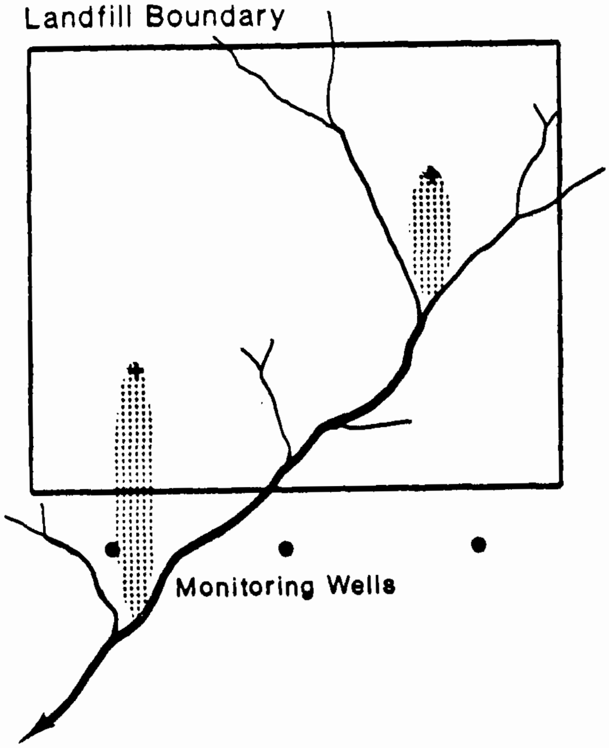

Figure 3. The contaminant plume at a hypothetical landfill in a karst aquifer with a high density of solution conduits. Leachate is released at two discrete points. The presence of conduit porosity is indicated by the black branching line.

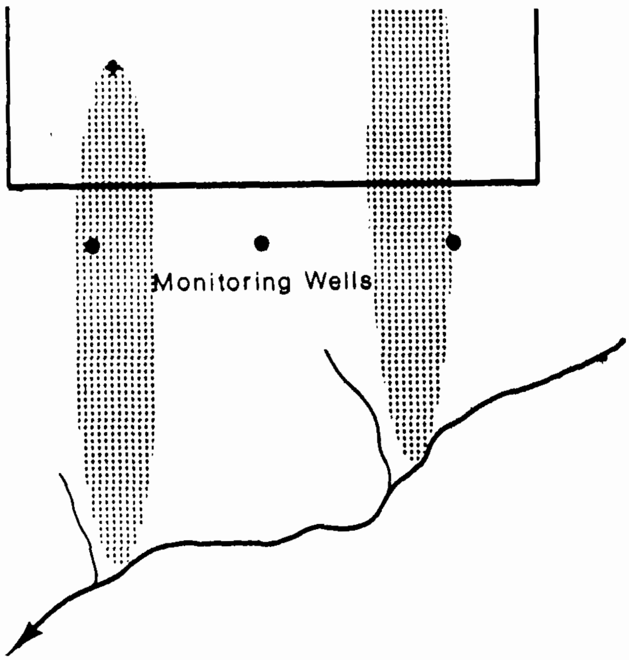

Figure 4. The contaminant plume at a hypothetical landfill in a karst aquifer with a low density of solution conduits. Leachate is released at two discrete points. The presence of conduit porosity is indicated by the black branching line.

If the landfill leaks at discrete points, the point source case, there is a considerable likelihood that the traditional monitoring wells will not function as they were intended. Quinlan and Ewers (1985) made this point, although they did not specifically describe the development of plumes and their spatial relationship to the conduits. The probability of the wells working as intended in carbonate aquifers will depend not only upon the geometric relationship of the points of leachate release and the monitoring wells, but also upon the density of conduits in the aquifer. If the conduits are widely spaced, the probability of their being located between the contaminant source and the monitoring wells will be less and the monitoring wells are more likely to be efficacious (Fig. 4). Conversely, if the density of conduits in the aquifer is great, it would be unlikely that the wells would be useful.

If the landfill is a non-point source, virtually every joint and bedding plane beneath the landfill and immediately down-gradient would be affected by leachate. Although some portion of the leachate may be intercepted by conduits, any nearby down-gradient monitoring well will intercept some leachate. However, samples from such a well are unlikely to be representative of the landfill leachate unless the waste is homogenous.

A Kentucky Landfill Example

A recent study at a Western Kentucky landfill confirmed the contention that traditional monitoring wells may be ineffective in limestone aquifers. Two down-gradient monitoring wells at this site were tested for efficacy by injection of a tracer into the up-gradient background monitoring well. The landfill is located in a karst region. The aquifer beneath the site is a Mississippian age limestone, the upper member of the St. Louis Formation.

Four pounds of 20 percent Rhodamine-WT dye was injected into the up gradient well. The dye was diluted in 50 gallons of water before injection. The well was first tested by an injection of several hundred gallons of potable water. This insured that the well would accept the dye and also served to wet the pathways that the dye could follow. After the dye injection several hundred gallons of potable water was injected into the well to flush the dye into the aquifer.

The results of the dye study are depicted in Figure 5. Rhodamine-WT from the well injection appeared in the Town Spring at Cadiz, Kentucky and an adjacent spring. These two sites are probably part of a local distributary system. This was the principle location of dye recovery from the well injection. Rhodamine-WT was also found in very low concentrations at two other springs, one north and the other southeast of the landfill. The

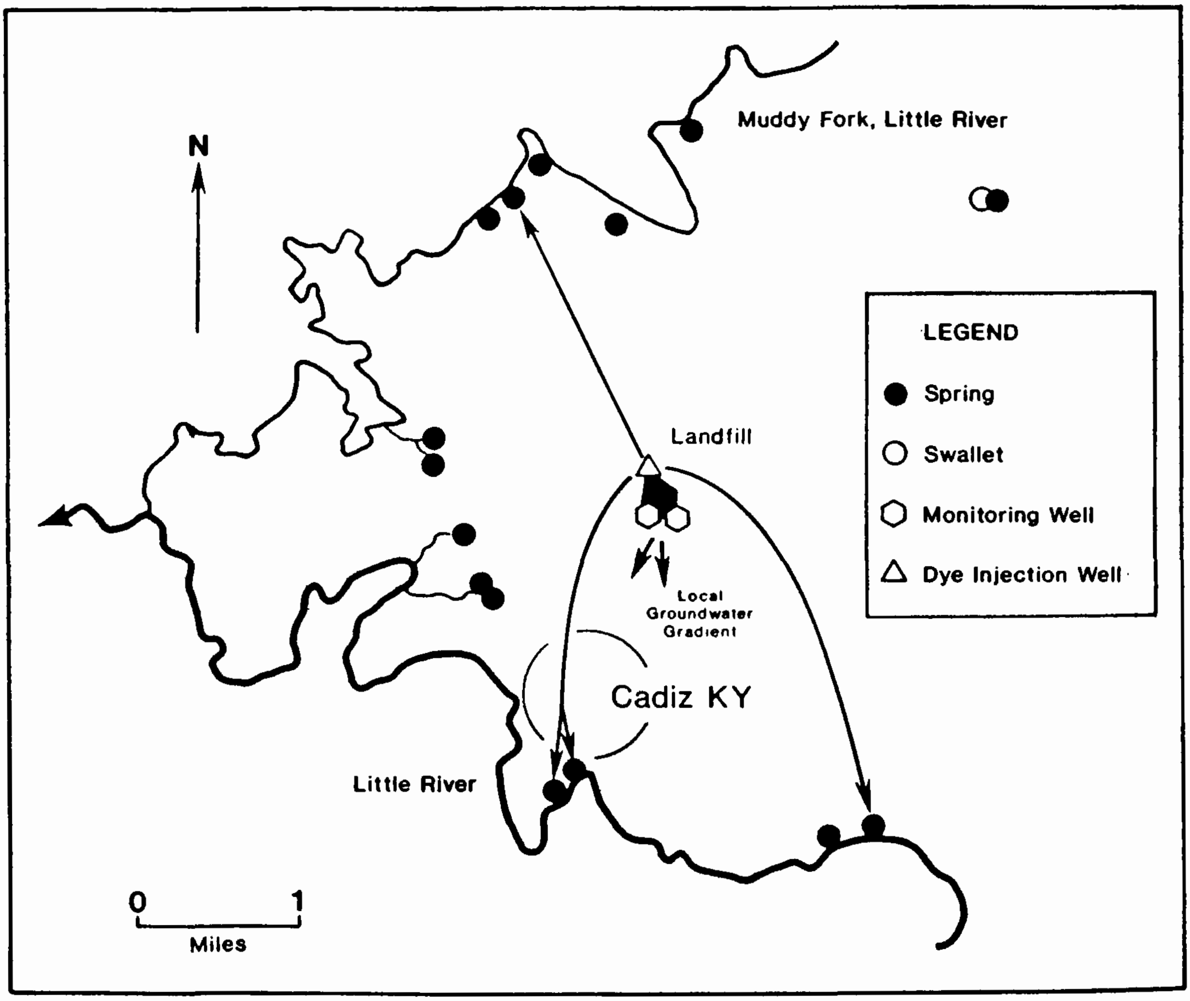

Figure 5. Results of a dye study at a Western Kentucky landfill. The long arrows indicate the schematic path of the dye. The principal dye recovery point was at the Cadiz town spring. The local gradient at the landfill is indicated by the short arrows.

dye reached all of these sites nearly three miles distant in less than four days.

No dye was recovered from charcoal dye detectors in the two proposed down-gradient monitoring wells located 1100 and 1300 feet from the dye injection well. Both of these wells were equipped with airlifts which continuously circulated the water in the well over the dye detectors. Dye detectors were exchanged regularly over a period of 38 days. Apparently the tracer was intercepted by conduit porosity before reaching the monitoring wells.

Appropriate Methods Of Monitoring In Karst

Quinlan and Ewers (1985) suggest that appropriate monitoring methods in karst should utilize springs. When it can be demonstrated by proper dye-tracing procedures that a particular spring drains a landfill site it may be the most appropriate and dependable monitoring site. However, springs are, in some instances, demonstrably inappropriate monitoring sites, even though they are shown to be connected to the area of potential contamination release. This is the case at the cited Kentucky example for two reasons. First, the spring is quite large and therefore the dilution of landfill leachate would be very great. Only a near catastrophic release would likely be detectable. Second, the probable recharge basin for the spring contains many potential sources of contamination. It could be very difficult to assure that a given contaminant was the result of landfill leakage or due to some other problem.

A alternative monitoring procedure which is more likely to be specific to the landfill would be to employ a pumped monitoring well at the site. During dye tracing programs in karst, wells are frequently employed as dye recovery sites. Those that are even modestly pumped during the tracing period often show the dye, those which are not pumped rarely do. An excellent example can be found in the work of Aley at a proposed landfill site near Pindall, Arkansas (Aley, 1986). In this study, dye was recovered from five wells, ranging up to a mile from both the dye injection point and the spring resurgences where the dye appeared. Apparently, local gradients serve to carry dye, and presumably contaminants, to the conduits, establishing the convergent flow which is characteristic of these aquifers. Pumping stress apparently reverses these local gradients, extracting the entrained dye from the conduit (Fig. 6).

If tracer dyes can be recovered from pumped wells in karst, by inference, contaminants could also be recovered from such wells. A protocol for establishing a pumped monitoring well would be as follows.

1- Locate the lowest point on the potentiometric surface, down-gradient from the site to be monitored. This could be done with

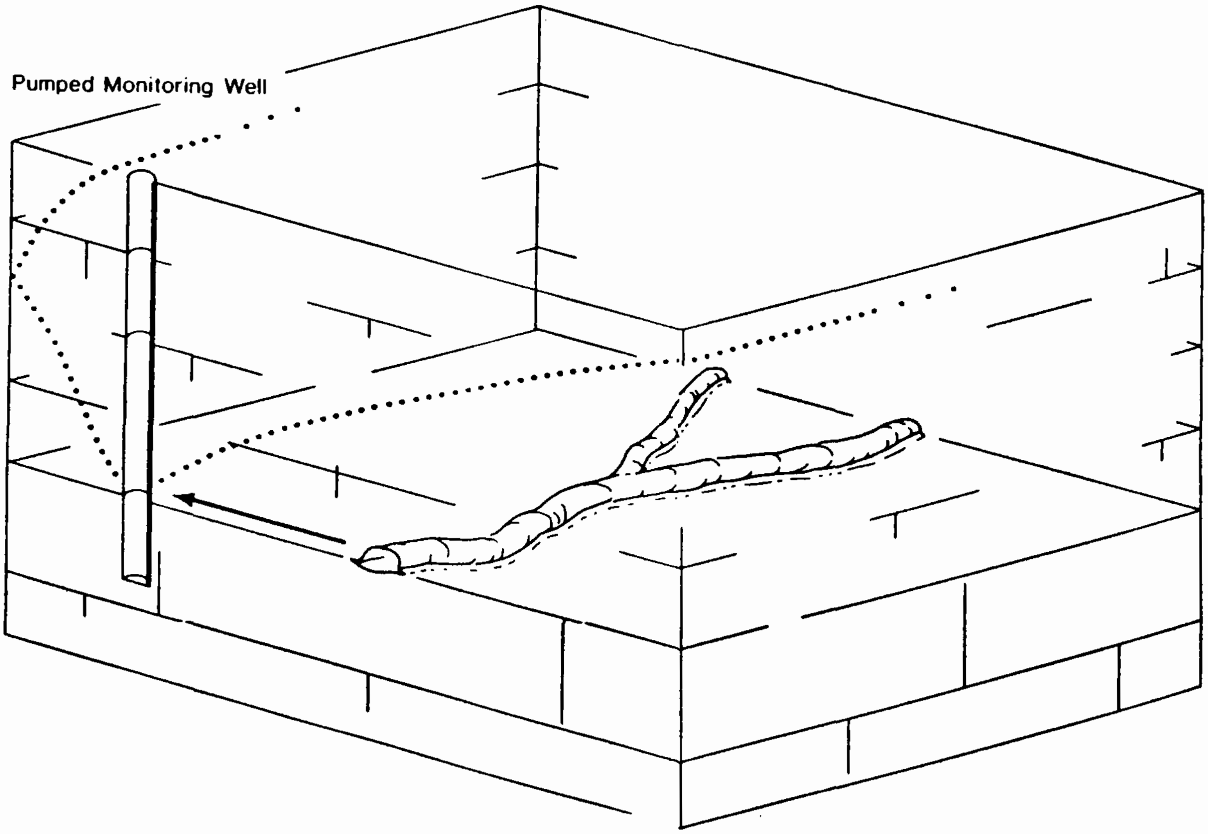

Figure 6. A pumped monitoring well lowers the potentiometric surface (dotted line) near a conduit which is carrying contaminants. This pumping stress draws contaminants to the monitoring well that would otherwise remain entrained in the conduit.

several small-bore temporary piezometers. Low points of the potentiometric surface correlate well with the location of conduits (Quinlan and Ewers, 1981).

2- Install a well at this point and outfit it with a submersible domestic well pump having a capacity sufficient to create drawdown in nearby piezometers in a short time. A site so located should be near conduits which may drain from the landfill.

3- Qualify the well with a dye test from an appropriate up gradient site or sites. This would help to define the capture zone of the proposed pumped monitoring well. If the capture zone is sufficient to encompass the site it could be used as a monitoring point.

4- Sample the well for contaminants on the required schedule after an appropriate amount of pumping has occurred. The amount of pumping required could be determined from "3" above.

Two monitoring wells of this type are ready to be installed at two sites, one in Kentucky and one in Puerto Rico.

Conclusions

Clearly, the traditional monitoring wells in the cited example cannot be relied upon to accurately indicate the performance of the landfill's clay liner. If one of the wells should show the presence of leachate, one cannot be confident that it monitors the entire landfill or that it fairly represents the character of all of the leachate that is released. Neither contaminated nor uncontaminated traditional monitoring wells in karst aquifers should give us confidence in this method of monitoring. Springs or pumped monitoring wells offer a more reliable alternative.

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Biographical Sketch

Ralph O. Ewers is professor of geology and director of the Groundwater Research Laboratory at Eastern Kentucky University, and a principal in Ewers Water Consultants, a consulting firm specializing in carbonate aquifers. His B.S. and M.S. degrees in geology were earned at the University of Cincinnati and his Ph.D. was earned at McMaster University (1982) . Professor Ewers's special interests include the applications of tracer and electronic monitoring techniques to the solution of practical environmental problems in karst groundwater. He has more than 30 years of research and consulting experience gained throughout much of North America and Europe and he serves on several state groundwater advisory boards. He is author or co-author of more than 100 papers and reports dealing with groundwater in karst. For one of the papers concerning procedures for monitoring ground water in karst terraces, he and co-instructor James Quinlan received the 1986 E.B. Burwell Award from the Geological Society of America for a "work of distinction which advances knowledge concerning principles of practice of engineering geology."

The Response of Landfill Monitoring Wells In Limestone (Karst) Aquifers To Point Sources and Non-Point Sources of Contamination

Ralph O. Ewers

1. In England and Wales, 56 domestic (municipal?) waste sites are located on the karstified carboniferous limestone. Unpumped monitoring wells have been successfully used to monitor contaminant plumes leaving the sites, yet only 4 springs have been affected. Why do you think so few springs have been adversely affected? Why do you think the monitoring wells have been so successful?

ANSWER- I suspect that the reason so few springs seem to be contaminated is that dilution of the contaminants is too great. This is one of the reasons I believe that springs are often inappropriate monitoring points.

A successful monitoring well can be defined several ways. If you define success as having no contamination, you may be deluding yourself. The aquifer may be contaminated you are just unaware of that fact. If you define success as showing some contamination, you may be unaware of the real scope of the problem.

2. Once you have determined the most appropriate well location (at the lowest point on the potentiometric surface), how do you determine the "appropriate" pumping rate?

ANSWER- I would judge the pumping rate and time on the basis of the tracer tests that are used to qualify the well. If pumping at a given rate gives a strong dye concentration which appears to plateau after pumping 2000 gallons of water, I would assume that would be an appropriate rate and quantity.

3. In many county landfills in Tennessee, no municipal sewer system is available to receive water pumped as you propose. The volume of water you imply would be pumped would be a problem. Do you have any suggestions for disposal of such pumped water?

ANSWER- I would store the water until after the analysis was complete and then release it, if it is clean, to some appropriate receiving stream, assuming the appropriate discharge permits were obtained. Alternatively, it could be trucked to a disposal facility.

4. How often do you propose to continue pumping to determine if there is leakage? What do you propose to do with the volume of water? Once the initial pumping test has been conducted and, for example, the results indicated that the landfill is not leaking are your proposing to cease pumping at the monitoring point until the next scheduled monitoring date, at which time you would begin the entire pumping and sampling procedure again? Would additional

wells be installed to enhance interception of the leakage?

ANSWER- I would pump as often as required by law or good science. Perhaps quarterly, I would prefer to pump at during low-flow conditions. I am not sure that high-flow samplings are necessary as they may be for spring monitoring.

See 3 above regarding disposal of the water.

I see no reason at this time to continue pumping between sampling periods.

Additional pumped monitoring wells may be required if the landfill is large or if the dye test results indicate that it is infeasible to extend the capture zone over the required area.

5. I suspect that the cost of continuing to analyze for pollutants in your well pumped to draw water from the conduit, plus the cost of water handling and disposal, would make it cheaper to combine a natural potential survey with your potentiometric map and drill several "pilot" holes (let's say 5) as potential monitoring wells. If one of the 5 holes hits the conduit (as proposed by tracing) you have maximum reliability and no problems with water disposal. Convince me that I am wrong!

ANSWER- I doubt that the natural potential method is developed sufficiently at this time to find the smaller conduits which are likely to be associated with the average landfill. If a large conduit is suspected, the type investigated by A. Lang and others, I would certainly advise using this technique.

DEVELOPMENT OF AN ASTM STANDARD GUIDE FOR THE DESIGN OF GROUND-WATER MONITORING SYSTEMS IN KARST AND FRACTURED ROCK TERRANES

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ABSTRACT

Permeability in karst terranes and many fractured rock terranes is developed in joints, fractures, and bedding planes. In karst terranes some of these three discontinuities have been solutionally enlarged, sometimes to form conduit-like caves. In both terranes the assumptions underlying the description of ground-water flow as taking place in a homogeneous, isotropic, porous medium are usually invalid. As a result, ground-water flow and pollutant flow in most karst terranes and many fractured rock terranes is not described by the radially dispersive characteristics of flow in a granular medium. Monitoring systems designed with the assumption of dispersive flow will not produce accurate, reliable, or efficient monitoring of either ground-water flow or pollutant flow.

A Standard Guide, developed under the auspices of ASTM Subcommittee D18.21.09 on Special Problems of Monitoring Karst and Fractured Rock Terranes, has been deemed necessary to assist in the design and implementation of accurate and reliable monitoring systems in those hydrogeologic settings which depart significantly from the characteristics of a porous medium. This Guide is based on recognized methods of monitoring system design and implementation for the purpose of collecting representative ground water data. The design guidelines are applicable both to the detection of contaminant transport from existing facilities and to the assessment of proposed facilities. The recommended procedures of the Guide are designed to obtain representative aquifer and groundwater information. Use of the Guide will assist in the development of accurate conceptual hydrogeological models which are integral to the design of a time-sensitive, relevant, and reliable monitoring program.

INTRODUCTION

The American Society for Testing and Materials, better known as ASTM, was organized in 1898 and has grown into one of the largest voluntary standards development systems in the world. It is a non-profit organization that provides a forum for producers, users, and all others having a general interest in materials, products, systems, and services. Its 134 committees and additional subcommittees write standard test methods, specifications, practices, terminology, guides, and classifications for metals, paints, plastics, textiles, petroleum, construction, energy, the environment, consumer products, medical devices, computerized systems, electronics, and many other areas. More than 8500 ASTM standards are published each year in the 68 volumes of the Annual Book of ASTM Standards.

One of the 134 ASTM Committees, D18, is concerned with soil and rock. One of its subcommittees, .21, is concerned with groundwater monitoring, and it has been divided into 10 Task Groups, one of which, 21.09, is concerned with monitoring in karst and fractured rock terranes. This paper is written to tell what that task group is doing and why. But first, we wish to briefly establish that the problems of karst and fractured rock aquifers are different from those of other aquifers.

HOW KARST AND FRACTURED ROCK AQUIFERS PHYSICALLY DIFFER FROM OTHERS

The main physical difference between aquifers in karst and fractured rock terranes and aquifers in other terranes is the type of permeability (effective porosity). In aquifers consisting of unconsolidated material, and in many consolidated aquifers, the permeability is a primary feature of composition and deposition. In fractured and karst aquifers, however, the permeability is secondary, a result of fractures, bedding planes, and commonly dissolution along them. Underlying the description and analysis of ground water and contaminent flow is the implicit assumption that the aquifer is behaving as a porous medium. Even though it is widely recognized that ground-water flow in many rock aquifers is through secondary openings, the assumption that it is behaving as a porous medium is still made at some working scale of representation. The problem in many fractured rock aquifers, and most karst aquifers, is that this assumption is made uncritically.

HYDRAULIC AND HYDROLOGIC DIFFERENCES

A result of the porous media assumption is the further assumption that the aquifer is homogeneous and isotropic, and that flow is laminar. The range in aquifer characterstics within the spatial limits of the aquifer is frequently large in fractured rocks, and most karst aquifers are strictly non-homogeneous. Fractured rock and karst aquifers are typically highly anisotropic in three directions. Hydraulic conductivity and ground-water velocity can range over several orders of magnitude, depending upon the direction of ground-water movement. For example, in extreme cases in some karst aquifers, ground water appears to be moving parallel to the equipotential lines of a mapped potentiometric surface (Arnow, 1963). This is, of course, due to an inadequate amount of data points used to describe the surface. Nevertheless, until another means is utilized to check the ground-water flow direction, a potentiometric map can be misinterpreted to be a correct representation under the porous media assumption.

Flow in porous media is assumed to be laminar over the macroscopic scale of investigation. Although this assumption is reasonable for the average of hydraulic properties in a porous medium, water movement in fractures, especially solutionaly enlarged fractures, is often rapid and turbulent. This is due not only to size of the apertures in the rock, but also to the many direct inputs from the surface to the aquifer. This rapid and turbulent flow in both recharge and movement through the aquifer often results in large variations in head in response to precipitation, and large spatial and temporal variations in water chemistry. Monitoring wells installed in these terranes vary in the number and type of fractures encountered from well to well. Some wells encounter larger fractures with good connection to surface inputs or major conveyances; others encounter relatively "tight" and non-transmissive fractures. The result is potentiometric maps that can be difficult to interpret or are entirely misleading. For example, Figure 1 shows a potentiometric map for a site in an Indiana karst terrane during the February wet season. In fact, ground water from this site drains to the southwest, as demonstrated by tracer tests. Such a conclusion, however, could not be interpreted from this map. The tracer tests were needed to test and establish flow directions and destinations.

DIFFERENCES IN DESCRIPTION OF GROUND-WATER AND CONTAMINANT FLOW

Standard equations describing ground-water flow are based on Darcy's Law which assumes laminar flow in a porous medium, and is invalid in nonlaminar flow and other media. The equations are not valid for karst and fractured rock aquifers because flow in them is neither laminar nor in a porous medium. This means that the parameters for transmissivity and storativity which may be calculated from pumping tests are not representative of the aquifer as a whole. Consequently, predictions of contaminant movement based on assumed radial dispersion and advection, and calculated using these aquifer parameters, will be erroneous and misleading in both direction and magnitude. Monitoring systems designed on false assumptions will not produce accurate, reliable, efficient, or time-sensitive monitoring. Examples from case studies of the inadvisability of applying the porous-media assumption to the design of ground-water monitoring systems are given by: Aley (1988), Bradbury et al. (1990), Crawford (1988), Field (1988), McCann and Krothe (1991), Quinlan and Ewers (1985), and Quinlan

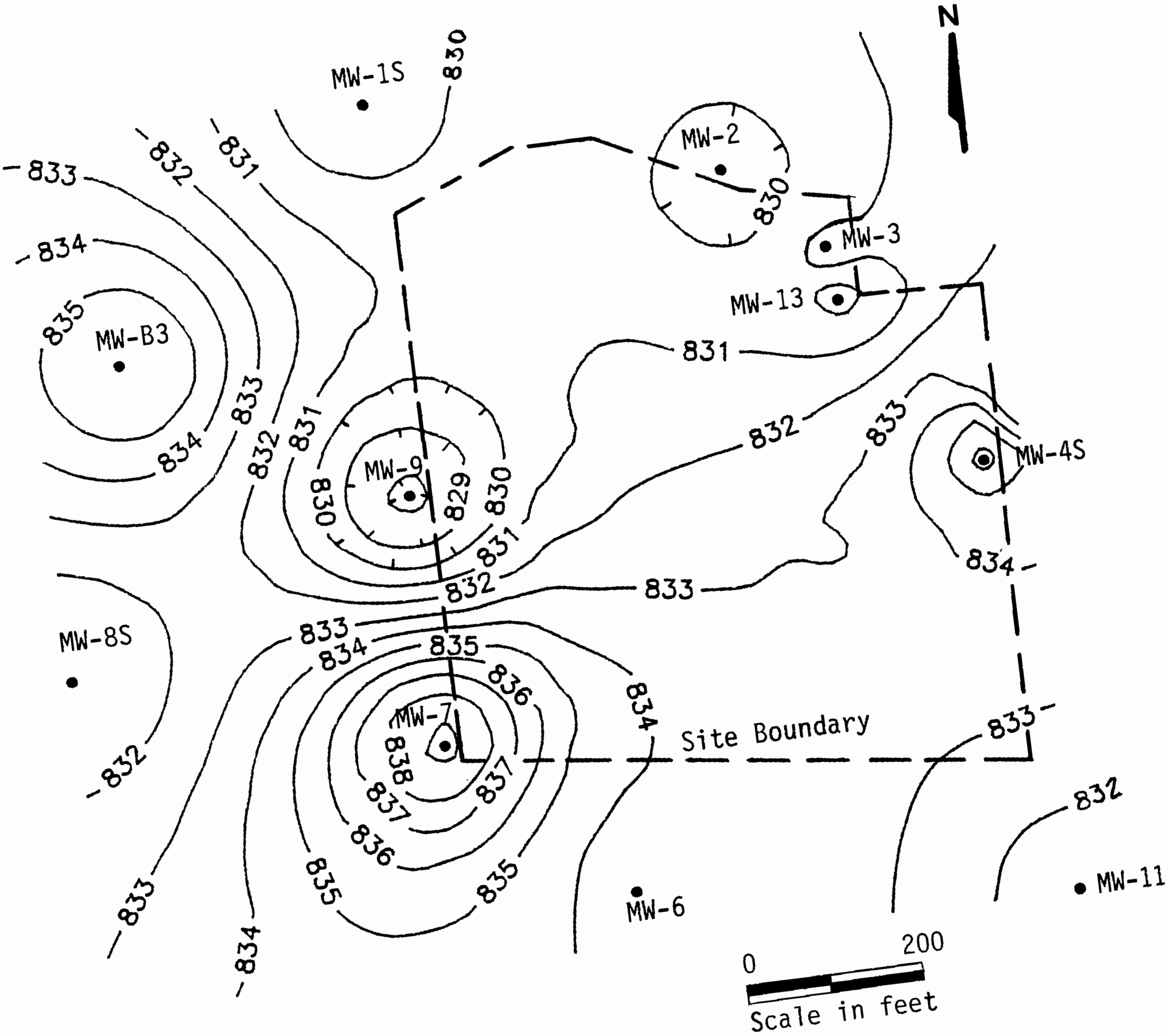

Figure 1 - Lemon Lane Landfill Potentiometric Map, February 1988
(after McCann and Krothe, 1991)

(1989). Table 1 gives a comparision of characteristics of porous media, fractured rock aquifers, and karst aquifers, and highlights their significant differences.

WHY A STANDARD GUIDE IS NECESSARY

Quinlan (1989), in the foreword of his document written for the EPA, "Ground-Water Monitoring in Karst Terranes: Recommended Protocols and Implicit Assumptions", stated four major reasons why that document was necessary. They are also four good reasons why an ASTM Standard Guide is necessary: 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. Although Quinlan (1989) focused specifically on karst terranes, ASTM Subcommittee D18.21.09 recognized that many fractured rock aquifers share similar problems in monitoring design and decided to include them so the Guide would have the widest practical application possible.

Almost 20% of the United States and 40% of the country east of Tulsa, Oklahoma, is underlain by carbonate rocks. The extreme vulnerabilty of carbonate aquifers in these terranes to contamination make it imperative that a reliable guide to monitoring be made available. Very few state agencies have guidelines that take into account aquifers that do not behave as porous media. However, some states such as Alabama, Indiana, Kentucky, Missouri, and Tennessee have begun to require water tracing studies to confirm ground-water flow directions in their karst terranes. Although some EPA documents do address the problem of monitoring in non-porous-media aquifers, most do not. In particular, the Technical Enforcement Guidance Document (TEGD) which governs monitoring at RCRA sites assumes a porous media exists in most applications of its monitoring guidance. Its one brief discussion of karst, p. 66-68, is ambiguous and probably erroneous; no data are given for the conclusions reached. Few universities, in their hydrogeology programs, specifically address how to characterize aquifers that do not behave as porous media.

A Standard Guide, developed through the extensive peer review and consensus process of ASTM, will provide a much needed and nationally disseminated standard on which to base design of monitoring systems in karst and fractured rock terranes. Such a Standard Guide does not yet exist. The thorough and complete ASTM

AQUIFER CHARACTER-ISTICS	AQUIFER TYPE		
	POROUS MEDIA	FRACTURED ROCK	KARST
Permeability	Mostly primary	Mostly secondary	Almost entirely secondary
Flow	Slow, laminar	Possibly fast and turbulent	Likely fast and turbulent
Isotropy	Most isotropic	Less isotropic	Highly anisotropic
Homogeneity	Most homogeneous	Less homogeneous	Non-homogeneous
Flow predictions	Darcy's law usually applies	Darcy's law may not apply	Darcy's law rarely applies
Storage	Within saturated zone	Within saturated zone	Both in saturated & un-saturated (epikarstic) zone
Head variation	Minimal variation	More variation	Can have extreme variation
Water chemistry variation	Minimal variation	More variation	Can have extreme variation

Table 1 - Comparison of porous, fractured, and karst aquifers (adapted from Bradbury et al., 1990)

process would best produce the necessary document which would have the widest practical application and acceptance. Such a document would be most useful to three groups of people: 1) Administrators who must evaluate existing or proposed networks for monitoring water quality in karst or similar terranes, but have minimal experience in non-porous-media hydrogeology; 2) Consultants and others who must design monitoring networks but may or may not have extensive experience in karst or similar terranes; and 3) The well-experienced water tracer who is already familiar with the hydrology and geomorphology of karst or similar terranes but has minimal familiarity with monitoring problems.

HOW A STANDARD GUIDE WILL ASSIST IN DEVELOPING A RELIABLE MONITORING SYSTEM

As to be explained within the Standard Guide, there are three main objectives to be accomplished in assisting the development of a reliable monitoring system: 1) The Guide will aid in the accurate characterization of karst- and fractured-rock aquifers because the development of a conceptual hydrogeological model that identifies and defines the various components of the flow system is a necessary first step to the design of a monitoring system; 2) The Guide will be based on recognized methods of monitoring system design and implementation for the purpose of collecting representative ground-water data. The design guidelines are applicable both to the determination of ground-water flow and contaminant transport from existing sites and assessment of proposed sites; 3) The objectives of the Guide are to recommend procedures for obtaining information on aquifer characteristics and representative water quality.

In order to accomplish these objectives, the Guide will discuss and compare qualitative differences in aquifer characteristics between porous, fractured, and karst media. The special characteristics of karst and fractured-rock aquifers will be pointed out, and explanations of why typical investigative procedures often fail to give accurate information will be provided. The Guide will recommend procedures to obtain representative aquifer and ground-water information. These recommended procedures include: 1) Criteria for determining whether or not an aquifer under investigation is behaving as a porous medium; 2) Procedures for potentiometric mapping; 3) Procedures for interpreting aquifer tests; 4) Criteria for determining when tracing is necessary; 5) Procedures for conducting tracer tests; 6) Criteria for determining monitoring station location; 7) Criteria for establishing sampling protocol; 8) Procedures for interpreting chemical and hydrologic data; and 9) Meeting regulatory requirements.

It is not the intent of the Guide to provide a rulebook on how monitoring must be conducted in non-porous-media aquifers. Rather, the members of ASTM Sub-committee D18.21.09 wish to provide the users of the Guide with the criteria and procedures to conduct an investigation so that they may determine the most accurate, reliable, efficient, and time-sensitive monitoring program possible.

People interested in assisting in formulating and reviewing the Guide described herein are invited to contact either of its Task Group co-chairman, the authors of this paper. Such participation involves attendance at meetings, review of documents, and voting on drafts proposed for adoption. Membership in subcommittees developing Standards or Standard Guides is open to all, but only members of ASTM can vote on them.

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DEVELOPMENT OF AN ASTM STANDARD GUIDE FOR THE DESIGN OF GROUND WATER MONITORING SYSTEMS IN KARST AND OTHER FRACTURED ROCK TERRANES

by Michael R. McCann and James F. Quinlan

It is appropriate to mention that funding of the entire ASTM ground water monitoring standards effort -- all 10 committees -- is being paid for by the U.S. Navy, the U.S. Environmental Protection Agency, and the U.S. Geological Survey.

We gladly acknowledge this support.

GROUND-WATER REMEDIATION MAY BE ACHIEVABLE IN SOME KARST AQUIFERS THAT ARE CONTAMINATED, BUT IT RANGES FROM UNLIKELY TO IMPOSSIBLE IN MOST: I. IMPLICATIONS OF LONG-TERM TRACER TESTS FOR UNIVERSAL FAILURE IN GOAL ATTAINMENT BY SCIENTISTS, CONSULTANTS, AND REGULATORS

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EXTENDED ABSTRACT

Not a single contaminated aquifer in the United States has been confirmed to have been successfully restored (remediated) by conventional "pump and treat" technology! At present, "No matter how much money the federal government is willing to spend, contaminated aquifers can not be restored to a condition compatible with health-based standards." (Travis & Doty, 1990). Numerous distinguished authorities agree (Feenstra and Cherry, 1988; Freeze and Cherry, 1989; Mackay and Cherry, 1989; Mackay, 1990; Rowe, 1991).

Many karst hydrogeologists, including the authors, have believed that aquifer remediation in karst terranes might be quicker and more easily achieved as a consequence of the rapid flow-through times and resultant short residence-times characteristic of most karst aquifers. This might be so. This might be especially so in highly integrated, maturely karsted aquifers typified by much of the Mammoth Cave area, much of western Kentucky and southern Indiana, and much of the midwest and southeast. We now, however, doubt the probability of quick and easy remediation and self-cleansing -- except where contaminants might enter the aquifer through swallets and sinkholes draining directly to underground streams and to springs. Recent multi-tracer investigations, begun in the epikarst zone of karst aquifers characterized

chiefly by diffuse flow, strongly suggest that the probability of remediation in most karst aquifers, diffuse- or conduit-flow, may range from unlikely to nil. (The tracer dyes were used as surrogate pollutants.)

Interpretation of judicious extrapolations of tracer-recovery data from a total of 9 randomly-located dye-injection wells at sites in the folded Appalachians of Pennsylvania and Tennessee -- as contrasted with discrete inputs at sinking streams or cave streams -- data from more than a thousand water samples and hundreds of activated-charcoal-detectors, all regularly collected over periods ranging from 6 to 20 months, strongly indicates that dye-residency times in the sub-water-table part of many karst aquifers can range from several years to many tens of years (Quinlan, 1992; Quinlan and Ray, 1991; Quinlan, et al., 1990a, 1990b, 1991).

Application of "pump and treat" technology at an Appalachian superfund site in Beekmantown (Knox) dolomites has prevented some migration of dye from it, but has not diminished the long-term concentration of dye in the aquifer. Similar long-term residency of dye (and, therefore, of pollutants) in parts of epikarst aquifers propinquitous to those parts with rapid flow-through and short-term residency has been recognized by others also (Smart and Friedrich, 1986; Alexander et al., 1991).

An assumption that rapid remediation might be characteristic of karst and epikarst aquifers in, for example, the Mammoth Cave area, ignores the following facts: 1) More than 99% of the approximately 600 dye-tests performed there have been from sinking streams and cave streams rather than from wells randomly drilled into the diffuse-flow part of the aquifer (between conduits and their tributaries), and 2) No tracer tests have been performed in the epikarst there. The time necessary for possible remediation of an aquifer in the Mammoth Cave area would be highly variable and a function of where pollutant input occurs above the aquifers. Remediation there could be practically impossible.

These conclusions are applicable to most karst areas and are relevant to interpretation of traces performed as part of spill response in most karst terranes.

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Ground-water remediation may be achievable in some karst aquifers that are contaminated, but it ranges from unlikely to impossible in most: I. Implications of long-term tracer tests for universal failure in goal attainment by scientists, consultants, and regulators

By: James F. Quinlan and Joseph A. Ray

Q. **Have you seriously questioned how many of your fluorescein recoveries are your dye and how many are from other sources? Also, there can occur what some believe to be "breakdown products" of Rhodamine WT that have special characteristics similar to those of fluorescein. We have seen this in long-term tests in Missouri. Also, we sometimes find a long-term background for fluorescein.**

A. Yes. Paranoia is a healthy state of mind. For procedural reasons too lengthy to summarize here, we are satisfied that no tests we have run have had recovery of fluorescein from the tests of others, from industrial or household background, or from Rhodamine WT breakdown products. In brief, however, the possibility of such false positives can be minimized by repeatedly monitoring for them in background before the tests and by sampling often enough so that one can closely and reliably monitor the breakthrough, peaking, and recession of all dyes used, and by recognizing that landfills and gasoline stations can be a source of fluorescein (from the colorant in antifreeze).

The breakdown of Rhodamine WT to something that might be confused with fluorescein was first reported by Jim Duley in the proceedings of the first NWWA karst meeting, in 1986 (p. 396-397), but we are unaware of any further study of such compounds. Dave Sabatini has studied the properties of Rhodamine WT (Ground Water, 1991. 29(2):341-349) and shown that it actually consists of two structural isomers with different sorption tendencies (EOS, 1991, 72(44):154). These isomers explain why Rhodamine WT breakthrough curves commonly have a plateau at a C/C_0 value of about 0.4.

As for the long-term fluorescein background in Missouri, all one can do is repeatedly monitor with a scanning spectrofluorophotometer, not trace indiscriminately, and (seemingly contradictorily) use enough dye to unambiguously exceed background and the known fluctuations of it. The problem of accumulation of fluorescein background within a karst aquifer has also been recognized in Switzerland and is the subject of a recent paper by Aurèle Parriaux and two colleagues (Hydrogéologie, 1990. (3):183-194).

Q. **When will regulatory agencies recognize the futility of complete remediation in Appalachian karsts? How do we make them realize this? They insist on pumping and treating for up to 10 to 20 years. Costs skyrocket and frustration rules.**

A. We wish we knew. We are fully sympathetic to the physical impossibility of remediation of some aquifers and agree with the authors we cite. Nevertheless, regulatory agencies are required to develop, implement, and enforce regulations as enabled by state or federal legislation. Many states have a non-degradation policy or legislation that requires remediation to background or detection limits. Such legislation is created by politicians, many of which do not necessarily understand the complex and technical issues involving hazardous waste remediation problems. They do understand that the voting public has zero tolerance for any level of contamination -- irrespective of risks, potential exposure, or cost of remediation. Sometimes, the fault is with well-meaning regulators, legislators, and legislative staff who, for various reasons, have written rules or statutes more stringent than is necessary or intended. We do not mean to seem naive, but we see the first steps in improvement of communication between regulatory agencies and both the regulated and their consultants are to be education of all parties as to the nature of reality, establishment of mutual respect for the technical competence and the integrity of both consultants and regulators, and creation of a non-adversarial climate. If mutual good faith is established, and if there are alternative interpretations of the regulations, the more reasonable one can be sanctioned. These desirable goals are more easily stated than obtained.

Session VIII:

Emergency Response and Ground-Water Management

The Use of Groundwater-Level Measurements and Dye Tracing to Determine the Route of Groundwater Flow from a Hazardous Waste Site in an Area of Karst in Hardin County, Kentucky

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ABSTRACT

The nature of karst groundwater flow dictates that groundwater investigations in karst areas be approached in a different manner than those in more traditional hydrogeologic settings. Water-level measurements over relatively large areas and dye tracing have proven to be suitable techniques for determining characteristics of groundwater flow in karst areas. This paper presents a case study of a two-phase investigation consisting of, first, water-level measurement and identification of hydrogeologic features and, second, dye tracing to determine the groundwater flow route from the Middleton Hazardous Waste Site in Hardin County, Kentucky.

Groundwater measurements were obtained over an area of approximately 70 square miles. From these measurements a water-table contour map was constructed. Based on the slope of the water-table gradient and the form of the water-table contours, a hypothesis was formed: groundwater flowing from the Middleton Hazardous Waste Site would resurge at a spring 4.5 miles to the west. This hypothesis was tested by performing dye traces.

Passive dye receptors were placed at all groundwater resurgences, interface locations and, where possible, cave streams in the study area. An automatic water sampler was placed at the hypothesized resurgence. Dyes were injected into a sinkhole collapse adjacent to the site and at a nearby sinking stream. After allowing sufficient time for the dye to flow through the system, the passive dye receptors and water samples were analyzed for presence of the dyes. The hypothesis was proven correct by the dye traces.

Based on interpretation of the water-table contour map and results of the dye traces (both qualitative and quantitative) the direction, destination, velocity, and general route of groundwater flow from the Middleton Hazardous Waste Site were determined.

INTRODUCTION

Groundwater investigations typically involve the use of monitor wells to measure the movement of groundwater and/or contaminants (Fetter, 1988) (Driscoll, 1986). In karst aquifers the majority of groundwater flow is turbulent (White, 1988) compared to flow through porous-media aquifers which is generally laminar. Relatively rapid transmittance of groundwater through discrete conduits in karst aquifers precludes the use of monitor wells as a means of accurately measuring the movement of groundwater and/or contaminants. In karst settings water and/or contaminants are typically on the ground surface for only a short time before sinking into the subsurface through features such as sinkholes, sinking streams, and soil macropores. These features provide direct access for liquids and solids to migrate from the surface into the subsurface with little or no filtration by soils. Once in the subsurface, the turbulently flowing groundwater can rapidly transport liquids and solids considerable distances. Most water wells and springs in karst areas become muddy after hard rains because solid material transported by surface water flows from the ground surface into the aquifer through conduits large enough to transmit the particles. The subsurface conduit system will then transport solids (sediments) much like surface flow systems, the more turbulent the water, the greater its capacity to transport solid material. Due to the unique characteristics of groundwater flow in karst areas, investigations should be designed to encompass a larger geographical area of study than the site itself and incorporate water-table measurement data to identify general flow directions and dye tracing to determine general flow routes.

This paper presents a case study of a two-phase investigation designed to determine characteristics of groundwater flow in the vicinity of a hazardous waste site in a karst area. The initial phase of this investigation involved the identification of all applicable hydrogeologic features in the study area and the preparation of a water-table contour map based on water-level measurements from private water wells and elevations of surface and subsurface streams. From the information acquired through these activities, the second phase of the investigation, dye tracing, was designed and performed. The results of both phases of the investigation determined the direction, destination, velocity and general route of groundwater flow from the hazardous waste site.

SITE BACKGROUND

Several hundred electric transformers were burned over a period of years at a rural "junkyard" located 10 miles south of Elizabethtown in Hardin County, Kentucky (Figure 1). The "junkyard" is referred to as the Middleton Hazardous Waste Site (MHWS). The transformers were burned in order to facilitate the extraction of copper wiring housed inside the transformers. As a result of the activities at the site, oil laden with polychlorinated biphenyls (PCBs) and associated metals were released into the soil. PCBs have a strong affinity for organic materials in soils and therefore constitute a threat to groundwater in karst areas due to the direct surface-subsurface link provided by karst features. The uppermost aquifer in the vicinity of the MHWS is a karst aquifer. At the time of this investigation the aquifer served as a water supply for the majority of residences in the area. Due to the potential impact of the "junkyard" activities on the quality of the area's groundwater, a karst groundwater investigation was performed (Crawford et al., 1989) as a component to the overall site remediation activities of the Region IV U.S. EPA.

Physical Setting

Hardin County, Kentucky is located in the Mississippian Plateaus Physiographic Province. The area of study encompasses approximately 70 square miles of mature karst topography which is bounded on the east, north and west by the Nolin River as shown in Figure 1. This river serves as the base-level stream for groundwater flow in the area. Another surface stream, Dorsey Run, flows toward the west and sinks 2 miles south of the MHWS.

The geology of the study area is comprised of two slightly westward-dipping (<1 degree) limestone formations of Mississippian Age. The oldest unit is the St. Louis Limestone, which is bedrock in the eastern

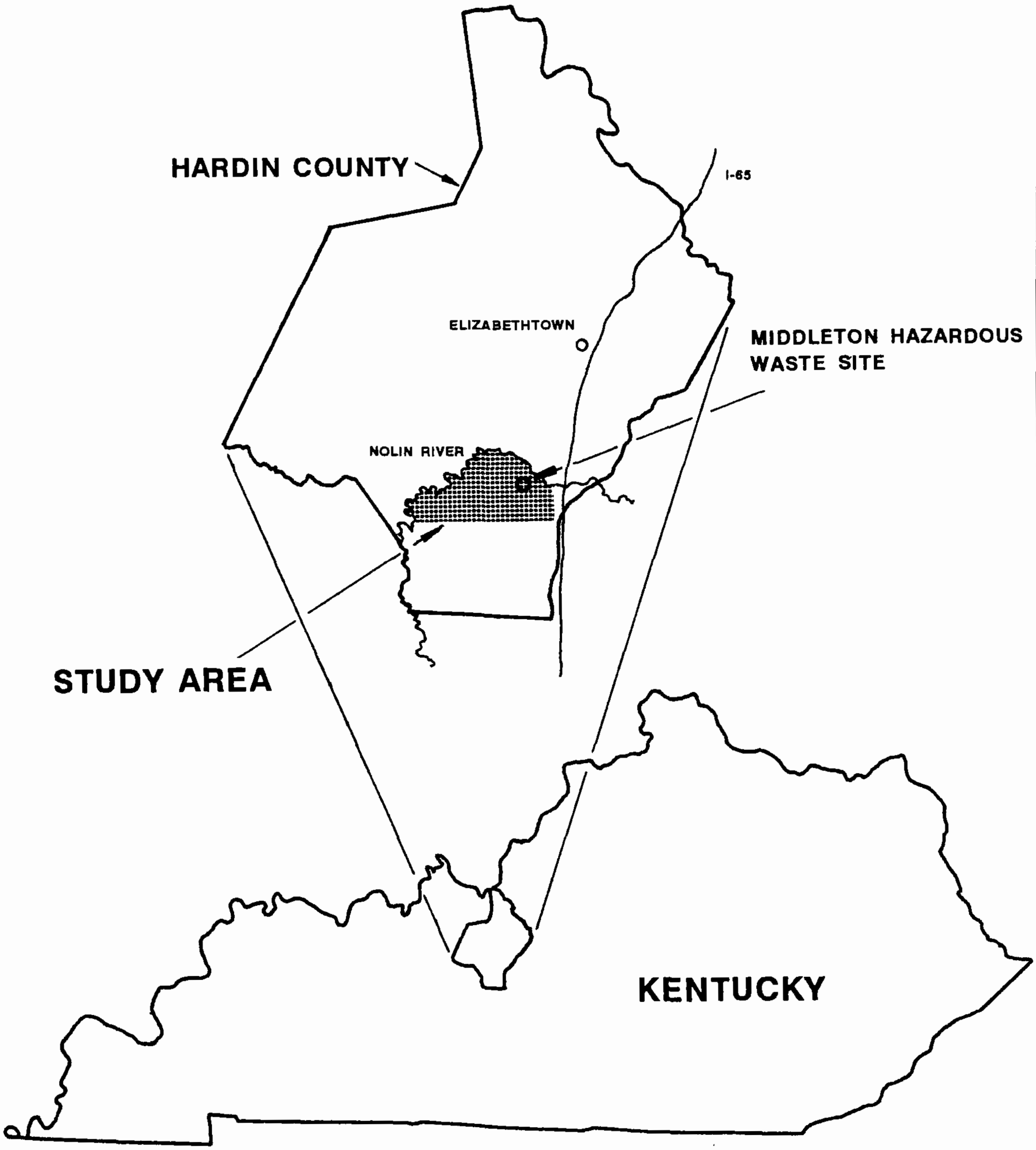


FIGURE 1 - Location map for the Middleton Hazardous Waste Site Groundwater Investigation

two-thirds of the study area. Bedrock in the western one-third of the study area is the younger Ste. Genevieve Limestone (Moore, 1964, 1965). The study area exhibits features common to areas underlain by relatively homogeneous carbonate rock.

METHODOLOGY

The groundwater investigation consisted of two phases. The initial phase involved determining the water-table elevation in as many water wells as possible in the area and collecting information concerning locations of applicable hydrogeologic features from landowners, area residents, and members of the Fort Knox Grotto of the National Speleological Society. The second phase involved dye tracing from a sinkhole adjacent to the MHWS and the sinking stream located south of the MHWS.

Phase One - Identification of Hydrogeologic Features and Determination of the Water-Table Gradient

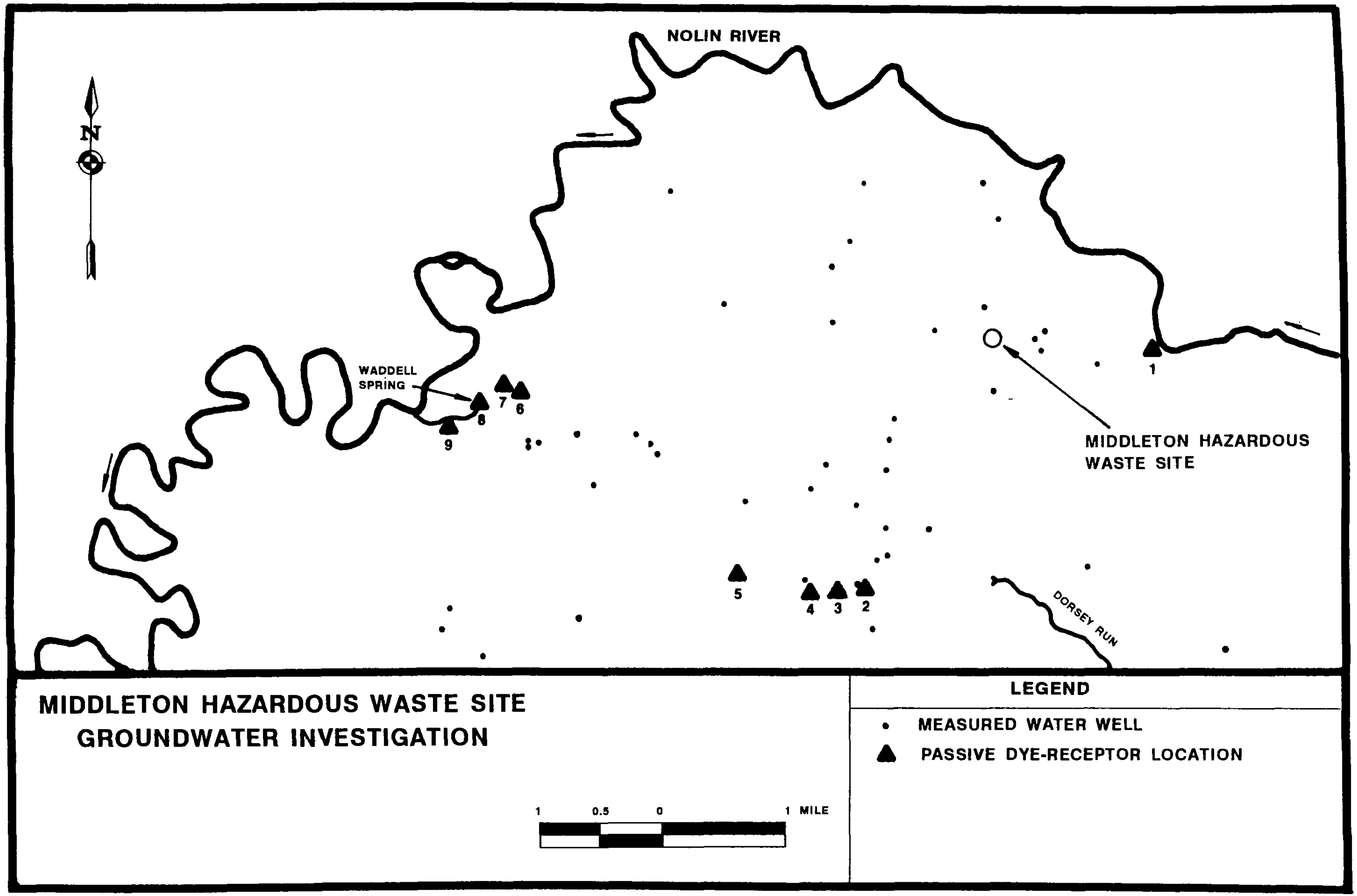
A door-to-door survey was conducted to identify residences with accessible water wells and to acquire permission to obtain water-level measurements. Measurements were made with water-level indicators. Only data representing the water-table aquifer were collected. Information from well owners indicated the majority of the wells were cased only to bedrock. The wells were uncased from bedrock to the total depth of the well. Due to the size of the study area and the distribution of wells, it was impractical to determine the precise elevation of the wells. Surface elevations were estimated from the Sonora and Summit 7.5 minute topographic quadrangle maps (USGS, 1967 and 1972). The contour interval of the topographic quadrangles in the vicinity of the site was 20 feet. Estimates of surface elevations from these large contour interval maps decreased the accuracy of the water-table elevations in the area but did not preclude the use of the technique in providing a generalized map of the water table.

During the acquisition of water-level measurements from wells, inquiries were made of landowners and area residents concerning locations of various hydrogeologic features such as springs, surface streams, sinking streams, sinkholes with water flow, and caves, in particular those caves with streams. These features, as well as those identified by local cavers, were investigated and located on topographic maps. In anticipation of the second phase of the investigation, passive dye receptors were placed at the aforementioned hydrogeologic features. These receptors (strips of untreated surgical cotton and packets of activated coconut charcoal) once analyzed, served as an indicator of the presence or absence of fluorescent dyes in the groundwater flow system. Placement of receptors at the identified hydrogeologic features throughout the study area prior to dye tracing provided a measure of background fluorescence to which the post dye-trace receptors would be compared. Before initiating the dye traces, all background dye receptors were removed and replaced with fresh receptors. Locations of measured water wells and passive dye receptors are identified in Figure 2.

The information acquired from water-level measurements and the estimated elevations of surface and cave streams were used to prepare a water-table contour map (Figure 3). Interpretation of the water-table contour map influenced the second phase of the investigation. Due to its proximity to the MHWS, a spring located 1.3 miles east of the site (Location 1) was initially hypothesized as the primary resurgence of water from the site. However, the slope of the water-table gradient and the form of the water-table contours suggested the most likely resurgence was at Waddell Spring, 4.5 miles to the west (Location 8). Anticipating the resurgence of dye at this spring, an ISCO automatic water sampler was installed and programmed to collect a sample every four hours. Collection and analysis of these samples provided information necessary to make a quantitative interpretation of groundwater flow from the MHWS.

Phase Two - Dye Tracing

Six pounds of fluorescein dye (C.I. Acid Yellow 73) was injected into a collapsed sinkhole approximately 200 feet east of the site (Figure 4). Fifteen-hundred gallons of water was injected to insure the drainage capacity of the collapse while at the same time wetting the soil to reduce dye sorption. Nine thousand-five hundred gallons of water was then used to flush the dye into the system.


NOLIN RIVER
N
WADDELL SPRING
1
2
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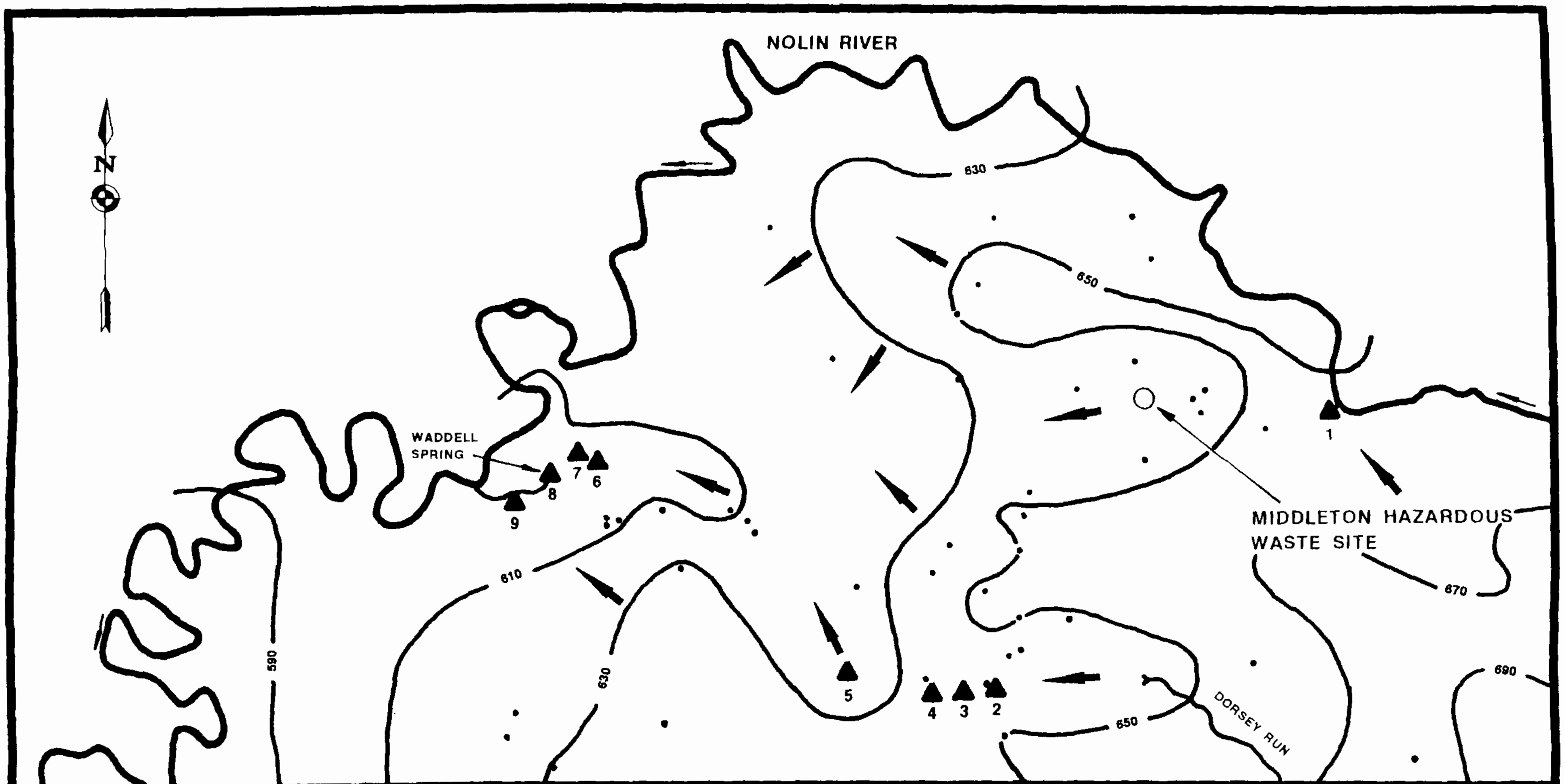
MIDDLETON HAZARDOUS WASTE SITE
DORSEY RUN
MIDDLETON HAZARDOUS WASTE SITE
GROUNDWATER INVESTIGATION
1 0.5 0 1 MILE
LEGEND
MEASURED WATER WELL
PASSIVE DYE-RECEPTOR LOCATION

Figure 2



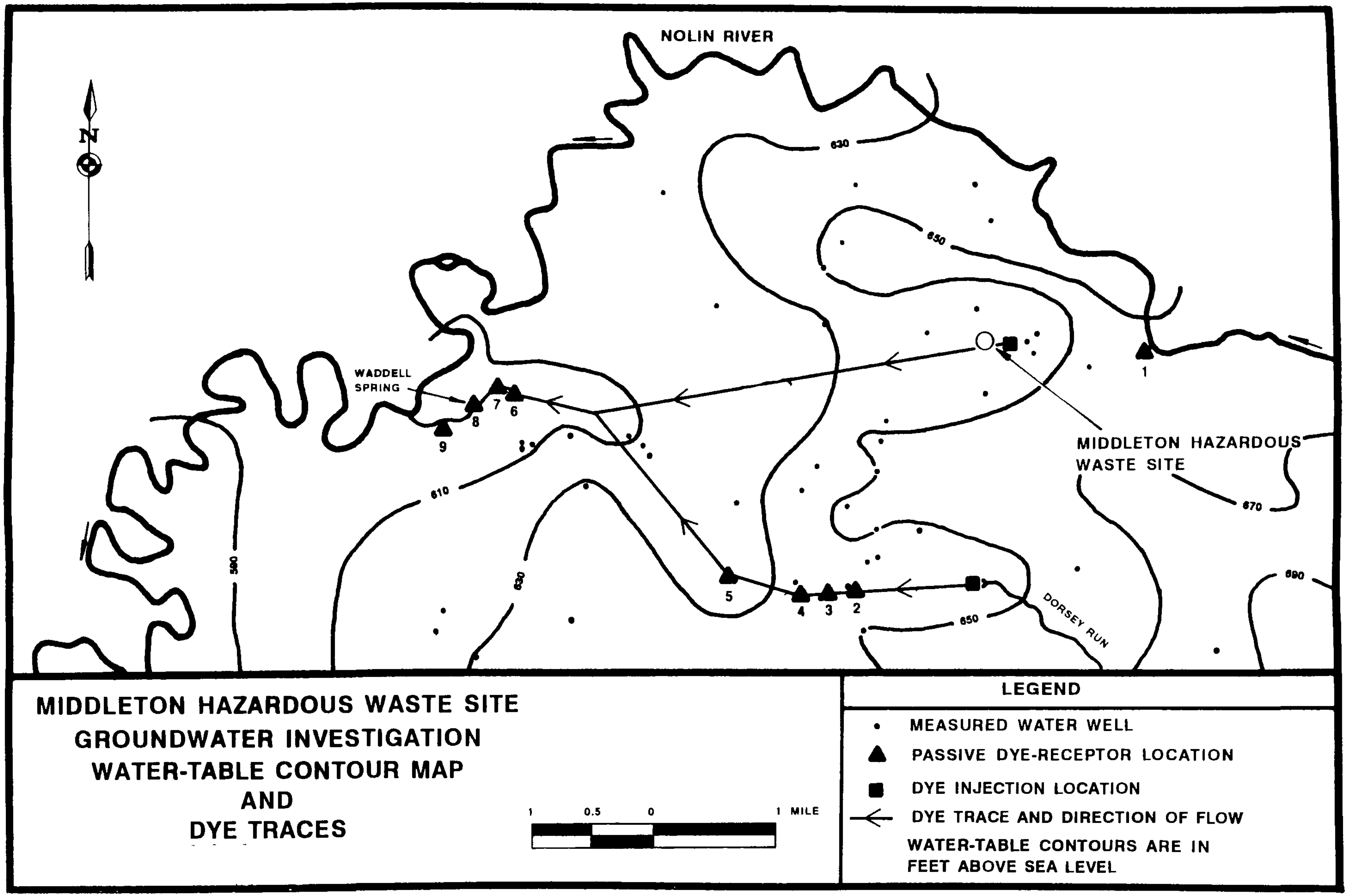
MIDDLETON HAZARDOUS WASTE SITE
GROUNDWATER INVESTIGATION
WATER-TABLE CONTOUR MAP

LEGEND

- • MEASURED WATER WELL
- ▲ PASSIVE DYE-RECEPTOR LOCATION
- ← GENERAL GROUNDWATER FLOW DIRECTION

WATER-TABLE CONTOURS ARE IN FEET ABOVE SEA LEVEL

Figure 3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MIDDLETON HAZARDOUS
WASTE SITE
DORSEY RUN
MIDDLETON HAZARDOUS WASTE SITE
GROUNDWATER INVESTIGATION
WATER-TABLE CONTOUR MAP
AND
DYE TRACES
1 0.5 0 1 MILE
LEGEND
MEASURED WATER WELL
PASSIVE DYE-RECEPTOR LOCATION
DYE INJECTION LOCATION
DYE TRACE AND DIRECTION OF FLOW
WATER-TABLE CONTOURS ARE IN
FEET ABOVE SEA LEVEL

Figure 4

To further characterize the groundwater flow of the area, 15 pounds of optical brightener (Tinopal 5BM GX, F.B.A. 22) were injected into Dorsey Run Swallet (Figure 4). This sinking stream provided an ideal location to inject dye, and receptors capable of capturing the optical brightener had been placed along with those being used to intercept fluorescein dye.

Every four days water samples being taken by the automatic sampler were removed and fluorometrically analyzed in the laboratory for presence of fluorescein dye. After allowing sufficient time for the dyes to flow through the system, all passive dye receptors were replaced with fresh receptors. The dye receptors removed from the locations were analyzed for the presence of both fluorescein and optical brightener. Analysis for the presence of fluorescein involved rinsing the coconut charcoal with high-pressure water to remove sediment and elutriating the charcoal with a solution of 9.5:1 isopropyl alcohol and potassium hydroxide. Both visual and fluorometric analysis of the elutriant and comparison to the background receptor elutriant indicated which hydrogeologic features the dye flowed through. Analysis for optical brightener involved observing the spray-cleaned strips of untreated cotton under a long-wave ultraviolet light. Receptors which fluoresced bright blue-white, as compared to background receptors from the same locations, were identified as locations which were in the flow path of the optical brightener introduced into the system.

RESULTS

During the inventory of the hydrogeologic features a number of springs, cave streams, and karst windows were located (Figure 2, Locations 1-9). Dye receptors were placed at each of these features. Because the Nolin River is the base-level stream in the area, it was unnecessary to include hydrogeologic features north of the river in the investigation. Passive dye receptors were positioned at several features south of the area depicted in Figure 2. However, these locations did not prove to be a part of the groundwater flow system in the vicinity of the MHWS and, therefore, are not indicated in this figure.

Groundwater elevations obtained by measuring water wells were combined with estimated elevations of the surface and cave streams to produce a water-table contour map (Figure 3). Data from forty-three wells were used in preparation of the water-table contour map. From this map it is apparent that groundwater in the vicinity of the MHWS flows west, contrary to the original hypothesis of flow to the east. Based on the relative size of Waddell Spring and the direction of groundwater flow, it was selected as the most probable location for the resurgence of groundwater from the site. The water-table contour map indicates that water sinking at Dorsey Run Swallet is part of the groundwater flow found at Locations 2, 3, 4, and 5. This water also flows in a westerly direction toward Waddell Spring. The flow of water from the spring at Location 1 originates south and southeast of the spring rather than from the west as originally hypothesized.

Qualitative Dye Trace Results

Dye receptors were collected and analyzed for the presence of fluorescein and optical brightener. The results of the dye traces and the water-table contours are shown in Figure 4. Activated charcoal dye receptors positioned at Locations 6, 7, and 8 were positive for the presence of fluorescein. All other receptors were negative for the presence of fluorescein. Surgical cotton dye receptors positioned at Locations 2, 3, 4, 5, 6, 7, and 8 tested positive for the presence of optical brightener, and all other receptors tested negative.

Quantitative Dye Trace Results

Water samples collected at Waddell Spring by the automatic sampler were fluorometrically analyzed for fluorescein. Results of this analysis are illustrated in Figure 5. The first arrival of dye at Waddell Spring occurred approximately 86 hours after injection. The concentration of dye at the spring peaked approximately 12 hours after first appearance and was no longer detectable 80 hours later.

FLOW-THROUGH CURVE FOR FLUORESCEIN DYE AT WADDELL SPRING

Dye Injected on September 7, 1989 at the MHWS

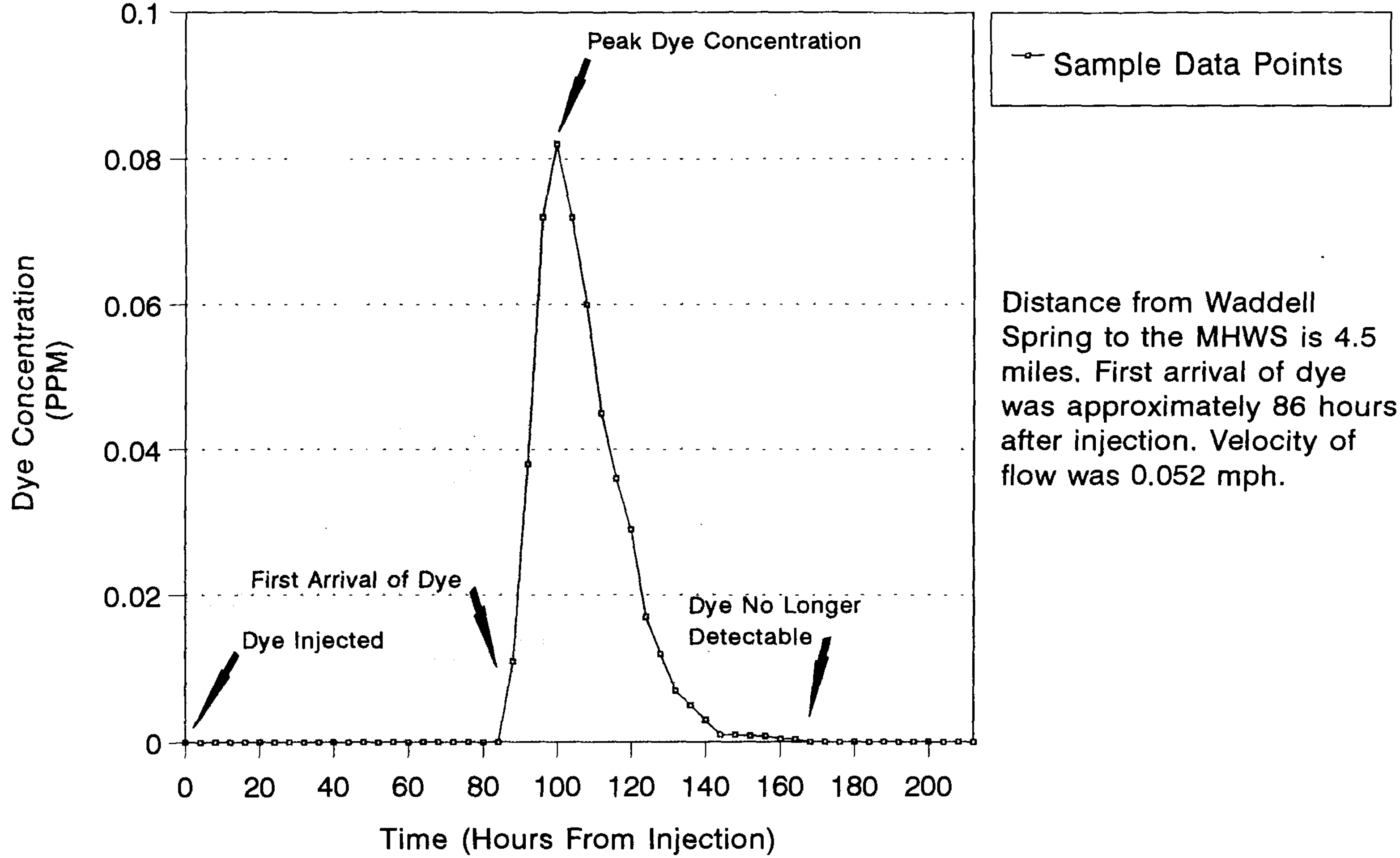

Figure 5 - Dye Flow-Through Curve

DISCUSSION OF RESULTS

Groundwater from the MHWS flows toward the west through karst windows at Locations 6 and 7 before resurging at Waddell Spring (Figure 4). Dorsey Run, which sinks 2 miles south of the site, flows through Locations 2, 3, 4, and 5 before joining the flow from the MHWS. The convergence of these two cave streams occurs between Locations 5 and 6. The rate of flow from the MHWS to Waddell Spring is approximately 0.052 miles per hour. This flow rate is based on linear distances between positive fluorescein detection points. The actual flow rate is higher given the tendency for cave streams to meander in a manner similar to surface streams.

CONCLUSIONS

A two-phase approach to the Middleton Hazardous Waste Site groundwater investigation determined the direction, destination, velocity, and general route of groundwater flow from, and in the vicinity of the site. The use of water-table measurement data produced a generalized water-table map of the study area and was beneficial in directing the second phase of the investigation. Dye traces identified the destination and general flow routes of groundwater from the site and in the vicinity of the site.

Error introduced to the water-table measurements by estimating surface elevations from 20-foot contour interval topographic maps would not typically be acceptable in hydrogeologic investigations. However, in this investigation the generalized water-table map proved useful in identifying the most probable resurgence of groundwater from the site. Based on this information, a water sampler was placed at that resurgence. Information gained from the fluorometric analysis of the samples was used to prepare a dye flow-through curve. The added certainty of the dye flow-through curve and the associated information it provides were necessary given the nature of the investigation. A dye flow-through curve more positively identifies the connection between injection point and sampling point than visual or fluorometric analysis of elutriant from passive dye receptors. From the dye flow-through curve it was possible to estimate the flow rate from the site to the spring. The water-table contour map and dye-trace information obtained from this study will also serve to focus any groundwater sampling activities associated with the site investigation.

If the water-table map had not been prepared prior to dye tracing, the water sampler would have been incorrectly located at the spring east of the site. Analysis of samples collected at that spring would not have identified the resurgence of groundwater from the site. Since passive dye receptors had been placed at all hydrogeologic features in the area, Waddell Spring would have been identified as the resurgence of groundwater from the site. However, it would have been necessary to perform another dye trace with the water sampler housed at Waddell Spring to provide information for a quantitative interpretation of groundwater flow from the site. An additional dye trace would have required more time and resources, and an accurate interpretation of the second dye trace may have been hindered by dye from the first trace remaining in the system.

This case study reinforces the need for hydrogeological investigations at hazardous waste sites in karst terrains to include regional groundwater data to determine general flow directions and dye tracing to identify the destination and general route of groundwater flow. The more traditional approach at hazardous waste sites using site monitor well data may result in a completely false characterization of groundwater flow.

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Dr. Nicholas Crawford

Nicholas C. Crawford, Ph.D. is a Professor in the Department of Geography and Geology and Director of the Center for Cave and Karst Studies at Western Kentucky University. He has written over 140 articles and technical reports dealing primarily with groundwater contamination of carbonate aquifers. The recipient of 25 grants for hydrologic research on environmental problems of karst regions, he was awarded Western's highest award for Outstanding Achievement in Research in 1985. As a consultant specializing in carbonate aquifers for the past fifteen years, Dr. Crawford has performed over 600 dye traces and worked on numerous groundwater contamination problems for private firms and for federal, state and local government agencies.

Mark J. Rigatti

Mark J. Rigatti is the program manager for the U.S. EPA Region IV Emergency Response Cleanup Services (ERCS) contract for O.H. Remediation Services Corporation. Mr. Rigatti holds a B.S. in aquatic biology and has been associated with several studies dealing with groundwater contamination in carbonate aquifers in Kentucky and Tennessee as a component to hazardous waste site remediation activities.

The Use of Groundwater-level Measurements and Dye Tracing to Determine the Route of Groundwater Flow from a Hazardous Waste Site in an Area of Karst in Hardin County, Kentucky

C. Britton Dotson, Nicholas C. Crawford, Mark J. Rigatti

Question 1 Was there any indication of site contaminants in the local wells or springs?

Question 2 - What assurances do you have that dye-tracing would intersect the same conduit system with the sinks at the waste site?

Answer 1 The primary objective of Crawford and Associates was the identification of flow direction and destination of groundwater in the vicinity of the site. Crawford and Associates was not involved in any sampling activities. To date there has been no extensive sampling of water wells and springs in the study area. A domestic water well located on the site property was sampled during the initial EPA response. There were no site contaminants detected in this well.

Answer 2 - The uppermost aquifer in the vicinity of the hazardous waste site is a karst aquifer. Sinkholes on the site and in the vicinity drain into this aquifer. Since individual groundwater drainage basins exist within the aquifer, as evidenced by multiple groundwater resurgences, all sinkholes in the study area do not contribute to the same drainage basin. It is possible that water flowing into one sinkhole may contribute to a different drainage basin than water flowing into another sinkhole only 200 feet away. Based on the hydrogeologic inventory only two springs (Locations 1 and 8) were likely candidates for resurgence of dye injected at the site or in the vicinity. Dye that was injected into the sinkhole 200 feet east of the site resurged 4.5 miles west of the site at Location 8 (Waddell Spring). This fact, accompanied by information provided by the water-table contour map, indicated that the sinkhole where dye was injected contributed to the same drainage basin as the sinkholes on site. If dye had been injected into a sinkhole on the site it is unlikely that it would have flowed east to Location 1. To do this, flow would have been against the gradient, as indicated by the water-table contour map, and would be opposite the general flow direction identified by the dye trace.

RECOMMENDED ADMINISTRATIVE/REGULATORY DEFINITION OF KARST AQUIFER, PRINCIPLES FOR CLASSIFICATION OF CARBONATE AQUIFERS, PRACTICAL EVALUATION OF VULNERABILITY OF KARST AQUIFERS, AND DETERMINATION OF OPTIMUM SAMPLING FREQUENCY AT SPRINGS

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SUMMARY

A major problem of karst hydrology is an inadequate understanding by many regulators, planners, attorneys, geologists, hydrologists, engineers, and other non-karst specialists of what a karst aquifer is and how it differs significantly from other aquifers. As a first step in solving this problem, we offer the following definition. **A karst aquifer is an aquifer in which flow of water is or can be appreciable through one or more of the follow-**

ing: joints, faults, bedding planes, and cavities -- any or all of which have been enlarged by dissolution of bedrock.

The above definition is in terms of hydraulics, not landforms (such as sinkholes), grikes (soil-filled joints), or hydrologic features (such as swallets, sinking streams, springs, and caves). The presence of sinkholes and any or all of these other features is diagnostic of a subjacent karst aquifer, but the absence (or unrecognized presence) of any or all of them does not mean that a karst aquifer is not present. As a generalization, however, if carbonate rocks such as limestone, marble, or dolomite, or even more soluble rocks such as salt or gypsum, are present, assume that the terrane is a karst -- until or unless the opposite can be convincingly proven. Almost all carbonate terranes are underlain by one or more karst aquifers, but some terranes and aquifers are more karstic than others.

Having defined karst aquifers, we classify them according to three fundamental, independent attributes governing their behavior -- recharge, storage, and flow -- each of which can be visualized as forming orthogonal edges of a cube. Each of the three attributes is part of a continuum between two end-members, yielding a classification based on a total of 6 karst end-members, many intermediate members, and 2 non-karst end-members. The cube, modified after one proposed by Smart and Hobbs (1986), can be visualized as a 3-dimensional conceptual model within the boundaries of which various aquifers can be plotted (mapped) and their relations to one another can be seen, as shown in Figure 1.

All of the many types of karst aquifers can be grouped into a fourth continuum based on the sensitivity of their response to variations in recharge, storage, and flow, and their attendant relative vulnerability to the adverse effects of spills and improper disposal of waste: hypersensitive aquifers, very sensitive aquifers, moderately sensitive aquifers, and slightly sensitive non-karst aquifers. These new terms are plotted on the same cube.

Hypersensitive aquifers are characterized by concentrated recharge at discrete points, low to moderate storage, and conduit flow. Consequently, they are extremely vulnerable to groundwater contamination. Very sensitive aquifers are characterized by the occurrence of either conduit flow, point recharge, or low storage, or a combination of two of these, as shown in Figure 1. Again, they are sensitive to pollution, but less so than hypersensitive aquifers. Moderately sensitive aquifers have dispersed, slow recharge, high storage, and diffuse flow, as also shown in Figure 1; such aquifers are therefore less vulnerable to rapid transmission of pollutants than are very sensitive karst aquifers.

The continuum from hypersensitive to moderately sensitive karst aquifers can be discussed in terms of two end-members which have traditionally, but misleadingly, been called "conduit-flow aquifers" and "diffuse-flow aquifers". Recognition of the position of an individual aquifer in this continuum is essential to providing adequate aquifer protection and appropriate management of waste disposal and also to the design of reliable groundwater monitoring schemes. In general, movement of pollutants in hypersensitive karst aquifers is very rapid, commonly tens to thousands of feet per hour, depending upon stage. Reaction time, the time between the start of a recharge event (or when a spill occurs) and when it is first sensed by the aquifer, is short. Duration of the adverse effects of a pollution incident may be relatively brief (depending upon the properties of the contaminant and how and where in the system the contamination occurs). In contrast, movement of pollutants in moderately sensitive aquifers is usually several orders of magnitude less rapid than in hypersensitive aquifers, but it is usually orders of magnitude faster than in most granular aquifers. Reaction time is much longer than in hypersensitive aquifers. The results of a pollution incident can persist for tens of years.

The terms conduit flow and diffuse flow, as descriptors of aquifer type used for classification, should be abandoned because they prevent necessary consideration of the important roles of recharge and storage. Conduit flow and diffuse flow should be retained, however, as descriptors of flow type.

The distinctions between the two vulnerability end-members in karst aquifers can be made in several ways; they are most easily determined by interpreting variations in specific conductance (hereafter called conductivity), as characterized by its coefficient of variation (CV). The variations in conductivity are more important than the absolute values. The distinctions between aquifer vulnerability can also be determined from fluctuations in one or more of the following alternative water quantity or quality parameters: discharge, turbidity, temperature, and carbonate hardness. These distinctions can only be determined by site-specific fieldwork. The necessary data for these conductivity and alternative vulnerability-type discrimination measurements are easily and inexpensively obtained, but they are not included in geologic or hydrologic publications or reports.

The spring is the pulse of a karst aquifer. Monitor it in order to understand the nature of an aquifer. While springs provide the most accessible and perhaps the most representative monitoring points, alternative monitoring points may include cave streams, seeps, and wells that may intercept fracture zones, dissolution zones, or conduits. If monitoring is done merely to attempt to characterize an aquifer -- as contrasted with attempting to detect pollutants from a specific site -- the wells and seeps used as alternative monitoring points are likely to undergo much smaller fluctuations in chemical and physical characteristics than are springs.

Interpretation of variations in conductivity of a spring as an indicator of the vulnerability of a karst aquifer is probably reliable for the vast majority of North American karst aquifers and, we believe, essentially all those with a drainage-basin area smaller than about 500 square miles. Although the controls on conductivity are many and complex, it is a reliable measure and integrator of hydrologic and water-quality response to the dynamics of an aquifer. Conductivity senses and integrates the covariance of the complex response and allows a simple measure of it.

Six important caveats must be remembered:

1. An aquifer can not be classified as being "not a karst" simply because one can successfully model it numerically over so large an area that it seems to behave with the properties of a porous medium. The local perturbations that so dramatically affect site characterization and remediation are beyond the ability of analysis by present-day computer models. Such model codes are most often applicable only to granular aquifers where aquifer parameters do not vary as dramatically and over as short a distance as they can in karst aquifers.

2. Data from wells alone might not show karstification, even if it exists, because the scale of organization of flow in a karst aquifer is not likely to be adequately sensed by just a few wells. If wells alone are used for monitoring, they must be shown by rigorously designed and executed tracer tests to have hydrologic connection to the site to be monitored.

3. A karst aquifer within a single groundwater basin may be locally characterized by either of the two end-member types of vulnerability, or by an intermediate member. In part, this is because contaminants introduced into a sinkhole connected to a conduit will move more rapidly than if spilled on the ground or introduced in the area between swallets or open sinkholes, or introduced in an area where dispersed recharge and diffuse flow may predominate and conduits may be rare or relatively inaccessible.

4. Just because a spring flows from a conduit, it does not follow that it is a hypersensitive spring or that it is a discharge point of an aquifer dominated by conduit flow. It could be a moderately sensitive spring draining a similar aquifer, as determined by interpretation of variations in its water quality and discharge. For nomenclature decisions, aquifer response to recharge events takes priority over physical appearance.

5. Site-specific fieldwork may require examination and evaluation of geology, hydrology, and karst features within a radius of a mile from a site to perhaps as much as 40 miles away, depending upon the structural setting and probable size of groundwater basins.

6. The conductivity CV is not an infallible indicator of the nature and vulnerability of a spring and the aquifer it drains. Nevertheless, the CV is extremely useful and

probably reliable so often that it can be employed with a high degree of confidence. Its reliability can be tested by tracing.

An alternative to using conductivity at discharge points such as springs to characterize the nature of a karst aquifer is to obtain hydraulic conductivity or transmissivity data from aquifer tests (slug, bailer, or pump tests). Such data are characterized by a low mean and a high CV in aquifers dominated by conduit flow and by a moderate to high mean but a much lower CV in aquifers dominated by diffuse flow. Non-karstic carbonate aquifers generally have a low hydraulic conductivity mean and low CV (Smart et al., 1991). Interpretation of data from aquifer tests in karst terranes, however, should be done cautiously, for the reasons discussed.

Aquifer tests may yield valuable information (albeit inaccurate) about the flow and quantity of water in the portion of karst aquifer tested but say nothing about its recharge or storage beyond the immediate vicinity of a well. Such tests are complemented by interpretation of conductivity CV's which say nothing about aquifer parameters but do imply much about recharge, storage, and flow.

An appendix outlines two procedures for using conductivity data to prevent *aliasing* (loss of recognition of the signal from rapidly changing parameters, a consequence of insufficient frequency of sampling) and to determine an appropriate, practical, aliasing-free sampling procedure for using water quality data to characterize an aquifer, its spring, or a conduit-fed well, and for the establishment of sampling frequency during tracer tests.

CONTENTS

INTRODUCTION

This paper is written primarily for geologists, hydrologists, and engineers who are not specialists in karst but who work in carbonate terranes and deal with regulators, planners, and attorneys who must make important decisions concerning such terranes. It is also written for regulators and planners who are not likely to be trained in karst hydrology but who must educate some of the consultants they deal with. The discourse is written as a practical guide for "greater utility" (Franklin, 1781). Accordingly, we have tried to avoid the Feynman Effect (directing the content toward our colleagues, karst specialists, when we are supposed to be focusing on communication with non-karst specialists; Bartlett, 1992), but we believe that karst specialists will find it to be a valuable synthesis which revolutionizes concepts of karst aquifer classification and of what constitutes necessary sampling frequency. A less technical summary of this paper will be published elsewhere.

The Summary, the sections titled Definition of karst aquifer, Principles for classification of carbonate aquifers, and Practical evaluation of vulnerability of karst aquifers -- as well as the Appendix and the penultimate indented paragraph of the Discussion -- are the most important parts of this paper; the rest comprise the foundation for it and the necessary rationale. The Discussion is a potpourri of important points that, although extremely relevant, might have detracted from the clarity and continuity of the paper itself.

About 20% of the conterminous U.S. and 40% of the U.S. east of Tulsa, Oklahoma, is underlain by carbonate rock, mostly limestone. We maintain that almost all of these areas are, therefore, some type of karst and probably include some type of karst aquifer. It is far more judicious to operate on the assumption that they are karst aquifers -- until and unless conclusively proven otherwise. An understanding of the extraordinary hydrology of such areas is necessary if their waters are to be protected and used most wisely.

Groundwater movement in karst aquifers differs from that in granular media. It is commonly orders of magnitude faster, seemingly (but not actually) less predictable, and commonly through conduits or dissolutionally enlarged joints and bedding planes. Water movement in karst aquifers can be described by known physical laws but generally not the same ones that successfully describe and predict flow in granular media.

Karst terranes can be usefully classified according to eight major attributes: cover, rock-type, climate, geologic structure, physiography, hydrology, modification during or after karstification, and dominant landforms, each of which forms the basis for a separate classification that can be combined with the other seven (Quinlan, 1978, as partially summarized by White, 1988, p. 117-118; 1990b). Simple as the above classifications are, and useful as each of them and a combination of them may be for geomorphological and regional analysis, they are obviously much more than is needed for day-to-day use by professionals concerned with environmental, regulatory, and administrative problems of terranes potentially affected by hazardous materials.

Karst is not rare. Its hydrology is not unpredictable. When the appropriate methods of investigation are used, the properties and flow within a karst are more predictable than most professionals realize (Quinlan and Ray, 1989).

Although various sections of the long-awaited, newly issued Subtitle D regulations for RCRA (Resources Conservation and Recovery Act) and the CWA (Clean Water Act) can be easily interpreted to sanction special monitoring protocols that may be necessary for karst aquifers -- as they should -- there is only one section (Section 258.15) that specifically mentions karst, in referring to unstable areas. Karst is mentioned in the appendices that discuss this section and others. Unfortunately, in Appendix C (Supplemental Information), the discussion repeatedly and mistakenly refers to <u>karst terraces</u> where, from context, it clearly means <u>karst terranes</u> (U.S. EPA, 1991, p. 51,020, 51,047-48). Sinkhole development can be extremely important at waste disposal sites, but it is our considered opinion that the potential for groundwater contamination at such sites is far more common and nearly ubiquitous.

One of the more practical recent applications of academically-inspired karst research has been the recognition that springs and cave streams, shown by tracing to drain from a site, may be optimal points for monitoring groundwater quality and that only wells shown by tracing to drain from a site can be considered to be monitoring wells (Quinlan and Ewers, 1985; Quinlan, 1989, 1990a). These concepts have been well received by the hydrogeologic community (Beck <u>et al</u>., 1987) and the regulatory community (Quinlan, 1989) and are implicit in this paper.

There is a need for recognition of degrees of vulnerability of the water quality that can occur in different types of karst aquifers. Such recognition enables the user to think in terms of why one is concerned with the karstic nature of a terrane (Field, 1988, 1990). The "why" is answered in terms of sensitivity to effects of groundwater contamination and in terms of how to reliably monitor the various types of karst so that the monitoring is relevant, reliable, and time-sensitive.

Although much of this paper is concerned with the potentially severe consequences of spills of hazardous materials, no attempt is made here to discuss procedures for response to them. See Quinlan (1986) and Crawford (1991).

How, then, can karst aquifers be adequately protected? We will discuss why degree of vulnerability of karst aquifers is relevant to groundwater monitoring plans and suggest how to recognize a karst terrane and/or a karst aquifer. Then we will state principles for classification of karst aquifers. This will be followed by a discussion of flow, recharge, and storage within karst aquifers, relations between karst aquifer types and sensitivity to groundwater contamination, and characterization of carbonate aquifer sensitivity. We then give a simple technique for reliably predicting aquifer vulnerability insofar as it is relevant to problems of groundwater protection and monitoring, and close with a review of the use of aquifer tests to characterize flow in and vulnerability of karst aquifers, some caveats, a concluding discussion, and an appendix that describes a new procedure for reliably establishing the optimal frequency for sampling a spring.

A cautionary quotation is appropriate: "Of all the words in the hydrologic vocabulary, there are probably none with more shades of meaning than the term <u>aquifer</u>. It means different things to different people, and perhaps different things to the same person at different times. It is used to refer to individual geologic layers, to complete geologic formations, and even to groups of geologic formations. The term must always be viewed in terms of the scale and context of its usage." (Freeze and Cherry, 1979, p. 47).

An aquifer can be defined as a saturated permeable geologic unit that can transmit significant quantities of water under ordinary hydraulic gradients (Freeze and Cherry, 1979, p. 47). Stated another way, an aquifer is a body of rock or sediment capable of yielding usable quantities of water to wells and springs. If one uses or could use the water, its host is an aquifer. With this less precise definition, the rock adjacent to a cave stream that is perched above the water table could be classified as and regulated as an aquifer, even though most of the rock is not saturated. [Some state regulations would consider this cave stream as surface water because it is in a defined channel; there is case law to support such interpretation (Davis and Quinlan, 1991).]

The term <u>aquifer</u> is purposely imprecise with respect to hydraulic conductivity; it refers only to the ability to yield a usable quantity of water. A rock or sediment mass yielding less than one gallon per minute to a well and supplying one household is just as much an aquifer as one yielding thousands of gallons per minute to a well for a municipality or industry. Obviously, the latter is a better aquifer (if one needs a high yield), but the former is no less an aquifer.

By water we mean both phreatic water (water below the water table -- groundwater in the traditional sense) and vadose water (water in the unsaturated zone between the water table and the ground surface), as discussed by Domenico and Schwartz (1990, p. 9-11). Indeed, for the past 15 years, the nature and movement of water and contaminants in the vadose zone has been a major frontier in research of both groundwater hydrology in general (Devinny et al., 1990, p. 42-43) and karst hydrology (Smart and Friedrich, 1986; Bonacci, 1987, p. 28-35; Chevalier, 1988). In karst terranes, it is especially important to monitor and characterize the flow of vadose water because the vadose zone in karsts provide significant storage; flow from it to the saturated zone can be quite rapid. Monitoring of groundwater in the unsaturated zone, particularly in karst terranes, is an as yet unexplored frontier, both practically and administratively.

Several relevant definitions are given in recently issued Federal regulations concerning landfills (EPA, 1991, p. 51,017-18). "Aquifer" means a geological formation, group of formations, or port[i]on of a formation capable of yielding significant quantities of ground water to wells or springs." [The problem words here are use of ground water rather than water and the definition of significant. What is the difference between significant and appreciable? We would argue that significant, as defined by Webster's Ninth New Collegiate Dictionary, means "of a noticeable or measurably large amount", and that large is a relative term. In the southwestern U.S., a spring may flow a few quarts per hour, an insignificant quantity only by the standards of the humid east, but enough to sustain an entire local ecosystem and to be used as a water supply.] "Ground water means water below the land surface in a zone of saturation. ... Saturated zone means that part of the earth's crust in which all voids are filled with water."

The emphasis of monitoring sections of these new regulations is on water below a water table or in a confined aquifer. EPA proposed in 1988 to require monitoring elsewhere, including unsaturated zones. The agency is currently evaluating comments on that proposal and is preparing a final rule. The current regulations, however, do not preclude states from requiring monitoring in the unsaturated zone (U.S. EPA, 1991, p. 51,067).

An encyclopedic compilation of aquifer and groundwater definitions could be obtained from hydrology texts and case law, but such a compilation is beyond the scope of this paper. We are not obsessed with definitions, but we believe it is necessary to agree on them in order to achieve clear, unambiguous communication.

Four excellent texts in English provide thorough expositions on the hydrology of karst aquifers (Milanović, 1981; Bonacci, 1987; White, 1988; Ford and Williams, 1989), each with a different emphasis.

Even though there are numerous excellent texts on groundwater hydrology (Freeze and Cherry, 1979; Driscoll, 1986; Fetter, 1988; Domenico and Schwartz, 1990), applied chemical and isotopic groundwater hydrology (Mazor, 1991), and on hydrology in general (e.g., Gupta, 1989; Viessman et al., 1989), none of these texts or others describe the hydrology of karst aquifers in more than a superficial fashion, if at all. Most hydrology texts completely ignore karst. Otherwise excellent, state-of-the-art texts on environmental science and engineering and on watershed management don't even mention karst (Henry and Heinke, 1989; Brooks et al., 1991). Recent definitive books on groundwater development, on groundwater monitoring, on landfill design, and on hazardous-waste site management or remediation mention karst only in passing (Roscoe Moss Company, 1990, p. 14, 167; Nielsen, 1991, p. 64, 407; O'Brien and Gere Engineers, Inc., 1988, p. 51, 53, 56, 78), ignore it (Ward et al., 1990; Bagchi, 1990; Pfeffer, 1992; Goldman et al., 1986; Christiansen et al., 1989; O'Leary et al., 1986), or erroneously refer to karst terranes as being "rare" (Devinny et al., 1990, p. 145). A recent environmental geology text says nothing about the special characteristics of groundwater pollution in karst terranes (Montgomery, 1992).

Although the second edition of an authoritative water encyclopedia includes data on velocities characteristic of conduit flow and diffuse flow (van der Leeden et al., 1990, p. 286), little else of its content suggests the importance of karst aquifers in America. An otherwise excellent text on environmental geology includes a brief discussion of sinkhole development but says nothing about groundwater movement in karst aquifers, especially as related to groundwater contamination (Lundgren, 1986). A highly readable compendium on groundwater contamination in the United States says nothing about karst (Patrick et al., 1987) and neither does an otherwise excellent introduction to groundwater contamination written for executives, plant managers, and politicians (Bailey and Ward, 1990). A fascinating synthesis of geochemistry and fluid mechanics of permeable rocks ignores karst (Phillips, 1991) as does a reference handbook by the National Fire Protection Association written as guidance for response to hazardous materials incidents (Henry, 1989). Even the National Research Council totally ignores karst (Committee on Ground Water Quality Protection, 1986; Water Science and Technology Board, 1988, 1990a) or gives it only one paragraph in a book on groundwater models (Water Science and Technology Board, 1990b, p. 99-100). And the National Ground Water Information Center, operated by the National Ground Water Association, does not include karst as a Descriptor/Keyword on its index sheet for Proceedings volumes. The situation with respect to recognition of karst-related problems by non-karst specialists in Europe is little better, but there are some notable exceptions (Erdélyi and Gálfi, 1988; Fookes and Vaughan, 1986; Lallemand-Barrès and Roux, 1989; Nonveiller, 1989; Wynne, 1987).

The books cited above are some of the more recent on various topics. The same could be said about the inadequacy of exposition of karst-related topics in earlier books. Accordingly, most consultants, whether hydrogeologists or engineers, and most regulators, whatever their education, lack formal training in karst hydrology and rarely have the opportunity to learn about it. Many are unaware of the significance of karst. Too many others deny its existence or importance.

Only six North American universities and the National Ground Water Association regularly offer formal courses devoted entirely to various aspects of karst hydrogeology. Training in karst is also available at several additional universities where the topic is a research interest of a faculty member. All of these educational efforts are taught by well-experienced staff, but they reach an inadequate number of potential users of the concepts espoused.

IMPORTANCE OF AQUIFER VULNERABILITY TO DESIGN OF GROUNDWATER MONITORING PLANS FOR KARST AQUIFERS

In an ideal world, the concepts advocated in this paper would be regularly applied to planning and management. In the real world, however, the concepts can and should be so applied, but most applications will be to groundwater monitoring and responding to contamination situations that already exist.

All aquifers are sensitive to groundwater contamination, hence the need for aquifer protection policies and groundwater monitoring to ensure compliance with them. Knowledge of whether a spring (and the aquifer it drains) is hypersensitive rather than moderately sensitive to contamination will affect the design of its monitoring plan and can affect the design and operation of tracer tests used to establish that plan. Hypersensitive springs typically respond rapidly and possibly greatly to recharge events; they will require frequent sampling, beginning early in an event. In contrast, moderately sensitive springs typically respond slowly and in a relatively subdued manner; sampling frequency can be less and can start later in an event but sampling duration will be longer. These sampling matters are discussed in more detail by Quinlan and Alexander (1987) and Quinlan (1989, 1990a), where we used the traditional (but not equivalent) terms conduit flow and diffuse flow, respectively, and by Blavoux and Mudry (1988).

In this paper, we offer a simple technique for aquifer characterization on the basis of its relative vulnerability and the variability of its specific conductance because there is a great need for reliable, rapid, and inexpensive methods for site characterization and monitoring during remediation (Mackay, 1990) and, more important, before remediation is ever necessary. We suggest three other techniques that yield complementary results and may or may not give a higher reliability in assessment and characterization, but they involve aquifer testing and a significantly greater cost.

We recognize that evaluation of aquifer vulnerability is just one aspect of land-use planning and watershed protection often necessary for karst terranes (Rubin, 1991).

RECOGNITION OF KARST TERRANES

When environmental professionals think of karst, many of them think of the Mammoth Cave area of Kentucky with its tens of thousands of sinkholes, hundreds of sinking streams and springs, and hundreds of miles of accessible cave passages, some so large that two garbage trucks could be driven side-by-side through them. Most of these same people are unwilling to recognize, for example, the Nashville, Tennessee, area as a karst. To judge a carbonate terrane not to be a karst because it does not resemble the Mammoth Cave area is to judge the _National Enquirer_ not to be English prose because it lacks the style, grace, and wit of a Shakespearean comedy. Indeed, the classic examples of karst in the Mammoth Cave area and _The Taming of the Shrew_, however delightful and fascinating they may be, are trivially small non-representative examples of the spectrum of karst types and the spectrum of English prose.

Others have defined a karst terrane in terms of the number of sinkholes shown on a topographic map -- "X" sinkholes per square mile or per ninth of a 7.5-minute quadrangle. This is objective but naive, and not recommended. Aside from the fact that sinkholes are rare or only subtly present in some karst terranes, in others most sinkholes are not shown on the topographic map for many reasons, ranging from non-intersection of them by contours, to non-recognition of them by photo-interpreters making topographic maps, to farmers having partially filled them. Indeed, 85% of the 535 sinkholes in Winona Co., Minnesota, are not on the 7.5-minute topographic maps and were found only by field work (Dalgleish and Alexander, 1984).

A karst is not a landscape. Rather, it is a variable aggregate of diagnostic surface and subsurface features that are known to be genetically related to its development. [Admittedly, this definition is a bit circular.] We use _karst terrane_ interchangeably with _karst_ to refer to both the surface _and_ the subsurface, the lithologic expanse considered as a whole. Strictly speaking, however, _karst terrain_ refers only to the surface. [Consult Hansen (1991, p. 177) for discussion of the distinctions between _terrane_ and _terrain_.]

The necessary (essential) elements of a definition of _karst_ are that it be composed of both the landscape _and_ the subsurface features formed as a result of dissolution of bedrock (usually, but not necessarily limestone and/or dolomite), that it be characterized by a distinctive subsurface hydrology, and that it include any of a suite of distinctive surface and subsurface features that are sufficient (non-essential) for fulfillment of the definition.

The presence of any _one_ of the following types of distinctive surface and subsurface features is sufficient for diagnosing a terrane as a karst. No specific one of them is necessary. Lack (or apparent lack) of nearly all of them does not mean a terrane is not a karst:

1. Sinkholes (any closed depressions, with or without a discrete opening at their bottom, formed by dissolution and/or collapse of bedrock, with flushing and/or collapse of soil into a subjacent cavity) and internal drainage to them;
2. Dry valleys (in humid climates);
3. Springs (draining carbonate, sulfate, or halide rocks);
4. Caves (open to the surface or accidentally encountered by drilling);
5. Sinking streams (that sink at a hole known as a swallet);
6. Dissolutionally enlarged joints and/or bedding planes (as seen in cores or outcrops);
7. Grikes (soil-filled, dissolutionally enlarged joints or grooves; also known as cutters or soil karren);
8. Karren (dissolutionally, subaerially, water-carved grooves on rock, commonly subparallel).

A few examples of types 1 through 5 may be shown on a topographic map, but most of them and types 6 through 8 can only be found by field work. If one looks in the field, and does so diligently, some of these eight features, perhaps all of them, will be found. We are so confident of this that we say "If there is carbonate rock, there is almost certainly some type of karst."

The apparent lack of some or all of the above eight features in a carbonate terrane, or their non-recognition, does not mean that a karst terrane is not present. It probably means that more field work or drilling is needed to find them. Similarly, we consider the presence of just one of the features to be diagnostic of a karst because that one is usually representative of a much larger population.

The above definition of karst in terms of what is necessary and what is sufficient is totally (and independently) consistent with current European thought on the subject, thought which has evolved during the past 150 years. To paraphrase and abbreviate Gams (1991), karst is a territory formed by dissolution of bedrock and in which underground drainage of precipitation prevails. These he identifies as the absolute (necessary) elements of the definition. The so-called karst landforms and other surface features are the possible (sufficient) elements. Gam's definition of karst and ours were independently conceived but are remarkably similar.

Some people confuse a sinkhole with the karst itself. The senior author has reviewed reports that describe proposed landfill sites in flat-lying limestones such as the Mississippian-age St. Louis Limestone. These reports state, for example: "There is no karst on the property except for one, in the southeast corner, where there is an open hole." The report writers ignored the other closed contours (indicators of sinkholes) without an open hole at their bottom, and they truly confused an open sinkhole with the karst itself. This is like confusing Cyrano de Bergerac's nose with the man himself. All will agree that there was a lot more to Cyrano than just his nose. So also, there is far more to a karst than just a sinkhole or a cave. In the example cited here, the whole property is part of a karst terrane.

Like sinkholes, springs are commonly omitted on topographic maps. Fewer than 5% of springs relevant to regional movement of groundwater in the Mammoth Cave area of Kentucky are shown on 7.5-minute (1:24,000) topographic maps. The remaining 95% had to be found by field work (Quinlan, 1989, 1990a), but the presence of a few large springs not shown on the maps could be inferred from diagnostic contours suggestive of their alcove. Non-indication of springs is characteristic of topographic maps of the eastern and midwestern U.S. MORAL: In consideration of the unreliability of topographic map depiction of sinkholes and springs, absence of recognizable karst landforms or karst hydrologic features on a topographic map must not be interpreted to mean a karst terrane is absent. Topographic maps are useful -- but not reliable -- indicators of the karstic nature of a terrane.

Obviously, there are degrees of karstification, just as there is a range in the quality of English prose. One can quibble over the subjective judgement as to degree of karstification or quality of writing, but nearly all carbonate terranes are a karst and the *National Enquirer* is English prose.

The best map showing the distribution of American karst terranes, albeit incomplete, is the hitherto almost unknown useful map (in color) by Davies *et al.* (1984). Discussion of our strong disagreement with many of their classification criteria, however, is beyond the scope of this paper.

DEFINITION OF KARST AQUIFER

A *karst aquifer* is an aquifer in which flow of water is or can be appreciable through one or more of the following: joints, faults, bedding planes, and cavities -- any or all of which have been enlarged by dissolution of bedrock. This definition is in terms of hydraulics, not landforms (such as sinkholes), grikes (soil-filled joints), or hydrologic features (such as swallets, sinking streams, springs, and caves). The presence of sinkholes and any or all of these other features is diagnostic of the presence of a karst aquifer, but the absence (or unrecognized presence) of any or all of them -- on the ground or on a topographic map -- does not mean that a karst aquifer is absent.

The definition of *appreciable* (as related to the quantity of flow), like the definition of *aquifer* itself, may vary as the use varies. *Appreciable* is a relative term and does not refer to the same quantity of water in all aquifers. According to the standard American law dictionary, *appreciable* means "capable of being estimated, weighed, judged of, or recognized by the mind. Capable of being perceived or recognized by the senses. Perceptible, but not a synonym of substantial" (Black, 1989, p. 92).

Dissolutional enlargement of joints and cavities in limestone is initially extremely slow. Dissolution rates and their kinetics are described by Palmer (1991) and Dreybrodt (1988, p. 229-240). Routing of much of the initial flow may be like that described by Ewers (1982; partially summarized by Ford and Williams, 1989, p. 249-261, and Dreybrodt, 1988, p. 223-229, 240-245), Johns and Roberts (1991), and Tsang et al. (1991).

It is probable that most flow in most karst aquifers is turbulent (Ford and Williams, 1989, p. 142-148). We believe, however, that it would be a mistake to use such a criterion to distinguish a karst aquifer from one that is not. Aside from the fact that some flow in some karsts is probably laminar (Hickey, 1984), accurate measurement of flow velocities, fissure widths, and hydrostatic heads at inaccessible parts of an aquifer is difficult and ambiguous. To routinely spend considerable time and resources trying to make such problematic, unnecessary measurements distracts from the likely reasons for assessing an aquifer.

The presence of non-carbonate beds such as sand, clay, or sandstone at the surface of a terrane does not mean that the underlying carbonate beds do not comprise a karst aquifer. They usually do. Such mantled, covered, or caprock-protected aquifers, be they shallow or deep and possibly confined, can be regionally important.

RECOGNITION OF KARST AQUIFERS

As a generalization, if carbonate rocks such as limestone, marble, or dolomite are present, or if more soluble rocks such as salt or gypsum are present, assume that the water moving through these rocks is in a karst aquifer -- until or unless convincingly proved otherwise. This statement is correct probably 95% of the time. Assume also an Orwellian nature to the definition of a karst aquifer: All carbonate terranes are karstic and underlain by one or more karst aquifers, but some are more karstic than others.

If there is evidence for a karst terrane, assume that there is also a karst aquifer. The aquifer, however, may be several orders of magnitude less active than it once was, because it is a relict karst and no longer has its original climate or hydrographic setting, or because it is a buried karst which is completely or largely de-coupled from the present hydrogeochemical system (Ford and Williams, 1989, p. 507-512; Quinlan, 1978, 1990b; Samama, 1986, p. 200-242; Bosak et al., 1989; Wright et al., 1991).

The karstic nature of an aquifer can often be inferred from pumping tests in which the drawdown is stepped, even though pumping is continuous. Such stepped drawdown curves, however, can also be a consequence of flow through fractures that have not been dissolutionally enlarged. All one can be confident of is that flow is non-Darcian, so one still must be cautious. Conversely, however, a smooth drawdown curve does not mean an aquifer is not karstic.

Other indicators of the karstic nature of an aquifer -- or that it is fractured, or both -- are:

- Irregular cone of depression or no cone of depression, as defined by multiple observation wells around a pumped well;

- Non-linearity of the drawdown vs. discharge plot;

- Stair-stepped, irregular configuration of potentiometric surface (where there are enough data to depict the surface with reasonable accuracy);

- Strongly bimodal distribution of the logarithm of hydraulic conductivity of a suite of wells completed in the same formation (Smart et al., 1991). Caution: Karst aquifers may also have a unimodal log-normal distribution of hydraulic conductivity (Moore, 1988a, 1988b);

- Non-coincidence of water levels in closely adjacent wells;

- Bimodal or polymodal distribution of daily or continuous measurements of specific conductance (Ford and Williams, 1989, p. 213-214; Bakalowicz and Mangin, 1980);

- Significant variation in specific conductance and hydraulic conductivity, as interpreted from wellbore fluid logs (Pedler et al., 1990);

- Significant variations in the distribution of discharge, as measured by an impeller meter moved vertically during pumping at a constant rate (Molz et al., 1989);

- Significant variations in the distribution of flow in a pumped or unpumped well, as measured by a movable electromagnetic or thermal flowmeter in the borehole (Nyquist et al., 1991; Kerfoot et al., 1991; Hess and Paillet, 1990);

- Significant differences in the breakthrough curves in a well for different dyes injected into several wells and recovered in that well (Quinlan, 1992).

The first four of these indicators are illustrated and briefly discussed by Wisconsin Geological and Natural History Survey (1991). Proof of appreciable flow through joints, faults, bedding planes, and cavities can also be obtained by inference from cores, video logs, caliper logs, and temperature logs. Lacking such proofs, if the bedrock is limestone or dolomite, one would be well-advised to believe in the probable presence of a karst aquifer.

PRINCIPLES FOR CLASSIFICATION OF CARBONATE AQUIFERS

The behavior of an aquifer is dependent on three essentially independent properties: the mode of recharge, extent of storage, and type of groundwater transmission (flow) (Smart and Hobbs, 1986). In karst terranes, recharge may range between predominately point recharge (for instance, via sinking streams) to dispersed recharge (for instance, where water enters the karst aquifer over a broad area and through an overlying permeable sandstone, thick soil, or perhaps densely fractured rock with a thin soil). Similarly, flow can be considered as a continuum between conduit flow in large-diameter, pipe-like caves and diffuse flow via fractures and intergranular pores. Storage is largely controlled by porosity and geological disposition of the aquifer, but ranges from high, where there is a deep saturated zone, to low, where the aquifer is perched and has a limited extent and thickness. In order to characterize an aquifer, it is necessary to define its position on each of the three continua, effectively mapping it into a three-dimensional space defined by recharge, storage, and flow (Figure 1). Also shown in this figure are the general types of recharge, storage, and flow which characterize each continuum. These are discussed further below and the rationale for them is given by Smart and Hobbs (1986), but it is important to stress here that the boundaries are fuzzy. Each type grades into and overlaps with those adjacent.

Conduit flow, in the strict sense, has been and should be used to refer to flow through dissolution passages with diameters of centimeters to meters, following the usage of Shuster and White (1971), Atkinson and Smart (1981), and Smart and Hobbs (1986). Velocities are commonly high; flow is commonly turbulent. Diffuse flow, in the strict sense, is confined to tight fractures and pores with small, interconnected openings, with diameters of centimeters or less. Velocities are low; flow is commonly laminar and may be Darcian. These two terms, conduit flow and diffuse flow, have been subsequently (and erroneously) broadened by Quinlan and Ewers, 1985; Quinlan 1989, 1990a) and by numerous other writers to include implicit characteristics of water-quality and discharge variability. Between the two end-members, we can intuitively recognize a continuum of intermediate flow types. Pipe flow in open cave passages represents a flow type close to the conduit end-member. In other karst aquifers, dissolution has resulted in dense networks of open fissures which are intermediate in character between conduit and diffuse flow. Finally, flow through intergranular pores and fractures essentially unmodified by dissolution approximates the diffuse-flow end-member.

Diffuse flow, as used in describing karst aquifers, should not be construed to be the laminar, dispersed flow characteristic of granular aquifers and described by Darcy's Law. The term diffuse, as used here, is intended to mean slow, both laminar and slightly turbulent flow of water through a system of small, discrete pathways that are being dissolutionally enlarged, albeit extremely slowly. These pathways range from fractures and near-microscopic conduits to intergranular pores. Hence, a diffuse-flow karst aquifer may be described as having flow generally restricted to discrete pathways, with minimal turbulence rather than the extremes of turbulence characteristic of conduit-flow aquifers. The principle difference between a diffuse-flow part of a karst aquifer and a

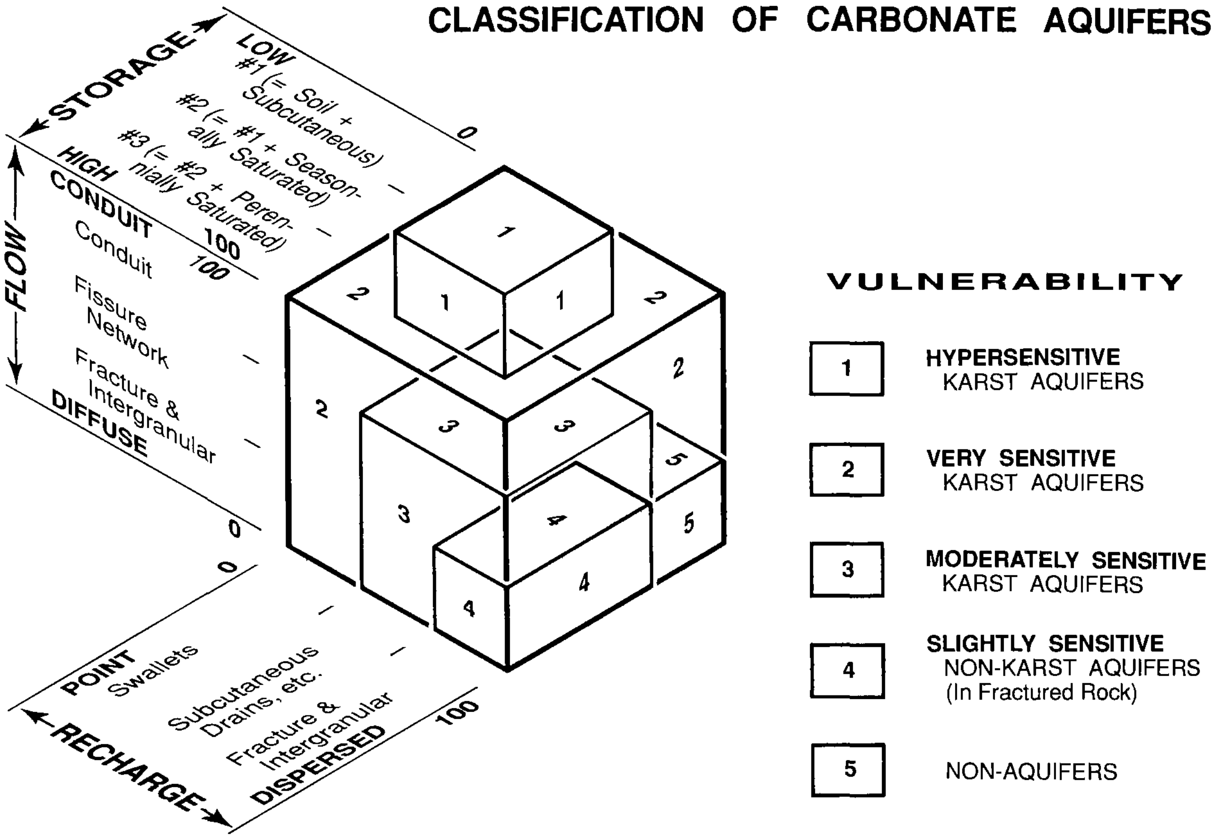

Figure 1. Conceptual model for carbonate aquifers which recognizes independence of recharge, storage, and flow. Volumes #1 through #4 depict the relative vulnerability of karst and non-karst carbonate aquifers, ranging from hypersensitive to slightly sensitive. Each of the three axes represents a different non-arithmetic continuum that ranges from 0% to 100% for each end-member. Boundaries between vulnerability fields are approximate, intuitive, and transitional. The relationships shown and implied are depicted within a cube because there are three independent variables, each of which may have a value up to 100%. The coefficient of variation (CV) of hydraulic conductivity, as discussed by Smart et al. (1991), and the CV of specific conductance (conductivity), as discussed in this paper, are each at a maximum at the upper far vertex of the cube and a minimum at the lower near vertex; the CV of both parameters decreases along the diagonal connecting them. Not all the boundaries of volume #5 (non-aquifers) are shown. This solid extends been used in two senses: the strict sense, referring to water transmission through variously sized apertures, and in the broad along part of the back edge of the cube but its edges have been omitted, for clarity. (Greatly modified after Smart and Hobbs, 1986)

Darcian granular aquifer is that the flow pathways in the former have been and are being significantly enlarged by dissolution.

To partially summarize, conduit flow and diffuse flow have been used in two senses: the strict sense, referring to water transmission through variously sized apertures, and in the broad sense, referring to aquifer type as represented by the sum of all responses to recharge, storage, and flow. The sense intended by a writer must be interpreted from context. We henceforth use them in the strict sense, to describe transmission.

Recharge ranges from point to dispersed. Point is used in the same sense that Smart and Hobbs (1986) used concentrated, but the former term was selected to be in conformity with its widespread usage in America. However, we have deliberately not used its false, apparent antonym, non-point, because it does not mean aerially distributed. Although large quantities of "high-strength" agricultural runoff may enter an aquifer at the swallet of a sinking stream, i.e., at a point, it is technically classified by most state agencies as non-point pollution. Accordingly, we have retained dispersed in the same sense as Smart and Hobbs (1986).

Point recharge is characterized by sinking and losing streams. However, even if these are absent, concentration of recharge may still occur in karst terranes with closed depressions where drainage is often via a shaft system (Smart and Friedrich, 1986). This may also occur in limestone aquifers overlain by a thick regolith and lacking obvious depressions. Aquifers with recharge through shaft systems occupy the center of the recharge continuum, grading into recharge through fissures toward the dispersed end-member. The latter is characterized by recharge through fractures and intergranular voids.

Storage ranges between high and low end-members (defined in terms of the ratio of storage volume to annual recharge). Two factors control storage, the effective porosity of the aquifer and its geological disposition in relation to regional base-level and any confining beds (White, 1977). In practice, the aquifer configuration dominates. Hence we divide the continuum into three transitional groups. The first comprises aquifers with limited storage, predominantly in the unsaturated zone, primarily in soils and the subcutaneous (epikarst) zone (Smart and Friedrich, 1986). The second is similar to the first but also has a seasonally drained saturated zone that may be part of a shallow water-table aquifer. Finally, the high-storage end-member comprises aquifers with an unsaturated zone, a seasonally inundated saturated zone, and either perennial storage in a confined aquifer or a water-table aquifer which extends well below spring level. Examples of such high-storage karst aquifers are the Madison aquifer, chiefly in Montana and Wyoming (Downey, 1984; Plummer et al., 1990), that in the carbonate-rock province of the Great Basin, chiefly in Nevada and Utah (Burbey and Prudic, 1991), and in the Floridan aquifer (Sun and Weeks, 1991, p. 5-9; see also U.S. Geological Survey Professional Papers 1403-A through I).

Storage in conduits (pipes), as a fraction of the total storage within an entire aquifer, is always very small. Thus aquifers having storage only in conduits would represent the extreme low-storage end-member.

Each of the three independent variables that determine the nature of an aquifer -- recharge, storage, and flow -- ranges from the extremes shown in Figure 1 by upper-case letters, POINT to DISPERSED, LOW to HIGH, and DIFFUSE to CONDUIT. The three major types of each of the three independent variables are shown in lower-case letters indented between their two extremes. These three sets of three major types are each part of a non-arithmetic continuum that extends from 0 to 100% fracture and intergranular recharge, 0 to 100% soil + subcutaneous + seasonally saturated + perennial storage, and 0 to 100% dominance by conduit-flow, as shown.

The eight corners of the cube correspond to the following end-members, most of which probably do not exist as pure end-members and two of which are not karst. As with any continuum, however, it is easier to think in terms of the end-members while recalling that most of reality lies between them. The pure end-member vertices are:

1. Point recharge, high storage, conduit-flow karst aquifer;
2. Point recharge, low storage, conduit-flow karst aquifer;
3. Point recharge, high storage, diffuse-flow karst aquifer;
4. Point recharge, low storage, diffuse-flow karst aquifer;
5. Dispersed recharge, high storage, conduit-flow karst aquifer;
6. Dispersed recharge, low storage, conduit-flow karst aquifer;
7. Dispersed recharge, high storage, diffuse-flow non-karst aquifer;
8. Dispersed recharge, low storage, diffuse-flow non-aquifer.

Obviously, there are numerous intermediate types of karst aquifers, but we do not propose to describe them or the end-members in detail here or to give examples of each of them. Instead, we have stated and illustrated the principles for classification of carbonate aquifers.

Remember, Figure 1 is a classification of carbonate aquifers, not karst terranes. As mentioned in the Introduction, there are seven other criteria by which karst can be usefully classified. To be practical, however, we use here only the criterion that is most relevant to the movement of and monitoring of pollutants -- aquifer hydrology.

Moderately sensitive karst aquifers may behave in a less flashy manner than hypersensitive karst aquifers because of either diffuse flow <u>or</u> high storage <u>or</u> dispersed recharge <u>or</u> any combination of them, not simply because of diffuse flow. In fact, chemograph/hydrograph criteria do not separate which of these is the case, only that aquifer responses are slower and more damped.

RELATION BETWEEN TYPES OF KARST AQUIFER AND SENSITIVITY TO GROUNDWATER CONTAMINATION

The particular susceptibility of karst aquifers to groundwater pollution is related to the rapid transmission of pollutants with little attenuation, dilution, or dispersion via dissolutionally enlarged fractures and cave conduits (Aley, 1986; Hoenstine et al., 1987; Hannah et al., 1989; Murray et al., 1981; Quinlan and Ewers, 1985; Edwards and Smart, 1989; Quinlan and Rowe, 1977). However, recharge and storage processes are also important. If recharge is via dispersed routes such as a porous sandstone caprock overlying the limestone, any pollutant entering from the surface will be considerably attenuated and delayed before reaching the cave conduit in the underlying limestone. The risks of severe contamination of a spring or other discharge point are thus somewhat reduced, although they could still be high if the contaminant had direct access to the subsurface, for instance, via a drainage well (Crawford, 1984). If there were a substantial body of water stored in phreatic fissures and pores adjacent to a conduit, further dilution and attenuation are possible before the pollutant is discharged at the spring or other discharge point. Thus, moderate to high storage and dispersed recharge will reduce susceptibility of discharge points to contamination.

Intuitively, we can infer the susceptibility of carbonate aquifers to groundwater contamination by considering the type of aquifer defined with reference to the recharge, flow, and storage continua described above. Accordingly, in Figure 1, we introduce and define four new terms for zones representing relative sensitivity to groundwater pollution: hypersensitive karst aquifer (at the upper far vertex, volume #1), very sensitive karst aquifers (volume #2), moderately sensitive karst aquifers (volume #3), and slightly sensitive non-karst aquifers (at the lower near vertex, volume #4). There is also a non-aquifer class (volume #5) in which the lack of concentrated flow and low saturation preclude practical abstraction of water. Effectively, we have added a fourth continuum to Figure 1 which ranges from hypersensitive to groundwater contamination to slightly sensitive thereto. Note that all aquifers of whatever type are sensitive to groundwater contamination and thus require proper management. For this reason, we rejected the end-member terms non-sensitive and robust.

Hypersensitive aquifers are the most karstic of all karst aquifers and are characterized by conduit flow in pipes (caves), concentrated recharge (via swallets, stream beds, or shaft drains), and low storage (limited saturated zone). There is a high probability that pollutants introduced at the surface will pass rapidly downward, without attenuation, directly into conduits where they will be transmitted rapidly via pipe-flow to springs (generally the only easily found high-yield source of water in such aquifers). Even if there is limited saturated storage, little dilution or attenuation of pollutants will occur as point recharge generally bypasses the storage sites (Atkinson, 1977; Smart and Friedrich, 1986).

Very sensitive aquifers are characterized by the presence of either well-developed flow in pipe-like caves (conduits), point recharge via swallets, or storage ranging from very low (unsaturated) to very high (saturated), or a combination of two of these.

Moderately sensitive aquifers do not have pipe-flow, point recharge through swallets and losing streams, or low storage. However, there is still a significant risk of groundwater contamination, with the possibility of rapid recharge through closed depressions, rapid transmission through dissolutionally-enlarged fissure networks, and limited dilution when seasonal storage is at a minimum.

As emphasized in our earlier definition, karst aquifers are characterized by appreciable flow through dissolutionally developed voids. It must be recognized, however, that for several reasons, some carbonate aquifers (extremely few, in our experience) have not undergone significant dissolution. These reasons include limited exposure time, presence of overlying carbonate-rich regolith, and low solubility of bedrock. These non-karst carbonate aquifers are, therefore, identical in character to clastic aquifers in which flow is via intergranular pores or fractures developed during unloading or exposure in the near-surface environment. They are shown as volume #4 in Figure 1. For these non-karst carbonate aquifers, special attention is not needed; conventional techniques of aquifer development, protection, and monitoring are probably applicable, but may have to be modified because of fracturing.

CHARACTERIZING CARBONATE AQUIFER SENSITIVITY

Introduction

The scheme described above uses observable criteria and provides a simple, intuitive basis for assessing the sensitivity of carbonate aquifers to groundwater contamination. However, we are not yet able to adequately fix the position of individual aquifers within the continua. Clearly, for administrative and regulatory purposes, we need a much simpler scheme, one which, although less rigorous, is capable of differentiating aquifers which have a high pollution risk from those which are less susceptible.

A potential solution to this problem is to use the geochemical or hydrological behavior of a major spring to characterize the nature of the aquifer it drains. We treat the aquifer as a black-box which transforms the "signal" from natural storm recharge (which has a distinctive chemical signature) into an output response (hydrograph or chemograph) at the spring, or possibly at a seep or well shown to intersect fracture zones, dissolution zones, or conduits. If this signal is substantially smoothed, delayed, and attenuated by passage through the aquifer, then the same will happen to a pulse of pollutant; the aquifer will be only moderately sensitive to contamination. If it is transmitted rapidly, with little or no attenuation or dilution, the aquifer is hypersensitive.

Background

Fortunately, characterization and interpretation of geochemical and hydrological variation of springs has been the basis for many studies of karst aquifers (see the summary in Ford and Williams, 1989, p. 193-214, and the analysis by Kiraly and Muller, 1979). In a classic paper, Shuster and White (1971) discussed the variations of water hardness, degree of saturation with respect to calcite and dolomite, and Ca/Mg ratios of springs in the Appalachians of Pennsylvania. They recognized two types of springs: (1) those with highly variable water quality, wide fluctuations in discharge, and rapid response to recharge events (which they called conduit-flow springs) and (2) those with much smaller changes in water quality parameters, higher saturation state, small fluctuations in discharge, and lagged, prolonged hydrograph response (which they called diffuse-flow springs). Similar studies by Newson (1971) in the Mendip Hills, England, supported the observed continuum in geochemical response but suggested that the highly variable springs there were dominated by point recharge at swallets, while the less variable springs were recharged predominantly by diffuse percolation. The proportion of diffuse to point recharge was similarly used by Atkinson (1977) in explaining Mendip spring hydrograph response. In contrast, Aley (1975, 1977) discussed the same hydrograph features in terms of the proportions of transit and storage water.

Despite the disparity in these explanations of an essentially identical range of spring behavior, all of them are correct. As we have discussed previously, the most karstified aquifers (hypersensitive aquifers) are characterized by low storage, pipe flow, and point recharge. There is a strong tendency for these three characteristics to be genetically and functionally linked. Thus maximal concentrated dissolution of the bedrock occurs where aggressive surface water drains into limestone as recharge from adjacent impermeable rocks. Caves develop (Palmer, 1990). Conduits eventually link the swallets directly to the springs which discharge the sometimes large flow entering at the swallets (Ford and Williams, 1989, p. 249-271). Where conduits are developed, rapid discharge of groundwater can occur, and the amount entering into long-term storage is relatively limited. The flashy, highly variable behavior of Shuster and White's conduit-flow springs is a consequence of the fact that much of their recharge is rapid, at discrete points, and into conduits, <u>and</u> the fact that the conduit flow reduces the effective aquifer storage. However, the more subdued response of their diffuse-flow springs could equally well be explained <u>either</u> by lack of point recharge, <u>or</u> by a geological disposition favoring substantial storage, <u>or</u> by the dominance of diffuse flow.

In Figure 1, variability in the chemistry and flow of a spring (or in the chemical composition and water level of a well that intersects a fracture zone, dissolution zone, or conduit) decreases systematically along <u>all</u> axes of the cube as one moves away from the most karstified end-member, at the upper far vertex. The most damped response is expected at the farthest distance from the hypersensitive vertex. As one might expect, this is the diagonally opposite lower near vertex -- aquifers that are not appreciably karstified and are thus only slightly sensitive to groundwater contamination.

Although the above generalizations by Shuster and White were based primarily on water hardness and related to discharge and lag-time, they are equally valid for numerous other water quality and quantity parameters of a spring or an aquifer. The easiest parameters to monitor, however, are specific conductance (hereafter referred to as conductivity), temperature, turbidity, discharge, and lag-time between onset of recharge and the time of hydrograph peak. The conductivity, an easily determined electrical property of water, correlates with the sum of dissolved major-ion concentrations in water and often with a single dissolved-ion concentration (Hem, 1985; Miller et al., 1988). It is a quick, extremely inexpensive surrogate for a chemical analysis when it is not necessary to know what specific major ions are present.

The nature of a spring or other suitable monitoring point (and the aquifer that feeds them) can be adequately determined by systematically measuring its conductivity and by observing and estimating its turbidity, discharge, and lag-time. These latter three parameters, as well as temperature, can be quantified, but it is not always necessary to do so. Indeed, variations in turbidity, discharge, and lag-time can often be estimated accurately enough for reliable characterization by interviewing local residents.

Geochemical Variation

The direct relationship between water hardness and conductivity allows us to use the same numerical values for discriminating spring type (or aquifer type) as did Shuster and White (1971). Diffuse-flow aquifers (as defined by them) are characterized by a conductivity in which the coefficient of variation (CV; standard deviation x 100 ÷ mean) is less than 5%. Conduit-flow aquifers (as also defined by them) are characterized by a range of conductivity in which the CV is greater than 10%. Aquifers that have a conductivity CV between 5 to 10% are interpreted to be a mixture of diffuse flow and conduit flow. [These empirical class boundaries are arbitrary but utilitarian. The general validity of using hardness at all, however, has been challenged, as discussed in the two paragraphs following the next. Nevertheless, conductivity CV has seemingly worked well to date.] We have assumed the same CV boundaries for conductivity as an index of aquifer sensitivity. For moderately sensitive aquifers the conductivity CV is 5% or less. For very sensitive aquifers it is between 5 and 10%, and for hypersensitive aquifers the CV is greater than 10%. Work is in progress to verify and, if necessary, adjust these boundaries.

We estimate that most of the other chemical parameters will probably have similar CV's.

Shuster and White's use of carbonate hardness for defining the nature of an aquifer was strongly criticized by Bakalowicz (1976) who emphatically asserted that conductivity is a regionalized variable -- in the sense of Matheron (1965) and as the concept is discussed by David (1977, p. 91-114) and by Henley (1981, p. 9-34) -- and that it provides a more accurate and sensitive indicator of the characteristics of an aquifer. Regular measurements by Bakalowicz (1979), cited by Bakalowicz and Mangin (1980) and Ford and Williams (1989, p. 213-214, Fig. 6.24), have shown the following about the frequency distribution of conductivity at springs in various types of aquifers:

Granular aquifers: unimodal, relatively high conductivity
Fractured aquifers: unimodal, relatively low conductivity
Karst aquifers: polymodal, wide range of conductivity

Bakalowicz and Mangin (1980) and Mangin (1984) stated that the CV of hardness is not a valid measure of the complexity of the polymodal distribution of hardness. This is partly because the accuracy of the standard deviation used to calculate CV is based on the validity of the assumption of a normal distribution (Hamburg, 1977, p. 42-44). They are technically correct, but we disagree with the necessity for their rejection of its use. The standard deviation can still be calculated for a polymodal distribution, and it is still a measure of dispersion about the mean. However, its meaning is different. For example, the mean $\pm$ two standard deviations might include only 80% of the conductivity values rather than 95.5% of them. We justify use of the CV of conductivity, imprecise as it is for a polymodal distribution, with a quote from the eminent statistician John W. Tukey: "Far better an approximate answer to the *right* question which is often vague, than an exact answer to the *wrong* question, which can always be made precise" (Tukey, 1962, p. 11). [This same rationale could be employed to justify cautious use of standard techniques for interpreting slug, bail, and pump tests in karst aquifers.]

Bakalowicz (1979) also showed that the frequency distribution for daily measurements of conductivity at a particular karst spring in the Pyrenees varied significantly from year to year over a 4-year period and was trimodal; the amplitude and frequency of the maxima changed from year to year and, as one might expect, from storm to storm. We have used the daily data in his dissertation to calculate that the CV of conductivity at this spring for the 4-year period was 6.4%, 7.9%, 6.4%, and 5.7% (mean = 6.6%) -- in spite of its high variability -- suggesting that it is a very sensitive spring but not greatly different from a hypersensitive spring. We interpret the slightly more than low value of its CV to be a function of fluctuations of recharge, storage, and flow in the aquifer studied. We do not have comparable long-term data for American or British karst springs, but a review of existing data and a study of other springs are underway.

The use of conductivity as a measure of the nature of an aquifer is not an issue here. The issues are whether its CV is a reliable measure thereof and what arbitrary CV boundaries can (or should) be drawn between spring and aquifer types. These matters are being evaluated by us and will be described in future publications. In the meantime, we consider CV < 5%, CV = 5 to 10%, and CV > 10% to be practical boundaries for discrimination between aquifer vulnerability types.

Conductivity CV's have been used to identify and quantify the difference between two groundwater basins (Quinlan and Ewers, 1981, p. 488, 490). A hypersensitive spring (mean = 326 micromhos per cm; CV = 14%) was clearly differentiated from a moderately sensitive spring (mean = 488 micromhos per cm; CV = 2%) with about one-thirtieth the drainage area of the hypersensitive spring and discharging in the middle of a distributary. The latter is used for water supply; the former is not.

Two generalizations, based on the field experience of the senior author, can be made about the relations between conductivity, spring and aquifer type, and groundwater basin size. First, for equivalent basin size, a moderately sensitive spring has a significantly higher conductivity than a hypersensitive spring; 20 to 30% higher is not unusual. Second, for both spring types, conductivity is directly proportional to basin size, which is a control on residency time (time for reaction) of water.

Hydrological Behavior

For hypersensitive aquifers, the ratio between flood-flow and base-flow discharge may well be 10:1 to perhaps 100:1. Lag-times between storm events and hydrograph maxima will range from a few hours to a few days and be directly proportional to basin size. Duration of response may be a few days to a few weeks. In moderately sensitive aquifers, the ratio between flood-flow and base-flow discharge may be 3:1 or less; lag-time will range from days to weeks or even months, again in direct proportion to basin size, as discussed by White (1988, p. 186), but as conduit-flow and diffuse-flow aquifers. Duration of response may be several weeks to several months.

One could use just the ratio of flood-flow discharge to base-flow discharge to determine flow-type (and, by inference, probable spring type or aquifer type), as discussed by White (1988, p. 184-187) concerning aquifer type, but such data are essentially nonexistent in either published or open-file form.

One might also analyze the exponentially decaying recession limb of several spring hydrographs and calculate the response time, t_R, that is characteristic of the spring and its aquifer type. Physically, t_R is the time it takes for the hydrograph to fall to about 37% (= 1/e) of its maximum. This time ranges from a few days to a few thousand days and depends upon many factors not yet fully understood. More accurately, t_R is the reciprocal of epsilon (what has been called the exhaustion coefficient), the constant in the variable exponent of e in the empirically derived equation that describes the relationship between spring discharge and time, as discussed by White (1988, p. 186). Epsilon is the slope of the straight line describing that relationship when the data are plotted on semi-log paper, as explained by Krumbein and Pettijohn (1938, p. 209-210). [This graphical procedure is mentioned here because a similar technique will be proposed, in the Appendix, for interpreting the recession limb of storm-related conductivity data and design of an optimal plan for reliably sampling springs and conduit-fed wells.] Such an analysis of discharge may be useful if one has the luxury of time and money for instrumentation of a spring and if it is physically possible to gauge it reliably. There are, however, other problems, as discussed by White (1988, p. 186):

> "Precipitation events are variable in intensity, spacing and duration. If t_R is much less than the mean spacing of precipitation events, hydrographs will be flashy and the yearly record will consist of a series of sharp peaks. If t_R is on the same order of magnitude as the mean spacing of precipitation events, individual storms will tend to overlap but seasonal changes in precipitation will appear. If t_R is much longer than the mean spacing of precipitation events, the hydrographs will be broad and relatively featureless. Large systems tend to have longer response times than small systems simply because of the longer times needed to transmit the storm impulse from input to output. Thus, short response times and high discharge ratios are indicators of conduit systems only for small basins."

White's analysis, above, is generally valid and we agree with it, but he assumes basin homogeneity and ignores the possible influence of local, near-spring input of storm runoff in a large basin. By inference, his conclusion about short response times and high discharge ratios would be also applicable to interpretation of conductivity variations. The validity of this inference depends upon the distinction between small basins and large basins.

Within our experience in the eastern, midwestern, and southwestern U.S., we estimate that damping of spring hydrograph and chemograph response caused by basin size is not significant at springs draining less than about 500 square miles. We conclude that, for whatever the several reasons that control a spring's behavior, monitoring of its conductivity and determination of its conductivity CV are reliable measures of spring (aquifer) variability and response to recharge which may include pollutants. Clearly, research is needed on basin hydraulics and response as a function of numerous variables.

<u>Limitations</u>

Using only the variation in conductivity CV and discharge for spring (aquifer) characterization is not infallible. For example, the fourth author of this paper, E.C. Alexander, Jr., and his students have been studying the hydrology of Lanesboro Spring, near Lanesboro, Minnesota. Its hydrographs, chemographs, and conductivity plots are those of classic diffuse-flow springs -- more than

99% of the time. Yet dye injected at either of two wet-weather swallets, 2.5 miles away, during base-flow conditions was recovered in only 8.5 hours and with a breakthrough curve having a very steep rising limb. The recession limb is complex, has multiple peaks that are reproducible in repeated tests, and takes 2 weeks to return to background levels. The occasional muddying of the Lanesboro spring waters (3 times in 2 years) and the results of these repeated tracer tests* during base flow can be explained in terms of infrequent storm-related point-recharge into an otherwise dispersed recharge, moderate to high storage, <u>conduit</u>-flow aquifer that is probably very sensitive to pollutants for more than 99% of the year and hypersensitive for less than 1% of the year. The behavior of this spring is totally consistent with the ideas expressed here and by Smart and Hobbs (1986), but at variance with those of Shuster and White (1971) if one thinks of the aquifer just in terms of diffuse flow (as suggested by the very low CV for its conductivity) rather than recharge, storage, and flow. The spring behavior also brings into question the reliability of conductivity CV in assessing aquifer vulnerability.

We do not have enough data to know how common such inconsistency is between low CV of conductivity and occasional hypersensitivity of a spring and its aquifer, or whether it occurs as a general phenomena. We do believe, however, that such storm-related behavior is probably uncommon and contributory to a distinctive-mode on a conductivity-frequency plot. Nevertheless, we also believe that use of the conductivity CV, as we advocate, will usually be reliable for an overwhelming majority of springs. This reliability justifies its use.

The valuable concepts of <u>conduit-flow</u> <u>aquifer</u> and <u>diffuse-flow</u> <u>aquifer</u> (as defined by Shuster and White, 1971) have been usefully applied for more than 30 years, even though Bakalowicz (1976), Bakalowicz and Mangin (1980), Mangin (1984), and Smart and Hobbs (1986) questioned them and called attention to their deficiencies. We recommend here that these two terms be retired as the basis for classification of aquifer type -- even though there is usually a

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* Briefly stated, most tracer tests involve injection of a recognizable non-toxic substance at one point and recovery of it at another, yielding information about flow-direction, -dispersion, -destination, and -velocity (Quinlan and Alexander, 1990; Atkinson and Smart, 1981; Lepiller and Mondain, 1986). A well-designed, properly performed, and correctly interpreted tracer test, most commonly employing one or more fluorescent dyes, is the most powerful technique for studying karst aquifers (Aley, 1986; Ford and Williams, 1989, p. 219-241; Quinlan, 1989; Quinlan and Ray, 1989) but, until recently, they have rarely been used, except by karst specialists. The value of tracing in all types of aquifers, however, is now widely recognized (Moltyaner, 1990a, 1990b; Maloszewski and Zuber, 1990; Tsang <u>et</u> <u>al</u>., 1991; Adams and Davis, 1991).

partial overlap of so-called conduit-flow aquifers with hypersensitive aquifers and usually a partial overlap of so-called diffuse-flow aquifers with moderately sensitive aquifers. The terms <u>conduit-flow</u> and <u>diffuse-flow</u> should be retained, however, as descriptors of flow type and to refer to aquifers dominated by one type of flow or the other.

PRACTICAL EVALUATION OF VULNERABILITY OF KARST AQUIFERS

Procedure

A spring is the pulse of the aquifer it drains. Understanding the variations in its discharge and water quality (or those of other suitable monitoring points) is the key to understanding the hydraulics of its (their) aquifer and assessing aquifer vulnerability.

We believe that interpretation of water quality and quantity parameters, as described herein, even though less meticulous than proposed by Smart and Hobbs (1986), is practical and justifiable for the purposes we propose -- to identify aquifer vulnerability -- and adequate for use by non-specialists. The classification procedure we advocate is less subjective than what they proposed. Smart and Hobbs are more rigorous, but we are more practical. Quantitative characterization of recharge, storage, and flow proportion are difficult, time-consuming tasks best left to specialists.

The conductivity CV of a spring (and, therefore, of a groundwater basin presumed to be representative of the aquifer it drains) should be determined from at least 20 to 40 daily, weekly, or other regular or random measurements of its conductivity. The measurements should be taken at least a day apart and <u>must</u> include both base flow and flood flow during the estimated peak discharge. We can not say how many samples are required; we can only advise that caution must be used to make the samples statistically representative of the various conditions within the aquifer. This is easier said than done.

It is critically important to include conductivity measurements collected under <u>both</u> base-flow and storm-flow conditions. Twenty conductivity measurements taken only under base-flow conditions, perhaps under the pressure of an arbitrary 30-day deadline, will probably have a CV of less that 5% and will "prove" that the spring is characterized by diffuse flow. Such a conclusion is wrong because it is based on data that do not represent the true variability of the system. Reliable interpretation of conductivity is aided by also having daily measurements of rainfall, the maximum daily rainfall, and the maximum rainfall per storm.

We reiterate: If the conductivity CV is 5% or less, the spring (aquifer) is moderately sensitive to pollution and is probably characterized by mostly diffuse flow. If the conductivity CV is greater than 10%, the spring (aquifer) is hypersensitive and is characterized by a much greater proportion of conduit flow. If the conductivity CV is between 5 and 10%, the spring (aquifer) is very sensitive and is probably a mixture of both diffuse flow and conduit flow. Also, reliable inferences can be made about whether an aquifer is dominated by moderate sensitivity or is hypersensitive, both by observing the turbidity and discharge during, immediately after, and shortly after storms of different intensities, and by interviewing landowners who are familiar with the springs and their behavior.

Not a single published American report known to us has enough data that would enable an investigator to run a few simple conductivity CV calculations and determine the vulnerability (sensitivity) of the groundwater basins in a given area. Therefore, if one needs such data for a specific area, it is highly likely that they can be acquired only by field work.

Difficulties and Limitations of Vulnerability Evaluation

The conductivity CV is a useful method for characterizing the vulnerability of aquifers; it must be remembered, however, that the hydrograph or chemograph (a plot of chemical concentration or conductivity vs. time) of a spring reflects the recharge, flow, and storage characteristics of the groundwater basin that it drains. For site-specific investigations in karst aquifers, the springs used to characterize an aquifer will probably not be on a site to be evaluated and may even be several miles away. Their possible relevance to the hydrology of the site must be evaluated with the aid of geologic maps of the area adjacent to the site; dye tracing may also be necessary to prove or disprove their relevance of particular springs to a given site (Quinlan et al., 1988).

The conductivity CV method has some limitations in areas where there are few springs and where springs are only located several miles from a site under investigation and tracing has demonstrated that flow velocities are, for example, 1 mile per year or slower. An alternative approach in these areas is to interview well owners and determine if their well water turns turbid or muddy after heavy rains. Be cautious. Approximately 90 to 95% of the wells in karsts dominated by conduit flow are in the diffuse-flow part of the aquifer that feeds the conduits. [These are usually hypersensitive aquifers.] Accordingly, many well owners, perhaps 150 to 200, need to be interviewed before one can be confident about using wells to ascertain whether an aquifer is hypersensitive rather than moderately sensitive. Some of these 5 to 10% of wells that turn turbid or muddy are the ones most likely to be positive for dye during a tracer test.

We hear the argument that "My client can't afford to interview 150 to 200 well-owners." We reply that "If you need the truth, you can't afford not to do so."

Evaluation of conductivity CV has severe limitations for some karst terranes, such as those in much of Florida, that are characterized primarily by recharge and flow rather than also by discharge. Discharge at springs may be several tens of miles away, perhaps beneath the ocean; flow velocities may be no more than a few hundred feet per year. The springs, if accessible, can be monitored for conductivity, but they will probably be inaccessible. Well-to-well tracer tests may be practical for determination of flow velocities for either the design of monitoring systems or for determination of wellhead and springhead protection areas based on time of travel, but tracer tests would be unsuccessful in delineation of wellhead or springhead protection areas based on basin boundaries because it would take too long to do so. Conductivity CVs of wells in such terranes are not likely to be meaningful.

Currently, some of the serious limitations in applying the conductivity CV at springs as a method of assessing aquifer vulnerability are: 1) the false assumption (by some agencies, clients, and consultants), that monitoring wells alone are sufficient to characterize a karst aquifer, and 2) an unwillingness by some of the same to extend monitoring efforts beyond the boundaries of a site under investigation, to all possibly relevant springs, wells, and base-level stream segments. [If such short-sighted views and restrictions are allowed to prevail, and they should not be, the quality and credibility of a tracing investigation is severely compromised. So also is the integrity of the investigator compromised.]

We argue that a limited number of boreholes, for example five at a given site, do not provide a statistically valid confirmation of the existence or probable non-existence of conduits in the subsurface. Benson and La Fountain (1984) have shown that drilling 1,000 3-cm holes per acre is necessary in order to have a 90% chance of intercepting a cavity about 2.3 m in diameter. A lesser number would be needed for finding a conduit of the same diameter, but many of the conduits of interest at a given site are likely to be only a tenth as wide (or smaller). A thorough survey of springs and existing wells (see above discussion), followed by a well-designed tracer test, properly performed, and correctly interpreted, will provide much more cost-effective and accurate information about the site.

We also hear the argument that "My client (and his attorneys) does not wish to perform any investigations off-site." In the real world, water-flows (and contaminants) do not respect property boundaries; they travel beyond them. A consultant may have to convince clients and their attorneys that judicious investigation that minimizes potential liability requires following the flow wherever it may go. Off-site field work is probably always cheaper than the off-site courtroom work it can prevent.

Well-meaning supervisors, clients, regulators, and consultants with whom the reader interacts may exert pressure to assess an aquifer from the office or without the amount of field work we are recommending as minimal. In response, ask whether he or she would respect or trust a C.P.A. who was willing to audit accounts over the telephone. A field problem can only be resolved by field work.

We wish to emphasize that:

1. If a terrane is underlain by limestone or dolomite, as indicated by an accurate geologic map and/or site-specific observations, the site is extremely likely to be part of a karst and have a karst aquifer;

2. With one exception, there is no way without fieldwork, as described, to determine whether the aquifers in a site area are hypersensitive rather than moderately sensitive to contamination and probably dominated by conduit flow rather than diffuse flow. The sole exception is when so much field work has already been done in the area that one can confidently interpolate or extrapolate on the basis of experience or pre-existing state-of-the-art literature.

USE OF AQUIFER TEST DATA TO CHARACTERIZE FLOW IN AND VULNERABILITY OF KARST AQUIFERS

In some karst areas, where domestic wells are sufficiently abundant and accessible, it may be possible to characterize flow within an aquifer by using aquifer tests and to make inferences about its vulnerability from such data. One advantage of this approach is that it directly measures the transmission (flow) properties of an aquifer, whereas the conductivity CV and hydrograph analysis methods (discussed above) integrate flow, recharge, and storage properties of the aquifer. In addition, slug or bailer tests and single-well pump tests are simple, rapid, and cost-effective methods for determining average of the vertical distribution of aquifer horizontal properties.

Briefly, the aquifer-test method of assessing aquifer vulnerability involves determination of hydraulic conductivity* from slug or bailer tests, single-well pumping tests, or multi-well pumping tests, calculation of CV of log hydraulic conductivity, and interpretation of cumulative probability plots of log hydraulic conducivity. The distribution of hydraulic conductivity tends to be log-normal. Therefore, the logarithm of hydraulic conductivity data should be used for calculation of statistics, and the geometric mean of hydraulic conductivity should be used to best represent the average of log-normally distributed values (Domenico and Schwartz,

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* Hydraulic conductivity, the volume of fluid (usually water) that will move through a medium in a unit of time, T, under a unit hydraulic gradient through a unit area measured perpendicular to the flow, has the dimensions L/T because the L^3 of volume divided by the L^2 of area cancels to L, a length. Consequently, hydraulic conductivity, a volume, is often erroneously thought to be a velocity.

1990, p. 66-67; Bradbury and Rothschild, 1985). The geometric mean and CV of the hydraulic conductivity distribution are then used to make inferences about aquifer vulnerability. As a rule of thumb, we suggest that at least 20 data points should be used to calculate these statistics. Additional data points, however, will provide a more representative estimate of the distribution of hydraulic conductivity within an aquifer.

For hypersensitive aquifers (those probably dominated by conduit flow), the CV will be very high (>500%), with wells penetrating both highly transmissive fissures and conduits plus non-conduit blocks with low permeability. [The probability of a well intercepting a conduit is extremely small, but doing so would simply increase the CV of the sample]. However, the geometric mean of the hydraulic conductivity will be low. For moderately sensitive aquifers (those probably dominated by diffuse flow), the mean hydraulic conductivity will be moderate to high, reflecting the dissolutional widening of openings in the limestone, but the CV will be much lower, reflecting the more homogenous distribution of flow within the aquifer (Smart et al., 1991). In both cases, the log-normal probability plot of hydraulic conductivity may well be polymodal.

This hydraulic conductivity approach has some advantages over the indirect analysis of specific conductance CV and hydrographs, discussed above, because it allows direct determination of flow characteristics of carbonate aquifers. The specific conductance and the hydrograph are a response to the processing by the aquifer of a signal that is not uniquely controlled by its flow characteristics and is also dependent on the nature of recharge to and storage in the aquifer. Fortunately, many of the most karstified (and therefore most troublesome) carbonate aquifers have conduit flow, concentrated recharge, and low storage, thus providing an unambiguous signal. Other aquifers show a more damped signal, implying diffuse flow, but this may result from an absence of concentrated recharge (for instance, where recharge is through a permeable caprock or thick soil mantle not breached by sinkholes) or high storage (for instance, in a deep, saturated zone of an aquifer).

There may be a methodological problem with the use of hydraulic conductivity for characterizing a carbonate aquifer. The thickness of the open interval of a borehole tested may bias an evaluation of its hydraulic conductivity. More specifically, long intervals may tend to average and dampen its true extremely high and low values. The measured high hydraulic conductivity of a well is always less than the actual value for a fracture or other cavity intercepted by it (Maureen Muldoon, Wisconsin Geological and Natural History Survey, written communication, January 1992). The possible relationship between the thickness of developed interval in a borehole and the hydraulic conductivity in different types of carbonate aquifers needs to be evaluated.

As appealing, objective, useful, and diagnostic as interpretation of, say, 30 hydraulic conductivity measurements may be, in actual practice obtaining them may be difficult, time consuming, and infeasible in some areas. There is little doubt, however, that evaluation of the CV of hydraulic conductivity from slug or bail tests is superior in many ways (but not all ways) to evaluation of the CV of specific conductance. The latter technique can be cheaper but it takes longer and may have to be delayed because of climatic seasonality.

Slug and bailer tests are reviewed by Fetter (1988, p. 196-200), Domenico and Schwartz (1990, p. 164-168), and Dawson and Istok (1991). There are many limitations to applying such tests, both theoretical and logistical. When an investigator uses the standard techniques for interpretation of bailer or slug tests (Hvorslev, Papadopulos, and many others) or for single-well pumping tests, he must be familiar with and willing to accept the assumptions inherent in those analytical solutions. All of these methods assume Darcian flow and assume also that the hydraulic conductivity is homogeneous over the interval tested. Since these assumptions are rarely, if ever, valid for karst aquifers, the resulting hydraulic conductivities <u>should</u> <u>not</u> be used to calculate flow velocities or times of travel or to calibrate computer models. The variation in the hydraulic conductivity distribution, however, may be used to draw conclusions about relative vulnerability of an aquifer.

Logistically, slug tests are probably the easiest aquifer tests to perform. The ideal method of characterizing the variation of hydraulic conductivity of a karst aquifer would be to drill wells specifically for that purpose, then perform packer tests at a set interval in them (Milanović, 1981, p. 237-251; Domenico and Schwartz, 1990, p. 163-164). This is not practical for most studies and, as a result, slug tests are commonly performed in monitoring wells and in existing domestic wells where the pump has either not yet been installed, or it has been pulled in order to perform the tests.

If an investigator is using domestic wells as points to measure hydraulic conductivity, single-well pumping tests (those without an observation well) can not be used to reliably determine hydraulic conductivity, but they do provide an advantage over slug tests in that the pump does not have to be pulled. The drawdown and recovery data (easily recorded with a downhole pressure transducer) from a single-well pump test can, however, be used to calculate residual drawdown and transmissivity (Domenico and Schwartz, 1990, p. 162-163; Kruseman and de Ridder, 1990, p. 193-197; Dawson and Istok, 1991, p. 59-61). A major problem, however, with any attempt to reliably determine transmissivity or even hydraulic conductivity from any test but a packer test, is that in a karst aquifer it is often difficult, if not impossible, to do so from a well that partially penetrates an aquifer in which the thickness is unknown or (in some cases) indeterminable.

The hydraulic conductivity measured by a slug test, a single-well pump test, or a multi-well pump test is the average hydraulic conductivity measured over the entire test interval. Using this "average" hydraulic conductivity to predict flow velocities or travel times will result in significant errors, however, because karst aquifers are characterized by highly heterogeneous hydraulic conductivity distribution (i.e., low-permeability blocks, and high-conductivity conduits or dissolution zones) and most of the flow is through the high-permeability zones.

Hydraulic conductivity can also be measured using multi-well pump tests. The use of these tests is limited to those projects with budgets that are sufficient to install a pumping well and several observation wells. The standard techniques for the interpretation of data from any type of pump test (Theis, Hantush-Jacob, etc.) require the erroneous assumption of isotropic conditions; other analysis techniques, however, can be used to calculate both the horizontal anisotropy ratio (Papadopulos, 1965), as well as the horizontal to vertical anisotropy ratio (Weeks, 1969). The assumptions of the different pump test interpretation techniques are clearly discussed by Kruseman and de Ridder (1990) and Dawson and Istok (1991). They also discuss techniques developed by Streltsova and Boulton and by Streltsova that may be more suitable for karst aquifers. Thiery et al. (1983) have also tried to develop a reliable technique for interpreting pump-test data from karst aquifers, but they too have been unsuccessful. The discharge of a well, particularly one in a karst aquifer, is the integral of the hydraulic character (hydraulic conductivity, dispersivity, storage, and tortuosity) of all the different dissolutionally-enlarged fissure-systems that feed it (Quinlan, 1992). The hydraulic conductivity calculated from pump test data still represents a value averaged over the interval tested and should be not used in calculations of Darcian or non-Darcian flow.

In spite of the potential errors in hydraulic conductivities calculated from slug and pump tests in karst aquifers, and their possible misuse, we do believe in their value for evaluation of aquifer hydraulic properties and probable vulnerability. The CV of specific conductance represents a response to the combined effects of recharge, storage, and flow, and gives a direct, integrated measure of the vulnerability of a karst aquifer to pollutants. In addition, the data are relatively inexpensive to collect. However, it may take longer to collect or be delayed because of climatic seasonality. Both types of information are complementary and needed. Neither type is intrinsically superior to the other.

A CAUTIONARY TALE

A detailed study, using geophysical logs, borehole video logs, lithologic core analyses, and unspecified aquifer tests, was made of the dissolution porosity and permeability at a Florida site (Robinson and Hutchinson, 1991). [The thoroughness of their well-designed and carefully executed study was much more than we have recommended here.] Tracer tests were performed by injecting a fluorescent dye into the open-hole interval of a well and measuring the movement induced by pumping 1000 gallons per minute at a well

200 feet away. Dye concentration at the pumped well was measured continuously with a filter fluorometer. Based on standard assumptions about flow in porous media and a porosity of 20%, the theoretical arrival time for the dye should have been approximately 40 days, with a 4-day persistence time for the dye. Actual breakthrough time was 5 hours. The peak was at 22 hours and the persistence time was 28 days. A second breakthrough of dye occurred at 36 days after injection and its persistence time was 8 days. This second breakthrough is not attributable to response to a storm (James Robinson, U.S. Geological Survey, Tampa, Fla., oral communication, December 1991). The bimodal distribution of tracer arrival time was interpreted to indicate dual porosity in the aquifer, but we suggest that bifurcation of a flow route, followed by rejoining of the trunk route, is a more plausible explanation (Tsang et al., 1991; Ford and Williams, 1989, p. 226-228).

This Florida study illustrates the heterogeneity of the aquifer and the impossibility of using carefully (and expensively) acquired aquifer parameters to make correct predictions about travel times in karst aquifers.* Most important, it shows that the assumptions of uniform distribution of porosity and attendant diffuse flow, as has been assumed and proposed in current strategies of wellhead protection for the Floridan aquifer (DeHan, 1988), are probably not valid. Again, the conceptual model inadequately corresponds to reality.

CAVEATS

The summary of this paper lists six important caveats. They require no additional explanation. Please go back and reread them.

DISCUSSION

Experience has shown that, in the midwest and south, there is sometimes a rough correlation between spring morphology and flow type. Springs dominated by conduit flow tend to have high-energy discharge from a large alcove that is eroding headward, commonly have dry high-level orifices that function only in response to storms, and may have a deeply eroded channel. In contrast, springs dominated by diffuse flow commonly have low-energy discharge from a very small alcove, shallow channels, and fontaphilic (spring-loving) vegetation. These correlations are guides, for first ap-

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* The discrepancy between actual tracer velocities and those based on predictions from analysis of cores and logs is partly a consequence of extrapolation from a borehole scale of study to a field scale, as discussed for karst aquifers by Kiraly (1975; summarized by Ford and Williams, 1989, p. 134, Fig. 5.4) and Smart et al. (1991), and as recognized in other aquifers by Domenico and Schwartz (1990, p. 84-87, 371-84) and many others.

proximations during the first few days of field work.

A review of post-1960 classifications of karst aquifers was made by Ford and Williams (1989, p. 166-170). No attempt has been made here to repeat it. The aquifer classification we have proposed is consistent with and complementary to those of Mangin (1984) and Ford and Williams (1989, p. 169). It is an update and revision of one first proposed by Smart and Hobbs (1986).

Figure 1 depicts a generalized conceptual hydrogeologic model of karst aquifers. As such, it incorporates all the essential features of their physical system. It is not, however, a site-specific, accurate representation of the conditions beneath and adjacent to a particular site. Although the degree of detail necessary for various sites differs, presentation of a site-specific conceptual model should be accompanied by maps and cross sections. Conceptualization is a means of achieving a graphic idealization of the actual geologic/hydrologic conditions, and it deliberately ignores minor features that are not important to the overall picture.

The arbitrary but convenient dividing points in the continuum between conduit flow and diffuse flow are 20% and 80%. The intermediate members of the continuum, having 20 to 80% conduit flow and 80 to 20% diffuse flow, have been known as <u>mixed flow aquifers</u> (Atkinson and Smart, 1981; Quinlan and Ewers, 1985). This term, too, should be retired. [Strictly speaking, all aquifers that are not 100% one flow-type or the other are mixed flow, but it has been convenient to draw intermediate boundaries at 20% and 80%.]

Diffuse flow can be better understood by making an analogy with the flow of water through a stack of bricks; flow is chiefly between the bricks rather than through them. We recognize, of course, that water also moves within the rock matrix of a karst aquifer, just as it would within the bricks, but we consider the intergranular Darcian component of the flow to be unimportant to this discussion. The importance of such dual porosity, especially during tracer tests, has been analyzed in a series of papers by Maloszewski and Zuber, the most recent of which is Maloszewski and Zuber (1990). A recent mathematical analysis of dual porosity has been made by Douglas and Arbogast (1990) and its role in water quality distribution within fractures and intergranular pore space has been briefly described by Edmunds (1981).

The complex relationships between conduit flow and diffuse flow, particularly as related to the role of dual porosity during recharge events, have been mathematically analyzed by Onder (1986). He showed that base-flow recession through a conduit may be related to aquifer properties, as might be expected.

A substantial amount of the water in an aquifer dominated by conduit flow moves as diffuse flow, as explained by Atkinson (1977). In the British aquifer he analyzed, 60 to 80% of the discharge is via conduits; active storage in the diffuse-flow part of the aquifer that feeds them is 30 times greater than in the conduits. Approximately 50% of the recharge is from sinking streams and sinkholes; the remaining 50% is via slow percolation through the soil.

There is a perception among some hydrologists and engineers that, because conduits comprise such a small percentage of an aquifer and because most of the rock surrounding them may be characterized by diffuse flow, overall, there is no significant problem of aquifer vulnerability. Wrong! This is like saying that the hole blown into the bottom of a battleship by a torpedo is only 10 feet in diameter but, because the rest of the steel is impermeable, there is no problem of sinking. Wrong again!

There are two further complications. First, almost all of the discharge from all aquifers dominated by conduit flow occurs at a point or group of points along a line. A small component of the discharge is as diffuse flow, as described by Atkinson (1977). Second, most karst aquifers dominated by diffuse flow also discharge at a point or group of points, but some may have almost all of their discharge over a strip or a broad area, as discussed by Quinlan (1989) as an exception to his Implicit Assumption #1. Our experience suggests that point discharge is predominant in most carbonate terranes. Accordingly, no consideration to such nonpoint discharge has been given in this paper.

In lieu of daily or twice- or thrice-weekly measurements of specific conductivity, it is likely to be far more cost-efficient, and could be more reliable, to continuously record it with a transducer and a data-logger. Also, having a continuous record of conductivity facilitates making an analysis that enables optimization of sampling frequency, as discussed below, and if such would be useful for a given project. The utility of the conductivity data could be inexpensively enhanced by using the same data-logger to also continuously record temperature with a platinum-resistance thermometer and turbidity. Interpretation of temperature data, although it can be quite subtle, can give significant useful insights into aquifer hydraulics (Meiman, et al., 1988; Davies, 1991).

Continuous recording of water-quality parameters also eliminates the possibility of *aliasing*, a sampling error that can introduce significant bias into estimates of the magnitude and frequency variation of a parameter and reliable characterization of it -- all as a result of sampling with insufficient frequency. Aliasing is a phenomenon in which a high-frequency component of a signal takes on the identity of one with a lower frequency. [The familiar illusion of stagecoach wheels in a movie first going forward, then apparently backward, then forward again is an example of aliasing.] Signal processing theory, information theory, spectral analysis theory, and time-series analysis theory all include a theorem which states that a continuous, periodic signal, with frequency components in the range $f = 0$ to $f = f_{max}$ per unit of time, can be reconstructed from a series of equally spaced samples if the sampling frequency exceeds $2f_{max}$ samples per unit of time. $f_{max} = 1/2\Delta t$, where Δt = the sampling interval and f_{max} = the Nyquist number, the fastest frequency detectable. The slowest frequency detectable is zero; it never repeats. Trend lines have zero frequency and infinite period. These concepts and many others relevant to interpretation of water-quality data are discussed by Meade and Dillon (1991, p. 16-19), Karl (1989, p. 4-8), Bandat and Piersol (1986, p. 335-339), Pierce (1980, p. 35-39), Bras and Iturbe (1984, p. 172-177), Davis (1986, p. 257-258, 349), Gardner (1988, p. 49-52), Chatfield (1975, p. 155-158), Gottman (1981, p. 15-16). Aliasing can be prevented by continuously recording a parameter or by taking samples extremely frequently, perhaps every 30 seconds. A procedure for using conductivity data to prevent aliasing of water quality data and for determination of an appropriate aliasing-free sampling protocol for using water-quality data to characterize an aquifer or its spring is given in the accompanying appendix. The procedure is similar to that already discussed as used to analyze exponential decay of the recession limb of a hydrograph. We believe the procedure is also applicable to the design and interpretation of tracer tests.

Aliasing of water-quality data can be a serious problem because it can inadvertently and seriously bias their interpretation, but it is not mentioned by Gilbert (1987) or in any other water-quality publications we have reviewed. Only Davis (1986), of the several geostatistical books known to us, discuss aliasing of spectral data, but there are several signal-processing texts such as that by Robinson (1980) which recognize aliasing of seismic data.

It is highly likely that most water-quality studies and monitoring of springs and wells characterized by conduit flow are flawed by aliasing unless they have been sampled frequently (perhaps daily) and/or as related to storm events. Unless they have, such effects are incapable of accurately characterizing the dynamic changes in water quality except by improbably good luck. Most assuredly, this is a fatal defect of the monitoring studies at Mammoth Cave National Park described by Meiman (1991); as designed, such studies are intrinsically incapable of achieving their purpose. Probable aliasing of water quality parameters is also a deficiency of the recent synthesis of data on the largest and most famous spring in France, Vaucluse (Puig, 1990).

Time-series analysis and spectral analysis of data from well-designed studies of regularly sampled and continuously sampled spring waters was first applied in various papers by Mangin and by Bakalowicz, as partially summarized by Ford and Williams (1989, p. 210-214). Application of these data-analytical techniques to other studies, like those described by Mazor et al. (1990) and those cited in the following paragraph, will lead to significant improvements in understanding of the processes operating in karst aquifers.

The complexity of flow within an aquifer dominated by conduit flow, plus interpretation of the lack of coincidence among the pulses of stage, conductivity, and temperature, are discussed by Meiman et al. (1988) and, to a lesser extent, by Ford and Williams (1989, p. 204-214). Interpretation of coincidence among the pulses associated with a different type of conduit flow is discussed by Idstein and Ewers (1991).

Further discussion of the interpretation of hydrographs of springs and streams is given by Bonacci (1987, p. 75-81), Ford and Williams (1989, p. 193-203), Domenico and Schwartz (1990, p. 15-17), and Hall (1968). Spectral analysis and time-series analysis of spring discharge have been used by Bakalowicz and Mangin (1980), and Mangin (1982, 1984) as a basis for aquifer classification. Chemograph separation is discussed by Ford and Williams (1989, p. 204-214). Detailed summary of these topics is beyond the scope of this paper, but some are discussed in the accompanying appendix.

Spills in karst terranes frequently occur on surfaces with a soil profile 5 to perhaps 50 feet thick and are sometimes confined to the soil. With luck, the entire spill is confined to the soil. Many workers would assume that Darcian flow equations adequately describe the movement of liquids through the soil. Unfortunately, non-Darcian macropore flow (Quinlan and Aley, 1987; Everts et al., 1989; Wells and Krothe, 1989) commonly occurs in the soil; contaminants can move rapidly through the soil to, and perhaps into, the bedrock. Fluid movement through soil with macropores can not be analyzed as though it were through a granular medium. Indeed, Watson and Luxmore (1986) found that macropores and mesopores in an Appalachian karst area, which together constitute only 0.3% of the soil volume, account for 96% of all infiltration. This topic is discussed in great detail by Wilson et al. (1991). Further, the pollutant may be perched at the soil-bedrock contact and may move hundreds of feet along it before intercepting a dissolutionally enlarged joint which allows access into bedrock of the karst aquifer below.

One of the more challenging frontiers in contemporary karst hydrogeology is the epikarst (also known as the subcutaneous zone). This is the dissolutionally well-developed zone immediately beneath the soil and above the relatively dry zone above the phreatic zone. In a sense, it is much like a perched aquifer, but usually there is no discrete perching bed. As discussed by Ford and Williams (1989, p. 120-121 and in many other places throughout their book), Williams (1983, 1985), Bonacci (1987, p. 28-35), Friedrich and Smart (1981), Smart and Friedrich (1986), Field (1990), Quinlan and Ray (1991), and McCann and Krothe (1991), there is significant water-storage capacity and sufficient interconnection to diffuse the movement of pollutants and tracer dyes widely. Drainage from the epikarst is not uniform but is down preferential pathways to the subsurface. Residence time for pollutants or tracers in the same epikarst can range from hours to tens of years (Quinlan and Ray, 1991). Much is being learned about flow and storage in epikarst, both in the U.S. and in Europe (Smart and Friedrich, 1986; Chevalier, 1988), but most of it has not yet been published. We believe that epikarst hydrology is the most enigmatic in all of karst. Assume that an epikarst exists beneath any soil mantle -- until you prove otherwise. Hydrologists, engineers, and regulators should be prepared to encounter epikarst. They should be prepared also to be confused and frustrated by it.

Some experienced hydrologists believe that slug and bailer tests measure little more than skin-effects and a small, local volume adjacent to the well bore. The results of such tests can be non-representative and misleading. They may be in vogue chiefly because they are quick, inexpensive, and readily create a favorable response in some regulators. All this is more true than not. But, as such tests are orders of magnitude superior to laboratory determinations of permeability of a core sample and more representative of aquifer hydraulic properties in a karst or any other aquifer, so also, pump tests are significantly superior to slug and bail tests. Nevertheless, tens to perhaps hundreds of pump tests are necessary to accurately characterize an aquifer. All this is to be expected.

As pointed out by Kiraly (1975) and quoted by Ford and Williams (1989, p. 134, Fig. 5.4), karst aquifers become both more heterogeneous and more anisotropic with passing time. Their average total porosity is directly proportional to the reference volume of rock considered and may range over three orders of magnitude -- depending upon the scale at which investigation is performed. Accordingly, it could be convincingly argued that a well-designed, carefully interpreted tracer test, because it may sample a far larger volume of an aquifer, can be superior to a pump test for characterizing <u>some</u> properties of an aquifer. The two types of test are complementary, however. Neither is intrinsically superior to the other.

Many topics have been discussed in this paper. Its most important points are that:

- Clear, accurate definitions of <u>karst</u> and <u>karst aquifer</u> -- as well as practical techniques for recognizing them -- are needed. We have fulfilled this need.

- Regulations concerning karst should focus on protecting waters of karst aquifers. Sinkhole collapse is a relatively minor problem.

- Karst aquifers should be classified according to the three highly variable major controls on their nature: recharge, storage, and flow, as shown on the cube depicted in Figure 1, and grouped into three classes of vulnerability: hypersensitive, very sensitive, and moderately sensitive.

- Karst aquifers should no longer be classified as conduit-flow aquifers or diffuse-flow aquifers. These terms should be retained, however, as useful descriptors of flow type.

- Systematic measurement of the specific conductance of springs during base flow and during and after flood flow, with evaluation of the coefficient of variation (CV) of such data, is a valuable and probably reliable predictor of not only the net effect of an aquifer's recharge, storage, and flow characteristics but also its relative vulnerability to the adverse effects of contaminants and mismanagement.

- A major deficiency of most attempts to monitor water quality of springs and wells in karst terranes is aliasing, the failure to detect and recognize high-frequency fluctuations (which can also be high-magnitude fluctuations) in water quality because the water is not sampled often enough. The highest frequency signal that can be resolved has a wave length that is twice the distance (time) between successive regular observations.

- Two objective, sensitive, flexible procedures for determining a cost-efficient, aliasing-free sampling frequency for reliable characterization of springs, aquifers, and conduit-fed wells now exist. They are described in the accompanying appendix. We believe these procedures are applicable to: 1) Reliable assessment of water quality, 2) Sampling for and interpretation of tracer tests in which dye-concentration is not monitored continuously, and 3) Design, operation, and interpretation of pulse tests (Ford and Williams, 1989, p. 226-228).

- The empirically established values for the CV of conductivity, recommended for use in separating the three types of vulnerability of karst aquifers, are < 5%, 5 to 10%, and > 10%. The reliability of these values is being reviewed by the authors, but we do not expect any substantial changes in them.

- Imperfect as their results in karst aquifers usually are, slug, bail, and pump tests should be performed in an effort to characterize the aquifer. An individual test has little significance, but evaluation of the geometric mean and the CV of a representative sample (at least tens) is a useful technique for approximating the hydraulics of karst aquifers and their differences.

Finally, a few words should be said about how this paper was written. Although a manuscript was completed in time for the meeting, it was constructive dialogue with participants there that forced all of us to rethink our assumptions, interpretations, and conclusions, and to reformulate our ideas. The three authors of the original manuscript believe that this final version has been significantly enhanced by the input and active participation of the others and the reviewers we have acknowledged. We know other meeting participants have had similar experiences in completing their manuscript. This is a good, satisfying way to do better science and is one reason why we believe that conference proceedings should be distributed <u>after</u> a meeting, not <u>before</u> or <u>at</u> it.

SOLICITATION

Readers of this paper are invited to correspond with the authors and suggest improvements in the definition of <u>karst</u> <u>aquifer</u> and ways to classify such aquifers meaningfully, usefully, and practically.

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APPENDIX

PROCEDURE FOR PREVENTION OF *ALIASING* AND FOR DETERMINATION OF APPROPRIATE SAMPLING FREQUENCY FOR A SPRING OR CONDUIT-FED WELL

The chemical character of a discharging spring can be considered to be a signal, a hydrologic time series that can be recorded (monitored) continuously or sampled regularly. The main problem in sampling, however, is how to choose the sampling interval, Δt. It is obvious that sampling leads to some loss of information and that this loss gets worse as Δt increases. Aliasing occurs. It is expensive to make Δt very small, so a compromise value must be sought. Determination of the maximum frequency of the signal in a spring chemograph, f_{max}, is quite simple if one is dealing with a sinusoidal function with a constant frequency, but hydrologic functions are less regular (less periodic or non-periodic) and have a highly variable frequency. Accordingly, we propose a method for obtaining a practical approximation of f_{max}, a method based on the common relationship in which specific conductance (conductivity; C) varies inversely with discharge (Q). Our rationale for the method will be similar to and complementary with that used for calculating the response time, t_R in the section of this paper concerned with characterizing hydrological behavior as a measure of aquifer sensitivity. Although we can compensate for the fact that some springs are characterized by a short-lived increase in conductivity at the onset of a storm event (because relatively mineralized water is pushed out by piston-flow), we will ignore these short-lived peaks in this description and utilize the well-known inverse relationship between conductivity and discharge of springs and streams, a relationship caused by the relatively low conductivity of precipitation and its dilution of groundwater.

The appropriate sampling interval, free of aliasing, will be determined by a technique we propose here. This technique consists of evaluation of a semi-log plot of conductivity vs. time, empirically determining the coefficients of the exponential equation describing the relationship between these data, and use of a coefficient in this equation to calculate the aquifer (spring) recovery time, T_R, using a procedure devised by Hess and White (1988). The sampling interval is then calculated from this recovery time. [We recognize the irony and seeming redundancy of designing a sampling plan to design a sampling plan.]

It is more reliable, more convenient, and far less expensive and less labor-intensive to monitor the conductivity continuously for the purpose of designing the sampling plan. It would be wise to do so through several storms, and for several months, so that the variation, if any, in spring behavior can be detected, so that fine structure (if present) in the chemograph can also be detected, and so that subsequent possible aliasing by discrete sampling can be prevented. Accordingly, the following description is based on the use of data from continuous records of conductivity rather than discrete samples for it.

We classify conductivity chemographs into three gradational groups, A, B, and C. For group A, chemographs with a smooth recession limb, perform the following 7 steps:

1. On a plot of conductivity vs. time (extending over several months), sketch a smooth curve for background conductivity,

C_B. [The background is likely to show a broad seasonal trend.]

2. On the chemograph for each storm event with a good record, pick the lowest point after the trough and on the recession limb where it begins to be linear. This is Day 0 (t_0) and C_0 for each storm event.

3. For each of the storm events selected for step #2, pick the post-C_0 values of C at 6 am and 6 pm, record them, the corresponding value for C_B, the difference between them (ΔC), and the values for time, t. Do so only for the near-linear part of the curve.

4. For each data set, plot the following on semi-log paper: ΔC (on the y-axis, the log scale) vs. t (on the x-axis, the arithmetic scale).

5. Determine b, the conductivity coefficient, in the following exponential equation in which b is the slope of its straight line on semi-log paper:

$$\Delta C = C_0 e^{-bt}$$

A procedure for easily solving this equation graphically is given by Krumbein (1937) and Krumbein and Pettijohn (1938, p. 209-210). Alternative graphical procedures are given by Mackey (1936, p. 113-115) and Davis (1955, p. 16-20). [Another alternative is to perform a computer-aided regression analysis.]

6. The reciprocal of b is the recovery time, T_R, for each storm event, as described by Hess and White (1988, p. 250-251). This is a measure of how long it takes the conductivity of a spring (aquifer) to rise to a value that is approximately 63% (= 1 - 1/e) of its pre-storm value. The T_R for several storms can be averaged. Alternatively, one might select the appropriate T_R for the recession limb of a particular storm being monitored by comparing its nascent hydrograph or chemograph shape with one already studied.

7. As a first approximation, we recommend sampling every $0.05T_R$ from the onset of a storm until $0.2T_R$ after its conductivity minimum, C_{min}, or stage maximum, Q_{max}, then every 0.1 T_R for the duration of T_R, and followed by $0.2T_R$ for as long as desired. One might change to sampling every $0.4T_R$ at $4T_R$ after C_{min} or Q_{max}. These sampling frequencies are a guide; sampling should generally be done at reasonable hours of the day rather than at 3:21 a.m. because that was the calculated value -- unless, of course, one is trying to characterize a high-frequency signal. Experience may show that use of different coefficients of T_R yield results more desirable.

For group B, conductivity chemographs with spikey or finely undulatory structure, select a sample interval, Δt, that is twice the highest frequency to be resolved. For example, a signal with 10-minute frequency peaks should be sampled at least every 5 minutes. T_R is ignored.

For group C, conductivity chemographs with a coarsely undulatory structure in the recession limb, one can follow the procedure for group B or sketch a smooth line that curvilinearly straightens them and then follow the procedure for group A. Use this sketched line to determine the values for t and C described in steps 2 and 3, above.

Although there is usually a slight phase-lag and an inverse correlation between chemograph and hydrograph structure, sometimes there is no relationship -- as when, for example, there is a semi-sinusoidal spring hydrograph from a siphon having a 20-minute period. Sometimes also, there may be a fine structure in the conductivity chemograph that is not present in the hydrograph or thermograph (Meiman et al., 1988).

The procedures described above minimize the possibility of aliasing of samples. If followed, they yield an objective, reliable, non-aliased, lowest cost, legally defensible sampling plan that maximizes the probability of accurate characterization of an aquifer with conductivity data and/or other water-quality parameters. We are excited at the prospect of testing these plans at a wide range of spring types and conditions.

BIOGRAPHICAL SKETCHES

Dr. James F. Quinlan, P.G., is president of Quinlan & Associates, Inc., a consulting firm specializing in problems of carbonate terranes. He was Research Geologist for the National Park Service at Mammoth Cave, Kentucky, for 16 years and has been an independent consultant on karst for more than 10 years. He earned a Ph.D. in geology at The University of Texas at Austin (1978). His experience includes 36 years of research and observations in karst terranes of 26 states and 25 countries, and work as a consultant in many of them. He has written or co-written more than 180 publications on karst-related topics. He is co-chairman of ASTM Subcommittee D18.21.09 on Special Problems of Monitoring in Karst and Fractured Rock Terranes.

Dr. Peter L. Smart is a lecturer in geography at the University of Bristol, England, from which he also received his first degree and doctorate. He has extensive research experience in karst hydrology and geomorphology in many different areas of the world, including the Bahamas, China, Indonesia, Malaysia, and several European countries.

Geary M. Schindel, P.G., has been Project Manager and Senior Hydrogeologist for Eckenfelder Inc. since 1990. He was previously

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Dr. E. Calvin Alexander, Jr., is a Professor in the Department of Geology and Geophysics at the University of Minnesota. He has a B.S. in Chemistry (1966) from Oklahoma State University and a Ph.D. in Chemistry (1970) from the University of Missouri at Rolla. The central theme of his current research is the rate of movement of fluids in hydrogeology. He and his research group are utilizing a variety of methods to measure flow and residence times of water in aquifers, which can range from hours to tens of thousands of years.

Alan J. Edwards is a research student in the Geography Department of the University of Bristol, from which he also obtained his first degree. He is currently completing his doctoral dissertation concerning the hydrogeology of quarries and landfills in the karstified limestone aquifer of the Mendip Hills.

A. Richard Smith earned a B.S. in Geology (1964) from The University of Texas at Austin. He has been involved in various hydrogeological activities for two major corporations and is now Senior Geologist of the Ground-Water Protection Branch, Bureau of Solid Waste Management, Texas Department of Health. His experience in karst geomorphology and hydrology includes investigation of Texas caves as an associate of the Texas Speleological Survey, ongoing long-term study of gypsum karst in West Texas, and exploration for karst-related sulfur deposits in evaporite rocks.

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Recommended Administrative/Regulatory Definition of Karst Aquifer, Principles for Classification of Carbonate Aquifers, and Practical Evaluation of Vulnerability of Karst Aquifers

By: James F. Quinlan, Peter L. Smart, Geary M. Schindel, E. Calvin Alexander, Jr., Alan J. Edwards, and A. Richard Smith

Q. **Should the definition of karst aquifer include the modifier of rock type? Isn't the term too broad without such a modifier?**

A. Yes and no. More specifically, because the great majority of karsts are developed in carbonate rock, the word karst implies a limestone or dolomite karst. If one were referring to a karst developed in another rock type, one would refer to it as a gypsum karst, a salt karst, a carbonatite karst, etc.

Q. **There is an inherent problem in lumping all carbonate aquifers together as "karst", especially when the word karst is used in regulations. By your broad definition, 98% of Missouri, for example, could be classified as a karst terrane -- in spite of the fact that 60% of the state is not significantly affected by karst. Why not restrict the word karst to areas where there are environmental problems and use another word for the other carbonate aquifer areas?**

A. Approximately 60%, not 98%, of Missouri has limestone or dolomite cropping out at the surface, and approximately 80% of that 60% (50% of the state) is karst, as we have defined it in this paper. Without trying to nitpick over semantics, a key phrase in the above question is significantly affected. We can agree to disagree with the accuracy of the unknown questioner's estimate of 60% of the erroneous 98% but, nevertheless, address the principle raised. The definition of karst is vague enough without making it more so, as it would be if the questioner's suggestion were followed. We do not believe in changing traditional definitions. Those we have given herein are a codification, fine-tuning, and synthesis of traditional uses; they replace definitions poorly worded by non-specialists in karst. The new terms we have introduced, describing aquifer sensitivity, may achieve what the questioner suggested about using another word in lieu of karst, but it does so without changing the meaning of karst. Also, a key word in the question is are. It excluded might be. We believe it is better to err on the side of conservatism. Aquifer contamination, like death, can be so permanent.

Q. **Doesn't the work of Scanlon and Thrailkill (1987) in the Bluegrass Karst region, near Lexington, Kentucky, contradict what you claim about the reliability of inferring the nature of an aquifer from analysis of its spring waters?**

A. At first glance, the work of Scanlon and Thrailkill (1987) might seem to invalidate the relations we allege between water quality and physical characteristics of springs and their aquifers. They claim that the two spring types studied, major (large) and high-level (small), could not be discriminated on the basis of water-quality parameters. We agree. But our review of their published data and their description of behavior of both spring types during storms shows unambiguously that each of them is a conduit-flow spring. Their work does not question or invalidate our conclusion on the practical utility of conductivity measurements for assessing aquifer types or aquifer vulnerability.

LEGAL TOOLS FOR THE PROTECTION OF GROUND WATER IN KARST TERRANES

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ABSTRACT

Ground water in karst terranes is exceptionally vulnerable to contamination due to human activities such as waste disposal, septic tanks, chemical spills, and storage of gasoline in underground tanks. Because of this vulnerability, legal tools for the protection of ground water are extremely important in karst terranes. These legal tools include source controls, land-use controls, which may be utilized by state and local governments and by several types of quasi-governmental entities. Users of ground water can assert their water rights to protect ground water.

Source controls include statutes and regulations governing the licensing of potential sources of contamination and design and operating standards to prevent release of contaminants. Land use controls include zoning regulations and the power of eminent domain. These controls should recognize the exceptional vulnerability of ground water in karst terranes, but most statutes and regulations do not provide for any differential treatment of karst aquifers. Unfortunately, many of those that address karst only deal with the potential for surface collapse and not the rapid migration of contaminants in solution channels. Additionally, wellhead and sensitive area protection programs in karst terranes should not be based upon arbitrary circles around sources of water supplies, but upon well-designed studies delineating ground water flow in solution channels.

Finally, ground water users in karst terrane can rely upon principles of water law developed for surface water to protect their water rights. The principle of reasonable use has been applied to ground water, particularly ground water flowing in

discrete channels in karst terrane.

INTRODUCTION

Ground water in karst terranes is exceptionally vulnerable to contamination due to human activities such as waste disposal, septic tanks, chemical spills, and storage of gasoline in underground tanks. Water supplies dependent upon ground water in karst terranes are also subject to disruption by subsurface disturbances such as mining, and by diversion of flow by an "upstream" user.

Because of this exceptional vulnerability, legal tools for the protection of ground water are extremely important in karst terranes. These legal tools include source controls, land-use controls, and water rights. They may be utilized by federal, state and local governments and by several types of quasi-governmental entities.

Source controls include statutes and regulations governing the licensing of potential sources of contamination and design and operating standards to prevent release of contaminants. These need to be tailored to recognize this special vulnerability of ground water in karst terranes, but, unfortunately, many of these statutes and regulations only deal with the potential for surface collapse and not the rapid migration of contaminants in solution channels.

Land use controls include zoning regulations and the power of eminent domain. In addition to counties and municipalities, there are other quasi-governmental entities, such as watershed districts, that possess some of these powers. The federal wellhead protection provisions can act in the manner of land use controls to prevent the location of potential sources of ground water contamination.

Finally, ground-water users in karst terranes may be able to rely upon principles of water law developed for surface water to protect their water rights. The principle of reasonable use has been applied to ground water, particularly to ground water flowing in discrete channels in karst terrane.

SOURCE CONTROLS

Legal Framework

The protection of ground water was not a specific focus of concern until the 1970s, and the legal framework for ground-water protection on the federal level and in most states is fragmented. It is made up of various federal and state laws, some of which touch only indirectly on ground water protection. These controls and exercises of rights can be set in perspective by reading a recent critical review of public policy considerations that enter into the assignment of a regulatory scheme to one level of government rather than another.[1]

Federal laws governing potential sources of ground-water pollution as an area of major concern include the Safe Drinking Water Act (underground injection control and wellhead protection); the Resource Conservation and Recovery Act (solid and hazardous waste management and underground storage tanks); the Comprehensive Environmental Response, Compensation, and Liability Act (investigation and remediation of uncontrolled hazardous substance sites); the Toxic Substance Control Act (disposal of PCB's and asbestos); the Federal Insecticide, Fungicide, and Rodenticide Act (pesticide use); the Uranium Mill Tailings Radiation Control Act; the Surface Mining Control and Reclamation Act, and the Atomic Energy Act of 1954 (radioactive waste disposal).[2]

Of these, the Safe Drinking Water Act, the Resource Conservation and Recovery Act, and the Surface Mining Control and Reclamation Act allow states to assume primary management authority to carry out the federal legislation, if the state programs meet federal guidelines and are approved by the federal agency. State regulations under these laws can generally be more stringent than federal regulations. In addition to federal laws and authorized state programs under those laws, states have regulated other potential sources of ground-water degradation, including septic tank systems, oil exploration, and water withdrawal.

Specific Provisions for Karst Terranes

Federal statutes and regulations governing potential sources of ground water degradation have generally not addressed karst terranes. The only instance in which karst terranes have been addressed in federal source-control regulations is in the new EPA municipal solid-waste landfill regulations, promulgated under RCRA. Under these regulations, new and existing municipal solid waste landfills in "unstable areas" must include a demonstration that "engineering measures have been incorporated into the [landfill] design to ensure that the integrity of the structural components of the [landfill] unit will not be disrupted." An "unstable area," under the regulation includes karst terranes.[3] Guidance for the investigation of site-specific potential for subsidence and collapse is to be included in the technical guidance document the agency plans to issue within six months.

This regulation deals only with the potential for surface collapse and not with the potential for rapid migration of contaminants through solution channels and the difficulty of detecting the release of contaminants in karst terranes. Although none of the monitoring regulations specifically address karst terranes, they are sufficiently flexible to allow state agencies to require proper monitoring.

Several state source-control statutes and regulations address karst terranes specifically. One of the most comprehensive examples is the Minnesota Solid Waste Disposal Regulations, which deal with the difficulties of preventing and detecting movement of contaminants in karst terranes as well as the potential for surface collapse.

The Minnesota Technical Requirements for Solid Waste Facilities require the siting of a solid waste landfill only in an area where the ground water flow paths are known in sufficient detail to enable reliable tracking of pollutant movement and where it is feasible to construct a monitoring system with sufficient monitoring points to assure that pollutants can be detected and tracked.[4] This general siting requirement would make siting in karst terranes difficult without a dye-trace study to delineate ground-water flow paths and establish monitoring points.

The regulations go on to state:

> A land disposal facility must not be located on a site where: (1) there are karst features, such as sinkholes, solution channels, disappearing streams, and caves, which may cause failure of the leachate management system or prevent effective monitoring or containment of a release of leachate; ...[5]

Kentucky solid waste regulations also focus on the difficulties of characterizing ground water flow and detecting pollutant migration in karst terranes. In applying for a solid waste landfill permit in a karst environment, the applicant must characterize both diffuse and discrete flow conditions, and the state may require dye trace studies before finalizing a ground water monitoring plan.[6] In addition to the general requirement that a landfill may only be sited in an area where the uppermost aquifer is capable of being monitored in a manner that detects migration of pollutants and where corrective action is possible once pollutants are detected, Kentucky regulations prohibit placement of waste within 250 feet of a feature of karst terrane.[7] This buffer-zone requirement, however, gives a false sense of security, since waste placed within 250 feel of a karst feature could easily be still within the karst terrane.

Tennessee solid waste regulations establishing landfill location requirements for karst areas address both potential for surface collapse and migration of contaminants but fail to deal with the difficulties of reliably detecting contaminant migration in karst terranes. The Tennessee regulation states:

> If a facility is proposed in an area of highly developed karst terrane (i.e., sink holes, caves, underground

conduit flow drainage, and solutionally enlarged fractures) the applicant must demonstrate to the satisfaction of the Commissioner [of Environment and Conservation] that relative to the proposed facility siting:

(i) There is no significant potential for surface collapse;

(ii) The ground water flow system is not a conduit flow which would contribute significant potential for surface collapse or which would cause significant degradation to the ground water; and

(iii) Location in the karst terrane will not cause any significant degradation to the local ground water resources.[8]

Tennessee's ground-water monitoring provisions for solid waste landfills, however, do not address the difficulty of detecting contaminant flow in karst terrane, nor have dye-trace studies been required by the state for the karst demonstration envisioned by the regulation.[9] The regulation also gives far too much discretion to the Commissioner to decide when to require a karst demonstration and whether a conduit flow system could cause degradation to the ground water.

Alabama has a similar presumption against the siting of landfills in karst terranes unless a site specific demonstration is performed, but the site-specific demonstration required does not address the particular difficulties of determining ground water flow paths and tracking contaminant migration in karst terranes.[10] Other states that include specific standards for location of solid waste landfills in karst terranes include Florida and Georgia, but their requirements seem to be limited to preventing waste disposal in or near sinkholes.[11]

Local Government Source Control Programs

General police power ordinances which employ such regulatory mechanisms as permits, licenses, inspections, and standards to prevent potential sources of pollution have also been used by local governments to protect ground water. Health codes adopted by local governments, or regulations imposed by local health departments, have an advantage over land-use controls, discussed below, since they can be imposed retroactively to deal with existing pollution threats as well as future threats. They may also gain more public support than zoning laws.

In most states, unless state and federal programs preempt local authority, local source control efforts to protect ground water can proceed by any of several routes, including county and municipal ordinances, local health department regulations, watershed district regulations, and utility district regulations.

For example, the Cape Cod Planning and Economic Development Commission designed model groundwater protection ordinances for enactment at the municipal level and by local boards of health after designation of their region as a Sole Source Aquifer protectable under the Safe Drinking Water Act. One ordinance, designed for adoption by municipalities, regulates all firms handling toxic or hazardous materials. Another, designed for adoption by local health departments, regulates both the installation of new underground storage tanks and the maintenance of older ones.[12]

Quasi-governmental entities, such as watershed districts and utility districts, may also be able to regulate potential sources of ground water contamination. Many states provide for the creation of watershed districts that are given broad powers to conserve soil and water resources within the natural boundaries of a watershed.[13] Among the powers delegated to districts by the Tennessee legislature, for example, are "to take such steps as deemed necessary by (the) board of directors for the promotion and protection of public health within the boundaries of the district" and "to do all things necessary and proper for the protection of the watershed and the lands and waters within the boundaries of the district."[14] Although most watershed districts are created to deal with surface streams, the regulatory powers of a watershed district should be able to protect ground water in karst terranes where there is a clear connection between ground water and surface streams.

Many states also provide for the creation of water utility districts to provide for water supply. Where a utility district depends upon ground water from wells or springs, the power to maintain the source of supply that is granted by the state legislature may authorize regulations for the control of pollution sources which potentially endanger a ground water. For instance, under Tennessee law utility districts are empowered to acquire, construct, operate, and maintain systems for the furnishing of water, and to exercise all powers necessary for the accomplishment of these purposes.[15]

Local source-control regulations also need to recognize the particular difficulties in ensuring ground water protection in karst terrane. Where the federal or state source control regulations have not gone far enough in addressing these difficulties, local regulations may fill this gap.

LAND-USE CONTROLS

Land-use controls are well-suited for preventing ground-water contamination in karst terranes. Sensitive area protection programs and wellhead protection programs are appropriately implemented through zoning and other land-use controls.

Most local governments have broad zoning powers for the protection of the public health and welfare. Some states, such as Tennessee, have given local governments explicit power to zone for conservation of water supply.[16] Zoning may take numerous forms, including prohibitions of land uses with serious potential to contaminate ground water, design or performance standards on land uses, limitations on density or lot sizes, and overlay districts designating recharge or well-head protection areas.[17]

Another form of land-use control is the purchase or condemnation by local governments of restrictive easements across land in critical recharge areas. Most local governments can condemn land or rights in land for public purposes, including protection of water supplies. A ground-water protection easement allows the owner of property to retain the land and use it for any purpose that does not present a threat to ground water, while requiring less compensation by the local government than the purchase or condemnation of the entire property.

For land-use controls to be effective in karst terranes, careful mapping of ground water flow is necessary. Reliance upon arbitrary circular restrictive zones around water supply wells or springs is unlikely to protect water supplies in karst, since contaminants may flow long distances in solution channels with little attenuation.

Several communities have utilized land-use controls to protect ground water.[18] An example in karst terrane is the comprehensive watershed development ordinances adopted by the City of Austin, Texas, to protect the Edwards Aquifer. Austin has delineated special watershed zones for controlled development density and has prohibited almost all development within "critical water quality zones" in aquifer recharge areas.[19]

Although Lexington, Kentucky, and the surrounding county were primarily concerned with prevention of flooding of sinkholes in response to storms and blockage of sinkhole drains by sediment that runs off in response to erosion triggered by construction activities, their ordinance also protects groundwater quality.[20] Similar ordinances have been enacted by Bowling Green, Kentucky[21], and Springfield and Perryville, Missouri.

Clinton Township, New Jersey, and Macungie Township, Pennsylvania, both in the Appalachians, in a effort primarily to control sinkhole development but also to protect ground water quality in the karst aquifers that underlay them, have each enacted ordinances in 1988 that regulate but do not prohibit construction in order to lessen the probability of sinkhole collapse.[22] The legality and validity of the New Jersey ordinance has been upheld in the courts, and its operation has been successful.[23]

Local land use controls can be aided by federal programs found in the Safe Drinking Water Act. The Wellhead Protection Program, included in the 1986 amendments to the Act, allows states to develop and submit to EPA programs that protect areas around public water supply wells (Wellhead Protection Areas). These Wellhead Protection Programs often provide a mix of state and local source controls with local land use controls.[24]

COMMON LAW PROTECTIONS

Ground-water users can rely upon principles of water law developed for surface water to protect their water rights. The principle of reasonable use, where applicable, requires each property owner to make only reasonable use of the ground water so as not to interfere with use of ground water by other property owners. An unreasonable use can be enjoined by the courts.

Under English common law, water under the ground was generally considered the exclusive property of the surface owner. Thus, the surface owner could make whatever use of the ground water he or she wished, including uses destructive of ground water quality and quantity. In the United States state courts have gradually recognized the fact that ground water flows and that one landowner's use of ground water can affect other landowners. American courts have adopted the reasonable use rule, which holds that one user may only put ground water to reasonable use where it may affect the use of ground water by another user.

The first step in that recognition, and a distinction which still holds in some states, was to apply the reasonable use rule to water that flowed underground in a well-defined channel, as opposed to percolating ground water. Since water flowing in a well-defined channel is similar to surface water, the reasonable use rule, which was developed for surface waters, applied. Such conduit flow, of course, is an attribute of karst terranes.

Most states now apply the reasonable use rule to either conduit flow situations or percolating ground water where there is proof that one property owner's use has unreasonably affected the ground water use of another. For instance, the Tennessee Court of Appeals found in 1935 that the rights of a property owner to enjoin the unreasonable use of ground water that supplied a spring did not depend on proof of a well-defined channel, but simply of an "intimate" connection between the ground water on the two properties.[25]

But in states where the well-defined channel distinction is still important, knowledge of the subsurface conduit flow feature is critical to the protection of the ground water. Texas, for instance, still follows the English rule for percolating ground water, and there is a presumption that all ground water is percolating ground water. In a recent case, the Texas Court of

Appeals found the hydrogeological testimony insufficient to establish well-defined channel flow of a spring flowing through an underground cavity before it surfaced. This ruling permitted the property owner to utilize all of the flow of the spring for irrigation purposes.[26]

DISCUSSION AND CONCLUSIONS

There are ample legal tools available for the protection of ground water by state and local governments and by individual ground-water users. Most statutes and regulations, however, ignore karst aquifers because the framers never thought about them or had no knowledge of how they differ from other aquifers. When these legal tools are used for karst terranes, they need to be sharpened to address the particular features of karst.

There are three major problems with many existing statutes and regulations that specifically address the regulation of activities in karst terranes:

1. The definitions of karst are inadequate or ambiguous, particularly when they focus solely on the presence of sinkholes.[27]

2. The definition of karst, even when adequate, is often misinterpreted to apply only to areas with sinkholes.

3. There is too much emphasis on the surface of the karst and too little on the aquifer itself. Regulatory buffer zones of a few hundred feet radius around sinkholes, for instance, ignore the fact that, in most terranes, the greatest danger to ground water is migration of contaminants below the surface, not collapse of the surface.

Source-control regulations addressing karst terranes, including siting restrictions for potentially polluting activities, should focus on more than just the potential for surface collapse in karst terranes. Because of the extreme difficulties in preventing and detecting ground water contamination in karst, there should be a presumption against the location of certain activities, such as hazardous and solid waste landfills, in karst terranes. And before that presumption can be overcome, there must be a demonstration that contaminants would not enter water supplies and that reliable monitoring can be achieved as a result of detailed knowledge of ground-water flow paths.

Land-use controls must also recognize the unpredictable nature of ground-water flow in karst terranes. Arbitrarily drawn recharge zones for wells and springs may not only be unfair to property owners, but may miss conduit-flow pathways entirely. Again, detailed knowledge of ground-water flow is necessary.

Finally, water rights in ground water may be asserted by individual ground-water users. The extent of such rights, however, may actually depend on the ability of the user to prove that the ground water is flowing as a discreet underground stream.

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2. Safe Drinking Water Act, 42 United States Code Sections 300f-j; Resource Conservation and Recovery Act, 42 United States Code Sections 6901, et seq.; Comprehensive Environmental Response Compensation, and Liability Act, 42 United States Code Sections 9601, et seq.; Toxic Substances Control Act, 15 United States Code Sections 2601, et seq.; Federal Insecticide, Fungicide and Rodenticide Act, 7 United States Code Sections 136, et seq.; Uranium Mill Tailings Radiation Control Act, 42 United States Code Sections 7901, et seq.; Surface Mining Control and Reclamation Act, 30 United States Code Sections 1201, et seq.; Atomic Energy Act of 1954, 42 United States Code Sections 2014, 2021, 2021a, 2022, 2111, 2113, 2114.

3. 56 Federal Register 51019-20 (Oct. 9, 1991). "Karst terranes" are defined as "areas where karst topography, with its characteristic surface and subterranean features, is developed as the result of dissolution of limestone, dolomite, or other soluble rock. Characteristic physiographic features present in karst terranes include, but are not limited to, sinkholes, sinking streams, caves, large springs, and blind valleys." This definition is repeated on page 51047, but there "karst terranes" are twice mistakenly referred to as "karst terraces".

4. Minnesota Rules, Section 7035.2815, Subpart 2A. (1988).

5. Minnesota Rules, Section 7035.2815, Subpart 2C. (1988). Under Minnesota Rules, Section 7035.0300, Subpart 51, "karst" "means a type of topography that is formed from the dissolution of limestone, dolomite, or gypsum and that is characterized by closed depressions or sinkholes, and underground drainage through conduits enlarged by dissolution."

6. Kentucky Administrative Regulations, 401 KAR .. Under 401 KAR 30:010(104) "karst terrain" "means a type of topography where limestone, dolomite or gypsum is present and is characterized by naturally occurring closed topographic depressions or sinkholes, caves, disrupted surface drainage, and well developed underground solution channels formed by dissolution of these rocks by water moving underground."

7. Kentucky Administrative Regulations, 401 KAR 48:050, Sections 6 and 1, respectively.

8. Tennessee Administrative Code, Rule 1200-1-7-.04(2)(q). Under Rule 1200-1-7-.01(2) "karst" "means a specific type of topography that is formed by dissolving or solution of carbonate formations, such as limestone or dolomite; it is characterized by closed depressions or sinkholes, caves, sinking and reappearing streams, and/or underground conduit drainage flow."

9. Rule 1200-1-7-.04(2)(q) goes on to state that "the above-referenced demonstration may require the installation of piezometers, the developing of potentiometric-surface map of ground water, conducting geophysical surveys, dye tracing or other specific requirements deemed necessary by the Commissioner to evaluate the proposed site to his satisfaction." In our experience with solid waste landfill siting in Tennessee, the focus of the Division of Solid Waste has been more on the detection of potential for surface collapse through geophysical studies, and no dye trace studies have been required, despite clear indications of highly developed karst. Those traces that have been performed in an attempt to demonstrate the threat to karst aquifers have been ignored or misinterpreted by the Division.

10. Alabama Administrative Code, Sections 335-13-4-.01(6), 335-13-4-.11 through 335-13-4-.14. (as amended through October 2, 1990).

11. Florida Administrative Code, Section 17-701.040(2) (as amended through July 19, 1990); Rules and Regulations of the State of Georgia, Section 391-3-4-.05(g)5. (as amended through June 29, 1989).

12. Scott W. Horsley, Beyond Zoning: Municipal Ordinances to Protect Groundwater, Cape Code Planning and Economic Development Commission, Barnstable, Mass. (1982).

13. For example, Tennessee Code Annotated Sections 69-7-101, _et seq._ provide for the creation of watershed districts.

14. Tennessee Code Annotated Section 69-7-118.

15. Tennessee Code Annotated Sections 7-82-302, 7-82-306.

16. Tennessee counties may be pass zoning ordinances for the conservation of water supply. Tennessee Code Annotated Section 13-7-101.

17. See Douglas Yanggen and Stephen Born, Protecting Groundwater Quality by Managing Local Land Use, Journal of Soil and Water Conservation, Vol. 45, No. 2, pp. 207-210 (March-April 1990); F. DiNovo and M. Jaffe, Local Groundwater Protection--Midwest Region, American Planning Association, Chicago, Il. (1984).

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23. See Joseph A. Fisher and Robert J. Canace, Karst Geology and Ground-Water Protection Law, Hydrogeology, Ecology, Monitoring, and Management of Karst Terranes Conference (3rd, Nashville, Tenn.), Proceedings, National Ground Water Association, Dublin, Ohio. (1991) [in this volume].

24. 42 United States Code Section 300h-7.

25. Nashville, C. & St. L. Ry. v. Rickert, 89 S.W.2d 889 (Tenn.Ct.App. 1935), cert. denied (Tenn. 1936).

26. Denis v. Kickapoo Land Co., 771 S.W.2d 235 (Tex.Ct.App. 1989).

27. See James F. Quinlan, P.L. Smart, G.M. Schindel, E.C. Alexander, Jr., A.J. Edwards, and A.R. Smith. Recommended Administrative/Regulatory Definitions of Karst Aquifer, Principles for Classification of Carbonate Aquifers, and Practical Evaluation of Vulnerability of Karst Aquifers. Hydrogeology, Ecology, Monitoring, and Management of Ground Water in Karst Terranes Conference (3rd, Nashville, Tenn.), Proceedings. National Ground Water Association, Dublin, Ohio (1991). [in this volume]. This paper proposed the following definitions:

    karst - a landscape and subsurface formed as a result of dissolution of bedrock (usually, but not necessarily, limestone and/or dolomite) and characterized by distinctive subsurface hydrology that includes flow of water through caves. The landscape may include sinkholes, springs, and sinking streams, but these features are not essential to the definition and should not be confused with the karst itself.

    karst aquifer - an aquifer in which flow of water is or can be appreciable through one or more of the following: joints, faults, bedding planes and cavities, any or all of which have been enlarged by dissolution of bedrock.

    Each of these definitions is suitable for incorporation in statutes, regulations, and revisions thereof. We recommend their use.

BIOGRAPHICAL SKETCHES

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Session IX:

Ground Water Management

KARST GEOLOGY AND GROUND WATER PROTECTION LAW

By: Joseph A. Fischer, Geoscience Services, and
Robert J. Canace and
Donald H. Monteverde, NJ Geological Survey, Dept. of Environmental Protection & Energy

ABSTRACT

Clinton Township, Hunterdon County, New Jersey is situated partially in a Paleozoic outlier within the New Jersey Highlands Physiographic Province. Some 15% to 20% of the Township is underlain by solution-prone, folded and faulted carbonate rocks of Cambrian and Ordovician age. Faulting of possible Taconic, Alleghanian, and Triassic age has deformed these carbonates. The Flemington Fault, part of a major border fault system, traverses the Township.

Clinton is located at the intersection of one of New Jersey's principal east-west interstate highways and a major north-south State highway. Its rural environment and ready access to major highways places Clinton in a growth corridor. The State Planning Commission has recently identified the Township as a regional rural growth center.

Two state reservoirs extend onto Township lands, one underlain by carbonate bedrock. The construction of the Spruce Run Reservoir provided a great amount of geologic information on the local carbonate deposits, including the nature and extent of folding, faulting, and solution-channel development. Extensive mapping and exploratory drilling permitted the compilation of the occurrence of voids in specific geologic units, and a better definition of the relationship between solutioning and faults.

Enlightened Township officials introduced an ordinance governing construction in areas underlain by carbonate rock and those areas draining into carbonate lands. The Ordinance has been in effect since the spring of 1988. Although originally met with great protestation by development interests, it has proven to be a rational, working process that is presently accepted by both the public and private sector.

The ordinance requires a multi-phased data collection effort at various stages of a project. The legality and validity of this Ordinance have been tested and upheld in the court room.

INTRODUCTION

New Jersey is well known for its number of Super Fund sites and its rather stringent State regulations in relation to discharges into ground water and surface water bodies, as well as through injection wells. However, the only State regulations that give consideration to the problems of siting a facility in karst areas are provisos not to place septic systems in sinkholes, and to establish trend directions of solution channels in carbonate rocks and sinkholes for landfill sites.

There are five counties in the northwestern corner of the state that are underlain by major deposits of solutioned carbonates (Figure 1). However, there are no county regulations that address karst related concerns and the municipal officials of these counties have little guidance in either recognizing or dealing with karst-related concerns. In addition, the potential for ground water pollution from agricultural activities has also been neglected.

At one time, these valley lands were primarily either agricultural in nature or merely bucolic scenes. However, recent population growth in these areas is rapidly changing the use of this prime real estate. Ground water extracted from the high yield carbonate aquifers is the primary source of domestic and industrial water in this region. Reservoirs, water treatment facilities, shopping centers, office buildings, and corporate centers have been built above a variety of carbonate terrane, sometimes supporting the structures atop large (now, usually cement-filled) cavities. Some have been well-investigated and engineered while others have not.

The one municipality that has recognized these problems is Clinton Township, a State targeted growth area in Hunterdon County, New Jersey. Clinton Township is located at the intersection of a principal east-west interstate highway and a major north-south State highway, less than 50 miles from New York City. Some 10 years ago it was a rural, primarily agricultural community. At the completion of the interstate highway, the area became the target of both State and private development planning. The Township's complex geology (Figure 2), includes solutioned carbonates under some 20% of its' lands. Currently, wells are the primary source of domestic and industrial water supply for the Township. Development pressures, led some perceptive municipal officials to pass New Jersey's first "limestone" Ordinance to protect the ground water supply in this environmentally sensitive setting.

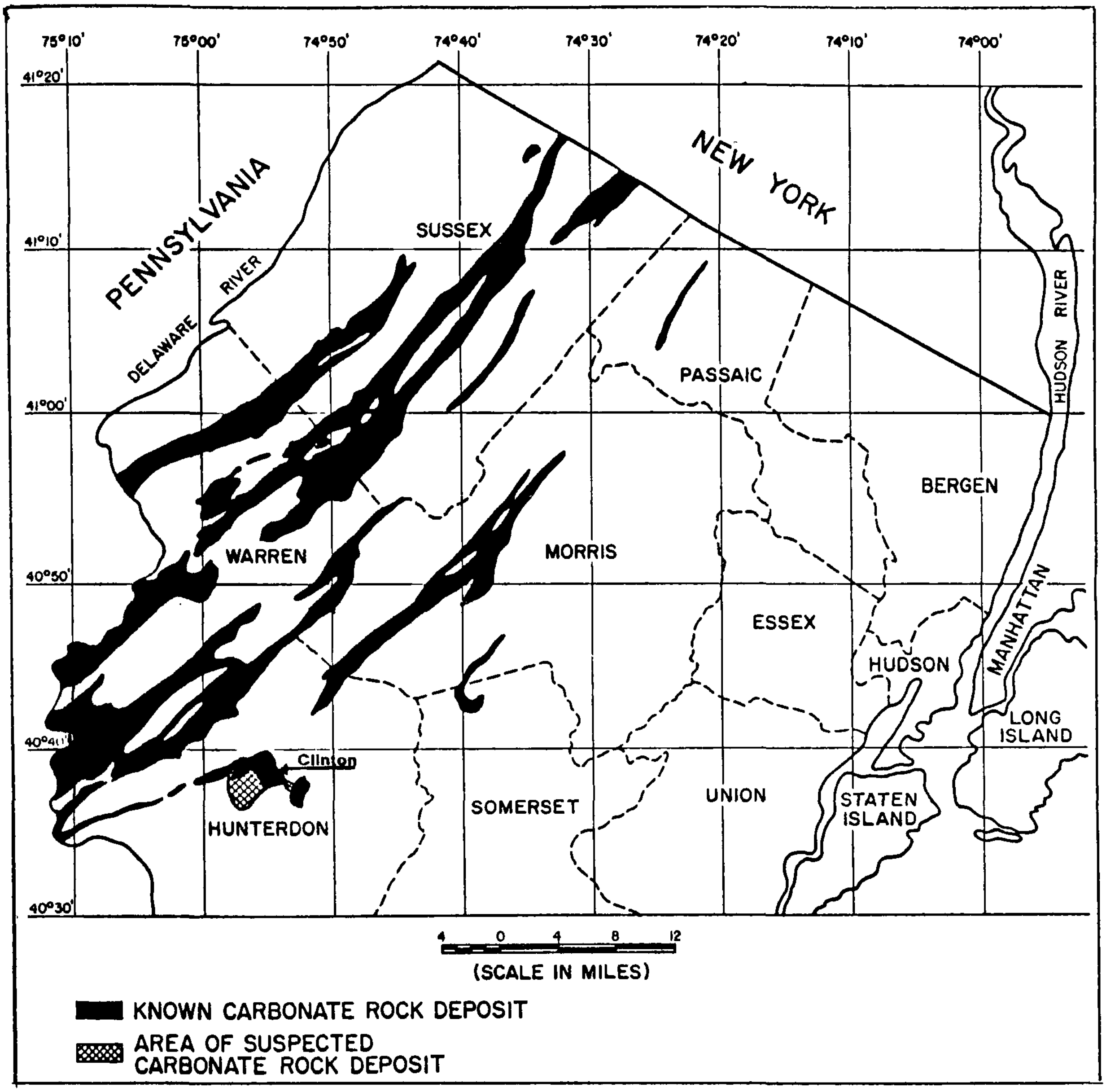

Figure 1. Carbonate rock deposits of northern New Jersey.

The mayor, an articulate, aware, and strong individual, led the struggle to secure a rational approach to the Township's development processes. The New Jersey Geological Survey encouraged the ordinance and provided technical guidance and a simplified geologic map identifying the general location of carbonate bedrock. Previous State funded geologic information provided a rational data base for the Ordinance and its implementation. Problem recognition, conceptual planning, and Ordinance formulation and implementation was a relatively swift process.

In brief, the Township, faced with increasing development pressures in 1987, placed a 150 day moratorium on development and formed a committee of lay and technical people to draft an ordinance that would:

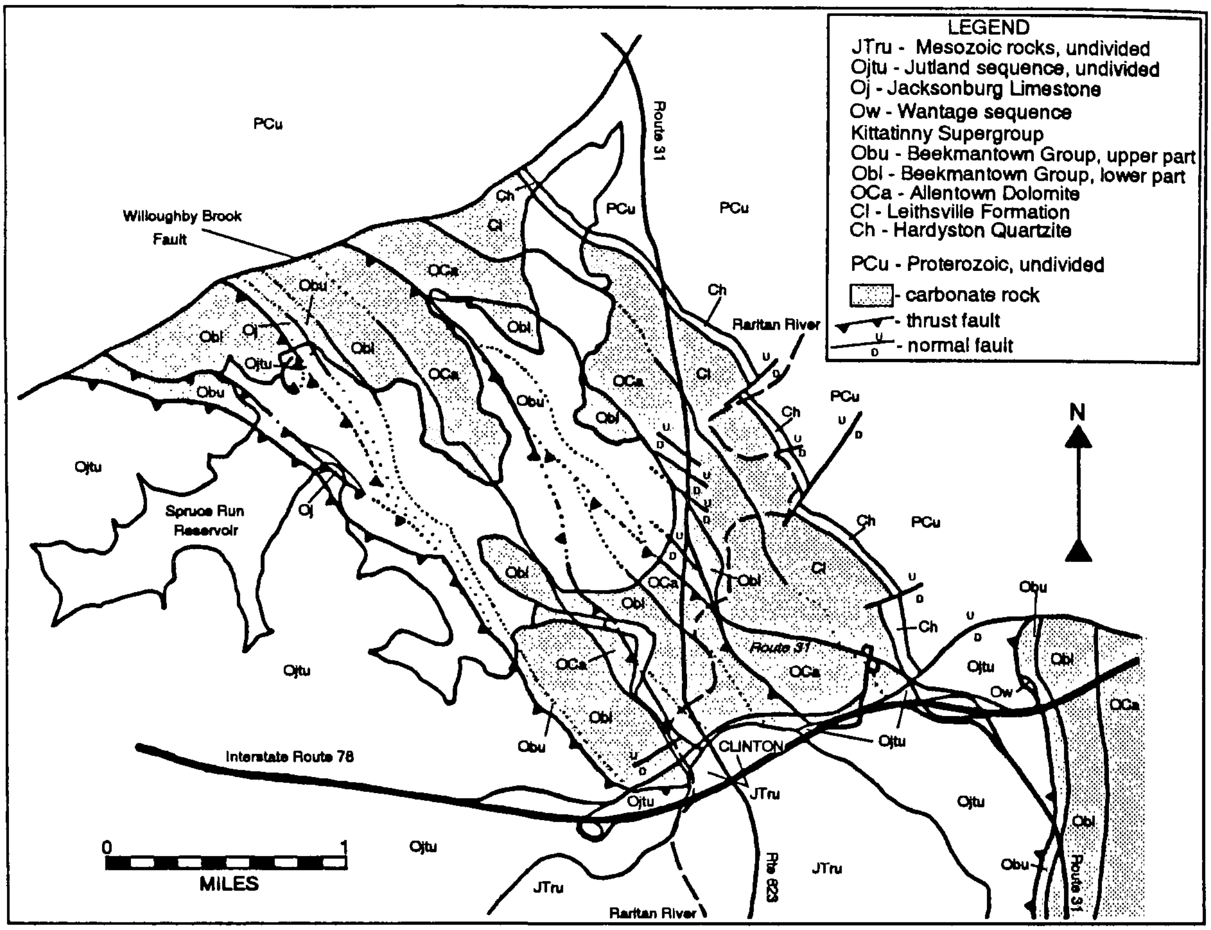

Figure 2. Geology of Spruce Run Reservoir and adjacent areas of Clinton Township, Hunterdon County, New Jersey.

1. Protect ground water in the carbonate rock areas.

2. Address public safety in relation to structural integrity.

3. Allow conscientious development.

The Ordinance was passed by the Town Council (although not unanimously) in May, 1988. Since that time, some 30 projects have been reviewed by the Township's geotechnical consultant (GTC) under the provisions of the Ordinance. The results of the review process have included the abandonment of projects in the exploration phase (even prior to the completion of the field explorations), the completion of the project as envisioned by the designers (with little to no changes), and the completion of projects that incorporated numerous changes from the initial design, some during construction.

The Ordinance has been tested in court by one of the developers affected by the moratorium. In a broad-based suit against the Township, the Ordinance was subject to legal scrutiny, was upheld, and found to be a most valuable document for achieving the Township's desire to insure that development is conducted intelligently.

The Township was fortunate in that geologic investigations both before and during the construction of two large State-funded reservoirs provided an extensive geologic data base as well as fueling public awareness of karst concerns. It is difficult to visualize the Ordinance coming into being, and its successful implementation, without the State supplied knowledge gained through the reservoir studies and the ongoing interest in carbonate geology of the State Survey.

TOWNSHIP GEOLOGY

The carbonate bedrock in Clinton Township consists of rocks of the Kittatinny Supergroup (Drake and Lyttle, 1980) of Cambrian through Middle Ordovician age, the Middle Ordovician Jacksonsburg Limestone (Kummel, 1940), and the Lower to Middle Ordovician Jutland sequence (Jutland Member of the Martinsburg Shale of Markewicz, 1967). The geology of the Clinton area was mapped by Markewicz (1967), and is currently being revised by Markewicz, Monteverde, and Volkert of the New Jersey Geological Survey. The geologic map provided to Clinton Township by the New Jersey Geological Survey, which has been updated in Figure 2, is a simplified version of the current geologic interpretation of the area. Data derived as a result of geotechnical investigations carried out under the ordinance has resulted in some modifications to the initial mapping.

As indicated above, geologic findings associated with mapping and subsurface exploration for the proposed Spruce Run Reservoir (Markewicz 1958-1961) provided insight into the degree of karstification of the carbonates in Clinton. Geologic mapping, field identification of sinkholes, seismic data, and numerous test borings revealed the presence of voids both at the surface and in the subsurface, indicating the potential for future subsidence. The reservoir was constructed after a program to grout any voids at dam locations and the installation of a peripheral grout curtain to prevent leakage at the reservoir's southeastern margin. Since construction, several sinkholes have opened adjacent to the reservoir, requiring remediation.

The degree of karstification of the various carbonate bedrock units found in Clinton, as elsewhere, is a function of lithology and proximity to faults, folds and other geologic structures. While most carbonates have a potential for karstification, regional studies suggest that geologic units that contain coarse-grained dolomite or fine-grained limestone are most susceptible to weathering (Rauch and White, 1970, Siddiqui and Parizek, 1969, and Dalton and Markewicz, 1972). Solution-channel development in the Kittatinny carbonates appears to be particularly accentuated in units that contain thin-bedded, coarse-grained dolomite.

Table 1 presents a summary of exploratory drilling data for the Spruce Run Reservoir project. Although the number of drill holes installed and total penetration varied among the geologic units present, the data tends to confirm that specific units may be more prone to cavity development than others. Figure 3, based on this drill data, presents a visual summary

Drill Data	Geologic Unit				
	Leithsville	Allentown		Lower Beekmantown	Upper Beekmantown
		Limeport Member	Upper Allentown Member		
Number of Borings	9	32	33	31	66
Total Footage Drilled	1,035	4,574	2,770	3,852	6,428
Number of Cavities	13	108	19	80	90
Average Footage Between Cavities	80	54(*)		48	71
Total Footage of void Penetrated	67	319	66	225	263
Voids as Percentage of Total Depth Penetrated	6.5	7.0	2.4	5.8	4.1
Voids as a Percentage of Bedrock Penetrated	9.4	8.3	2.7	6.6	4.6

Table 1. Occurrence of voids in borings for Spruce Run Reservoir (Dalton and Markewicz, 1972, Table 4).
*** - Indicates value for formation as a whole.**

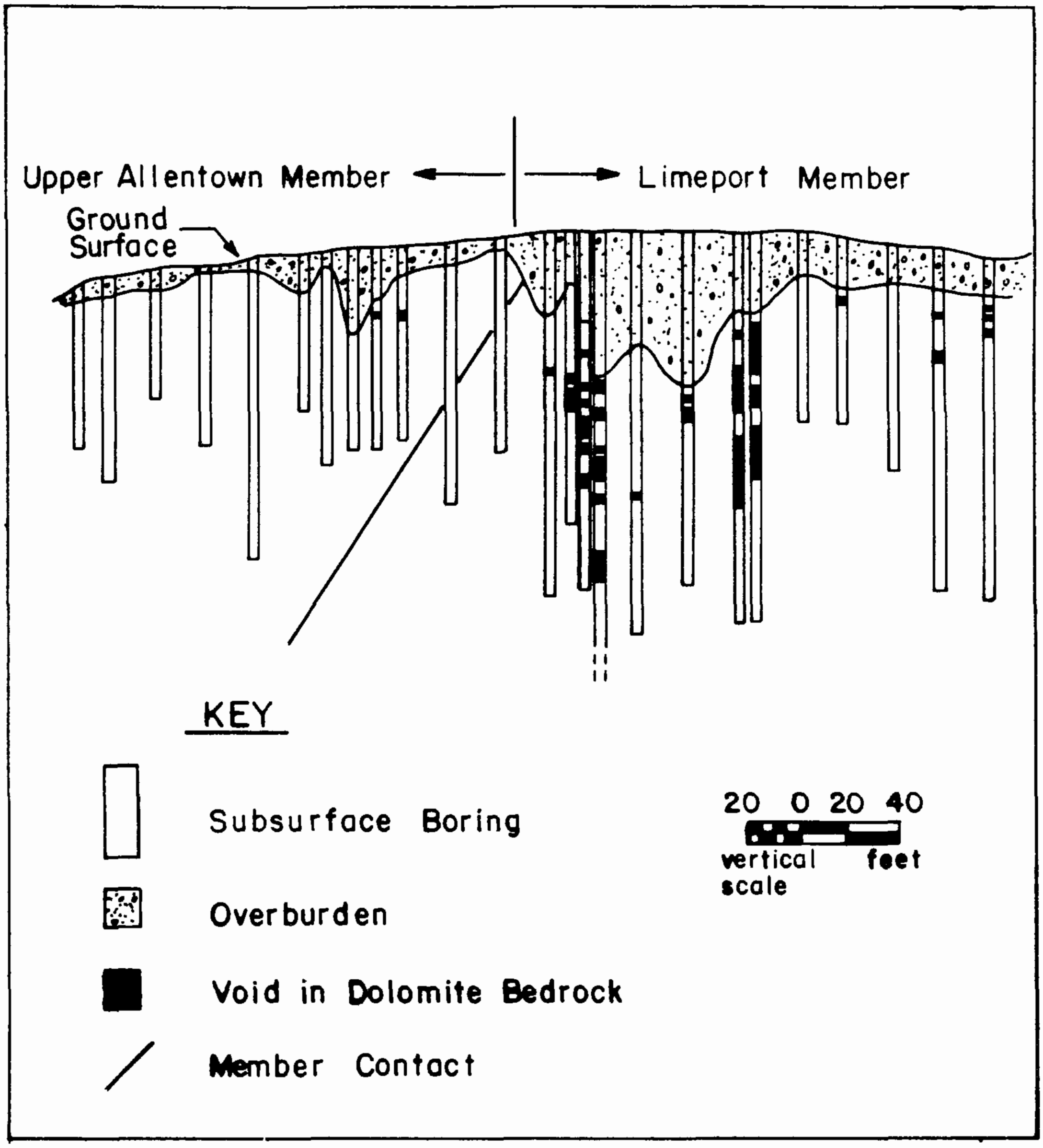

Figure 3. Subsurface profile of a portion of the Spruce Run Reservoir dam site, showing the contrast in intensity of weathering between the Limeport and Upper Allentown Members of the Allentown Formation (Markewicz, 1958-1961).

of the contrast in the occurrence of voids between the lower (Limeport) member and the upper (Upper Allentown) member of the Allentown Dolomite.

Another example of the presence of concentrated solutioning in the Spruce Run area is illustrated by borings drilled into the lower Beekmantown Group of carbonates in which 6 borings were installed within 45 feet of one another, to depths ranging from 72 to 142 feet. These borings revealed voids in the bedrock occupying between 14 and 52 percent of the total footage drilled for any one of the borings. In addition, 100 percent of the voids in this interval occur within the first 45 feet of rock penetrated, signifying that many near-surface voids are present that could contribute to the formation of sinkholes at the surface.

Faults involving the carbonate units consist of imbricate thrusts, tear faults transverse to the structural grain, and normal faults most likely associated with both rifting in the adjacent Newark rift basin and with earlier Appalachian deformation. The reservoir borings showed that cavities are often well developed in the vicinity of faults (Figure 4) that display brittle deformation. Fault locations were interpreted from mapping of the area and rock core displaying fracturing, recrystallization and other evidence of deformation.

An unusual occurrence of limestone is found in the Clinton area, in rocks of the Jutland sequence (Monteverde, unpublished). A test well drilled for Clinton, into this unit, in proximity to a major east-west fault, yielded nearly 400 gallons of water per minute, attesting to the well-developed permeability of this limestone. Additionally, test borings performed in the Jutland limestone for a proposed and subsequently constructed golf course/residential community revealed the persistent presence of small voids and weathered zones throughout the limestone section.

NATURE OF THE ORDINANCE

There are a number of ordinances that have been passed for locales underlain by the solutioned carbonates found in the Appalachian valleys of the eastern United States. These include ordinances which essentially require the preservation of present surface water flows into existing sinkholes, or those that establish performance standards in relation to known or suspected features. From the authors' reviews of various eastern U.S. "limestone" ordinances, many different tacks have been taken. In the valleys of the southern Appalachians, where development pressures are less than in the more northerly valleys, it seems that the primary direction of all ordinances has been toward preserving sinkhole integrity as a means of handling storm water flows without deleterious local flooding (e.g., Clarke County, Virginia). The secondary consideration seems to be to stop structures from being built directly over existing sinkholes.

In the northeast, some Townships have apparently tried to essentially eliminate growth in carbonate areas. For example, the Bucks County, Pennsylvania "Model Ordinance" requires that storm water management basins and "principal or accessory" buildings will not be located within 100 feet of features, which include; sinkholes, closed depressions, lineaments, fracture traces, caverns, ghost lakes, and disappearing streams.

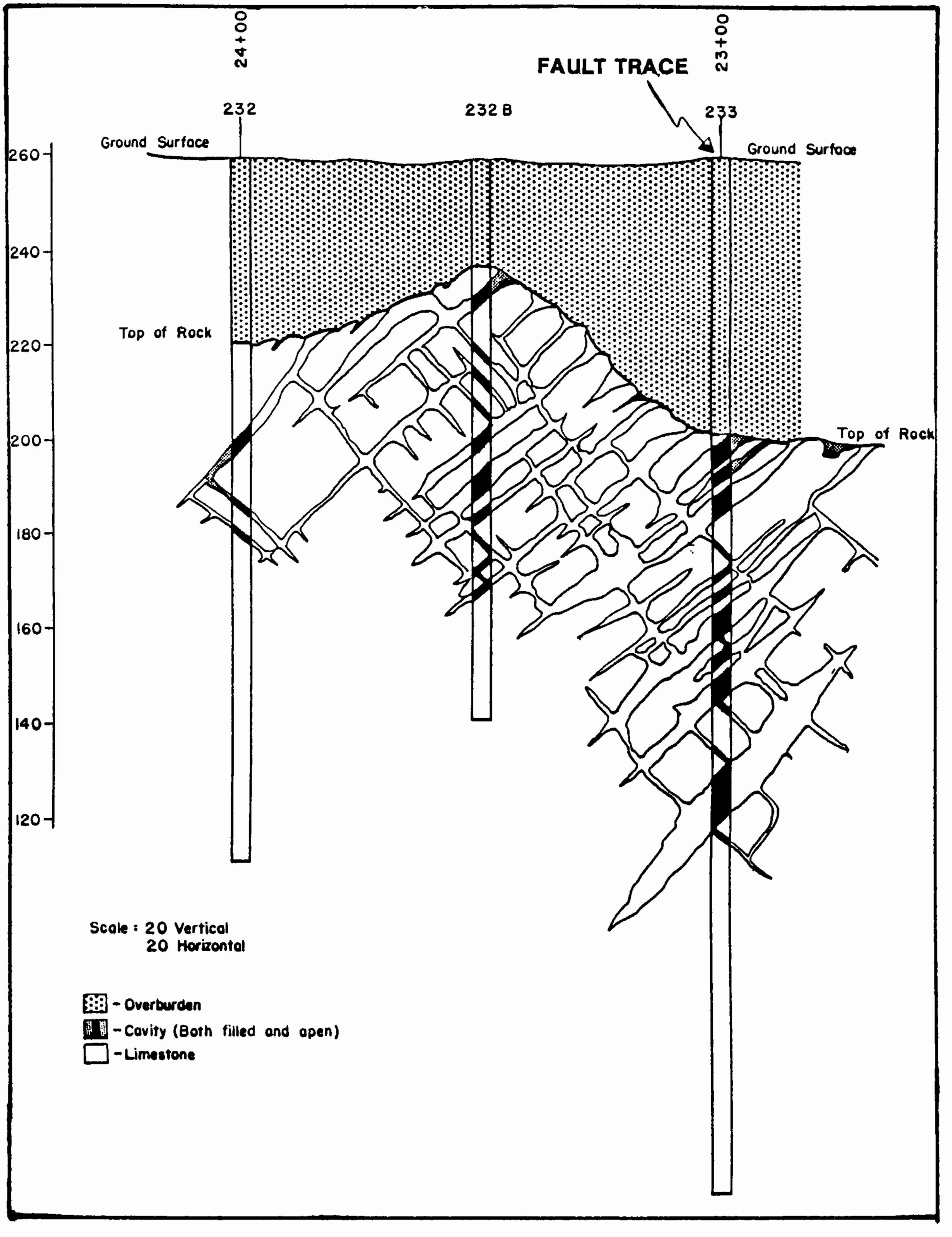

Figure 4. Schematic interpretation based on drill data of the occurrence of interconnected voids in the Allentown Dolomite in relation to the trace of a transverse fault.

It is obvious what an imaginative aerial photograph interpreter could do to any project in a municipality which has passed an ordinance in accordance with the Bucks County recommendations. In addition, the authors have little understand ing of what the magic is in the 100 foot dimension.

The route chosen by the Clinton Ordinance Committee was completely different and is still believed to be unique. The Ordinance was intended to help educate and provide guidance to a Planning Board applicant concerning the performance of a phased investigation and evaluation of the subsurface conditions at the development site. The investigation and evaluation process was derived from the concepts discussed in Fischer, et al (1987), and were developed from experience gained in numerous site investigations and facilities construction in karst.

A phased approach to the Ordinance requirements was instituted to allow an applicant to define site-related flaws and/or prepare preliminary cost estimates at an early stage of the development without incurring significant economic penalties. Judgement is used in defining the extent and nature of an investigation. Different concepts can be used for a proposed two-house subdivision than might be employed for a 59 acre, multi-use development.

The Ordinance encourages the applicant's geotechnical consultant to utilize any combination of investigative tools and techniques that are deemed reasonable, but with certain minimum requirements such as; reviewing the available geologic information, using aerial photography, and providing hard data (i.e., borings, test pits, etc.) on the site subsurface. The GTC's experience within the Township was made available to the applicant on an as-requested basis. The results of the explorations are submitted to the GTC for review and approval. Thus, a reasonable degree of consistency in the explorations performed by the various applicants has developed and any useful information produced at one site can be fed into future planning processes. The process also provides for the applicant to perform only the amount of work necessary for the actual site conditions, i.e., the more complex the site is, the more detailed the exploration and evaluation is.

The subsurface data obtained is interpreted by the applicant's geotechnical consultant and used to develop planning and design solutions to the geotechnical problems that exist at the site. Again, the GTC reviews and evaluates the subsurface model proposed for the site and suggested solutions to any subsurface-created concerns. The applicant is encouraged to use innovative engineering solutions and, in general, no set procedures are imposed upon the planning/design team. Final reviews and discussions are held with the Planning Board, the applicant's design team, and the GTC prior to giving conceptual approval to, or conversely, rejection of, the planned development.

As a result of the difficulty in performing a subsurface investigation and reliably interpreting the results for a karst site in any reasonably economic manner, the Ordinance provides for field inspection during construction in key areas by the applicant's geotechnical consultant as well as spot checks by Township forces. In enforcing the Ordinance, it is expected that changes in design and construction procedures can and will be made during the building phase.

We believe that the investigation, evaluation, and reporting costs required as a result of the existence of the Ordinance are reasonably consistent with what a prudent developer would ordinarily budget. This is not surprising as the Ordinance procedures are predicated upon concepts developed from commercial, not research projects. The separate cost of the GTC review are prepaid into an escrow account. The costs were estimated at $1,000 plus an additional $500/acre. The GTC review costs have usually been less than these initial requirements and only in one instance did the somewhat insensitive actions of an applicant require replenishment of the escrow fund.

EXAMPLES

Several examples of the different manners used to attack the issues can be used to illustrate some of the workings of the Clinton Township "Limestone" Ordinance.

<u>Case I</u> - The site was first investigated by a large national consulting firm during the building moratorium. Initially, no geotechnical study was envisioned. The firm (with the assistance of an outside geologic consultant) provided a "Geotechnical Feasibility Study" after the development complex plans for the 46 acre site had been fully formulated. The problems of force-fitting the investigative results to an existing layout became apparent. The existence of several sinkholes were minimized and none (including one elongated elliptical sinkhole some 25 feet in length) were considered "major" by the developer's consultant. A test pit excavated for the field investigation became a sinkhole. Faults crossing the site and their likely effect on solutioning were ignored. Only the more favorable portions of the work performed for the nearby reservoir were included. The most critical structures (service buildings and office towers) were located atop what was likely the most solutioned rocks at the site, the Leithsville Formation. An unlined, ornamental lake was sited over a line of small sinkholes. The underlying dolomites were impermeable in one section of the report but provided a good ground water resource in another. Rock Quality Designation (RQD) data was misapplied. The significance of drilling water, core, and sample losses were not understood.

These and other factors eventually resulted in an agreement to reconfigure the site and perform additional studies in conformance with the Ordinance. No work has been accomplished at this site since the agreement was made in 1990. As the business climate has been poor since that time, there is no way of telling whether the owner's eventual realization of the site subsurface deficiencies or the business climate forced the current abandonment of development plans.

Case II - Some 10 years ago a large international organization constructed a major research facility upon a small area of a roughly 100 acre parcel. A significant portion of the property is underlain by solutioned carbonates, but apparently the karst-related problems were not well recognized prior to, or even during, the site investigation. Two large national firms provided geotechnical consulting services. Cost overruns in the millions of dollars were experienced in both investigation and foundation construction. Untold quantities of cement were pumped into unexpected cavities and a complicated foundation system was installed. A spray irrigation field for waste water disposal was placed above shallow, solutioned limestone. Sinkholes have developed in this field.

The remainder of the site is currently planned for multi-use development by single and multi-family homes, offices, stores, open-space and recreational facilities. A two-phased field investigation has recently been completed by another well-known engineering firm. The work was planned and accomplished in complete cooperation with the Township GTC. Personnel experienced in carbonate rock studies were used both in the field portion of the work and in the report preparation. The owners design team is currently using the results of the phased explorations investigations, which cover site areas that are underlain by solutioned carbonates, competent metamorphics, and apparently fractured and faulted sediments, to prepare a conceptual layout. The information provided to the design team has blended the experience of both the consulting firm and the Township GTC.

Case III - A caveat emptor situation developed in a proposed single-family development that was investigated under the requirements of the Ordinance. The applicant's geotechnical consultant and the GTC came to an agreement on layout and design of structures, wells, and septic systems that incorporated the results of the applicant's studies. The revised conceptual plans were approved by the Planning Board, with the standard caveat of the need to provide knowledgeable construction inspection.

The site was sold to a new developer. The approved plans were provided to the buyer but apparently without the several geotechnical reports completed by the original applicant. Site work was initiated, then abandoned by the new owner after the first foundation wall failed as the result of the opening of a post-construction sinkhole beneath it.

CONCLUSION

The many fears in relation to ground water protection that both the lay and technical person should have, or in many instances do have, can only be exacerbated by the presence of solutioned carbonate rocks below the site development. The technical information needed to address concerns related to karst is available to the interested parties in any development exercise. However, the awareness of the need for technical expertise is certainly not wide-spread. Some communities have undertaken to express their concerns with ordinances.

Drill data, the mapping of caves, and field observations concerning the distribution of sinkholes, springs, and disappearing streams all indicate that certain units within the Kittatinny Supergroup display a higher degree of karst development than other carbonate rocks. Some units are particularly prone to solutioning. This concentration of karst features in specific geologic units was confirmed by the borings installed for the Spruce Run Reservoir in Clinton Township. Therefore, in planning and carrying out site investigations the geotechnical expert should keep in mind the likelihood and potential size of karst features as a function of the geologic unit(s) found on the site. An understanding of the probable severity of karstification at a development site can help direct the intensity of the exploratory program necessary for a suitable site investigation.

Despite the correlations that probably can be made between specific geologic units and structures and the potential for karst features, a decision was made to treat all carbonate bedrock in the township equally when enforcing the ordinance. This decision was made because of the ultimately unpredictable nature of subsurface voids and for the sake of equal treatment under the law. This approach is valid in view of the need to investigate for voids on a site-specific basis, where an understanding of the location of structures relative to voids is vital.

The authors believe a Clinton Township type "limestone" Ordinance which draws upon the available geologic information to establish guidelines, yet allows development to occur in karst areas is an appropriate solution in most instances. Laws should not be aimed at eliminating development or putting the developer into a strait jacket, but should incorporate appropriate provisions for insuring the protection of ground water resources and the safety of the public while allowing for ingenuity in design and construction.

There are, undoubtedly, economic penalties that result from the need to obtain adequate geologic information at a site and to provide appropriate engineering solutions. However, these economic penalties are relatively small in contrast to the penalties that can occur as a result of failures of both the infrastructure and any structures located atop solutioned carbonates, during or after construction when likely, but unforeseen problems surface.

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KARST GEOLOGY AND GROUND WATER PROTECTION LAW

By: Joseph A. Fischer, Robert J. Canace,
and Donald H. Monteverde

Question

1. What cities, counties, and states have ordinances near or similar to the one you describe?

None. We know of no State ordinances of this nature. We also unaware of any Counties or Cities with such an ordinance, although our knowledge on an interstate, local level is obviously limited. However, a Pennsylvania legislature's staff member informed Clinton Township that the Commonwealth was attempting to prepare a law and that the Township's ordinance appeared to be the best available based on what they had found.

2. To what extent are other governmental units deciding to emulate the ordinance you describe and adopt a "locally-suited" similar ordinance?

There are currently three to four New Jersey municipalities considering a similar ordinance. A model ordinance is being prepared by a "limestone committee" of the New Jersey Resource Conservation and Development organization (NJRC&D), the northern New Jersey branch of a nation-wide, county/USDA sponsored organization. Information concerning the NJRC&D model ordinance will be provided to numerous local municipal and private planning organizations in a series of spring (1992) meetings. The RC&D, nation-wide, will also be made aware of this and other NJRC&D work in relation to carbonate rock concerns. In addition, a number of inquiries from State and County level individuals, concerning the ordinance, were received by the authors both at the conference and subsequently.

ANALYSIS OF DRASTIC AND WELLHEAD PROTECTION METHODS APPLIED TO A KARST SETTING

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ABSTRACT

DRASTIC and Wellhead Protection methodologies were used in the analysis of an Inner Bluegrass Karst Region karst spring used as a municipal water supply. The area has been studied by Thrailkill and his students (Thrailkill et. al., 1982). Through dye tracing and field analysis, the area has been divided into groundwater basins and interbasin areas. Groundwater basins are characterized by numerous sinkholes, conduit flow and deep water table (33 meters). Interbasin areas are characterized by shallow water table, high level (upland) springs, fracture flow and sinkholes. Existing information was utilized in the DRASTIC analysis with inconsistent results due to the dichotomy of a deep water table found in the groundwater basins. Even though high values were used for most parameters, the deep water table reduced the index value, thus minimizing the potential impact.

The Wellhead Protection analysis and the delineation of a Wellhead Protection Area (WHPA) provides a more direct manner for the identification of the area that needs to be protected. The definition of the area based on Thrailkill's concept of a groundwater basin as defined by dye tracing provides a good planning tool. An important fact that must be dealt with in the planning phase is the 80 percent of the WHPA for the municipal water supply lies outside the jurisdiction of the community trying to protect its water supply. The adjacent community is currently addressing a planning issue that involves the WHPA because a major development is proposed for a part of the WHPA.

BACKGROUND

This analysis is based on data gathered over the last three years on contracts supported by the Divisions of Water and Waste Management of the Natural Resources and Environmental Protection Cabinet of Kentucky, the United States Environmental Protection Agency and the University of Kentucky's Institute for Mining and Minerals Research.

DRASTIC maps were developed for six areas in Kentucky to test the feasibility of producing such maps from data available in existing data bases, files and published maps and reports. Areas were chosen for each of the major physiographic regions of the state including cities whose public water supply depended on groundwater as all or part of the source. (Sendlein, 1989)

Wellhead protection work was conducted for three of the areas for which DRASTIC maps had been developed. This was a pilot study and because of the limited funds, complete analysis was not possible. In fact, the study was also based on available data and consisted of construction of the Wellhead Protection Area maps for each area and the compilation of sources of potential contamination from state and federal files. Each area included spot field inspection of the groundwater water sources and in one case a pumping test was attempted. (Sendlein and Fitzmaurice, 1990)

Kentucky is very fortunate because the entire state is covered by seven and a half minute USGS quadrangle maps including topography and geology. In the construction of the DRASTIC maps, the USGS maps were used as base maps, and each area studied included two adjacent sheets, except one of the areas was composed of four adjacent sheets. Because the geology is know at this level of detail, the construction of general environmental maps such as DRASTIC maps, does not add much information for an area that a qualified geologist could not glean from the geologic maps. However, the non-geologist lay person can benefit from the conclusions reached as a result of the DRASTIC analysis.

Considerable information is present in the files of the Divisions of Water and Waste Management in Frankfort, Kentucky. These data are scattered throughout the agency's various offices with permits kept in a central file room. Much of the tabular information is kept in computer files, but most maps are on paper. An exception to this is the map information kept in the Kentucky Natural Resources Information System (Croswell et al., 1982) utilizing the ARCINFO software. The scale of the information included in this database is on the order of 1:1,000,000, considerably larger than the 1:24,000 of the quadrangle maps used in the present studies. The DRASTIC maps were developed on the computers in Frankfort and are stored as files in their system. (Couch, 1988)

A recent water well law in Kentucky requires that all water wells drilled in the state be recorded and a driller's log be filed with the Division of Water. These logs, along with other bore hole information, are being compiled into a database by the Kentucky Geological Survey and will be an important database when it comes on line. Currently, the paper files can be reviewed in the Division of Water and a computer printout of some of the parameters are available from that office.

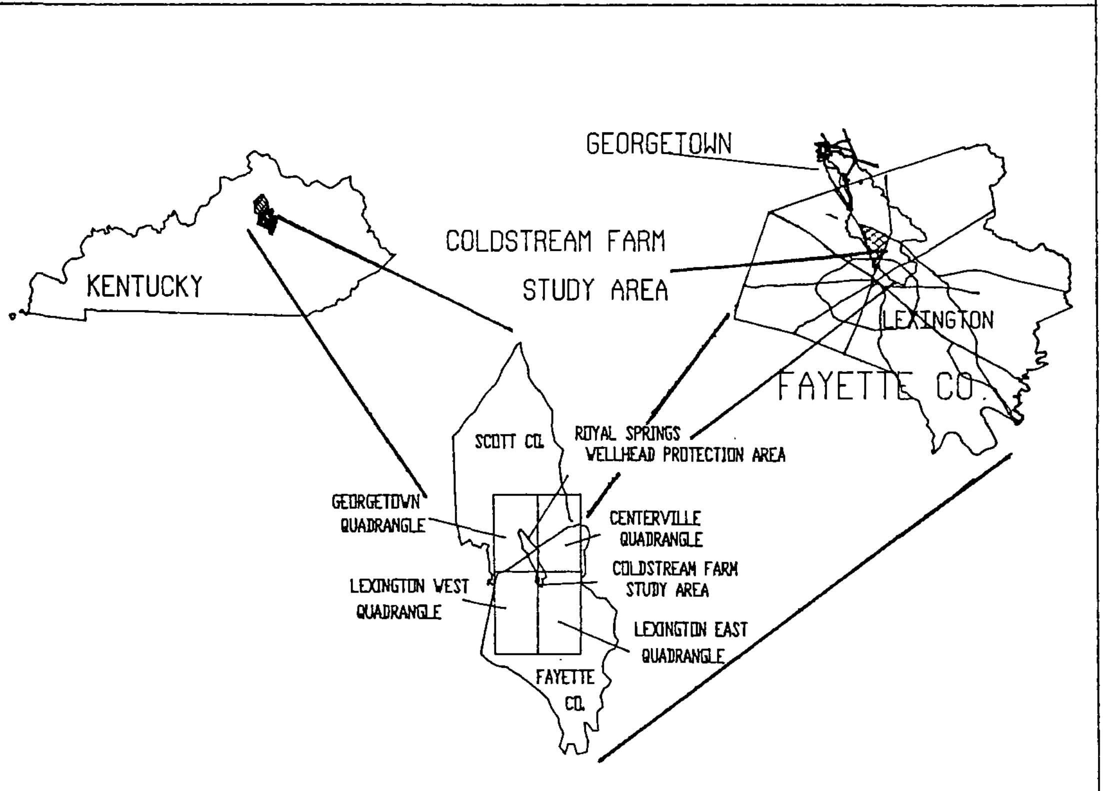
STATEMENT OF THE PROBLEM

A recent planning action of the Lexington-Fayette Urban County Government (LFUCG) focused attention on Royal Spring, the municipal water supply for the community of Georgetown, Kentucky. Royal Spring's flow ranges between 30 l/s (liters/second) on the low side to over 3,000 l/s on the high side with median flow between 300 and 1,000 l/s (Thrailkill et al, 1982). The LFUCG as part of the comprehensive planning process identified an area for special consideration for which the Mayor appointed the Coldstream Small Area Planning Committee in April of 1990. This area lies in the northern part of the city and has a large tract of land owned by the University of Kentucky and is used for agricultural research (Figure 1). The property at one time was considered to be in the country but now with the addition of I-75 and increased development in this part of the city, there are several reasons why land use revisions should be discussed.

As part of the deliberations of the Committee and the planning staff, the fact that this tract of land was part of the recharge zone for Royal Spring became known and was discussed at some length. The final report of the Committee includes a section that addresses this natural feature.

This location was an area included in one of four quadrangles mapped using the DRASTIC methodology. Likewise, the Royal

FIGURE 1
COLDSTREAM FARM STUDY AREA LOCATION



Spring was identified as a public water supply to be included in the Pilot Wellhead Protection Study of Kentucky. The availability of these maps and accompanying reports provided an opportunity to analyze the utilization of these data in the planning process.

DRASTIC MAPS AND RESULTS

Four quadrangles were mapped, Lexington East (MacQuown and Dobrovolny, 1968), Lexington West (Miller, 1967), Georgetown (Cressman, 1967) and Centerville, (Kanizay and Cressman, 1967) using the methodology defined by Aller (1987). A combined version of these four quadrangles are presented in Figure 2. It can be seen that DRASTIC Index values range from a low range of 100-119 to a high of 200+. Most of the background of the maps fall into the Index category of 180-199. These are considered high values and indicate a high potential for groundwater contamination. In constructing these maps, the work of John Thrailkill and his students (1982) was used because they identified areas drained by individual major springs as groundwater basins through dye tracing studies. These basins can be seen on the DRASTIC map as fat, teardrop-shaped areas with index values less than the areas surrounding them (160-179). The other areas (referred to as "background" above) of the map fall into what Thrailkill has identified as interbasin areas.

This difference in values was expected between groundwater basins and interbasin areas, but the fact that the groundwater basin has a lower DRASTIC Index is just the opposite relative to its vulnerability for contamination. Within the groundwater basin, there are more sinkholes and greater opportunity for surface water to move from the surface to the conduit below. This is an important defect in the DRASTIC method as applied to karst areas. The difference is produced because Thrailkill has shown that in the groundwater basins the conduit is the main drain and the water table is deep, at conduit level. In the DRASTIC method, deep water table is given less value because of the rationale that the deeper the water table, the less likely it will be contaminated by surface pollutants.

Royal Spring's groundwater basin is elongate and extends southward from the city of Georgetown. It is not represented by only one DRASTIC Index value but is divided into two regions with the higher index value (200+) in the southern part of the basin and the other region (160-179) occurring near the spring in the northern part of the basin. The difference between index values is controlled by the difference in surface elevation. Most of the groundwater basin underlies the surface valley of Cane Run, but in the lower region of the basin, closer to Georgetown and the spring, the groundwater basin does not underlie the Cane Run valley, because the valley abruptly turns toward the west. The lower end of the groundwater basin lies beneath a higher topographic surface, thus the deeper water table and lower DRASTIC Index value.

FIGURE 2
ROYAL SPRINGS - DRASTIC INDEX MAP

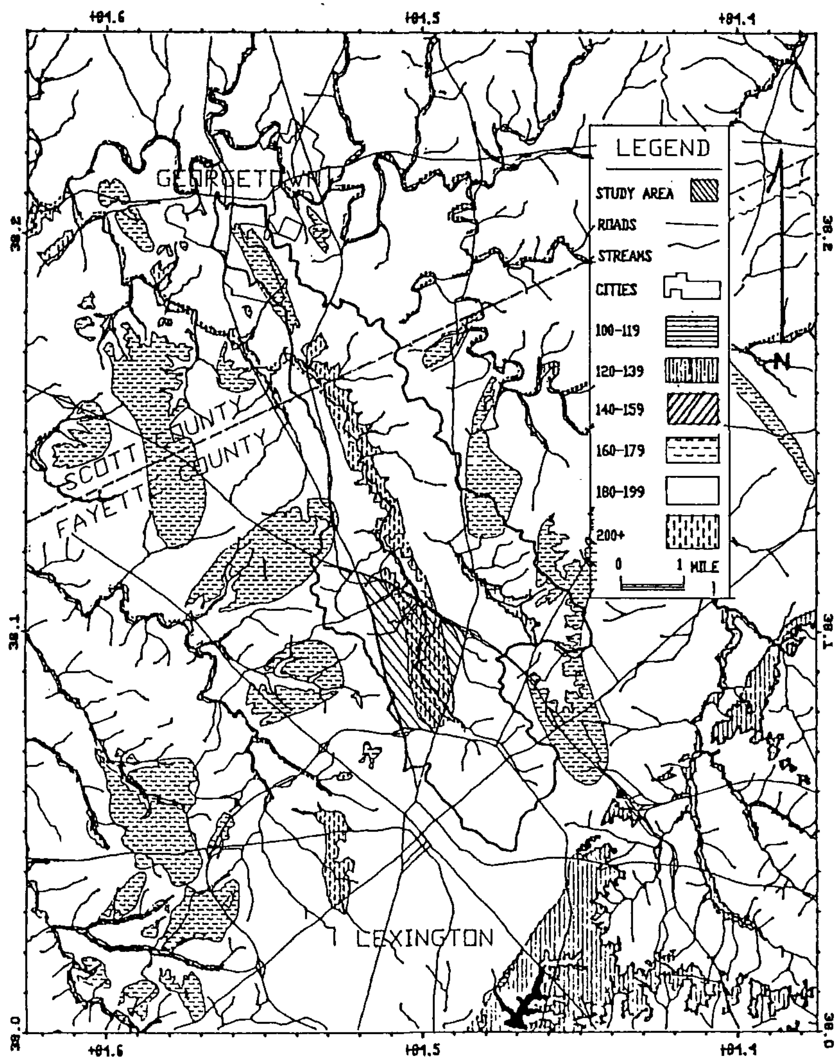
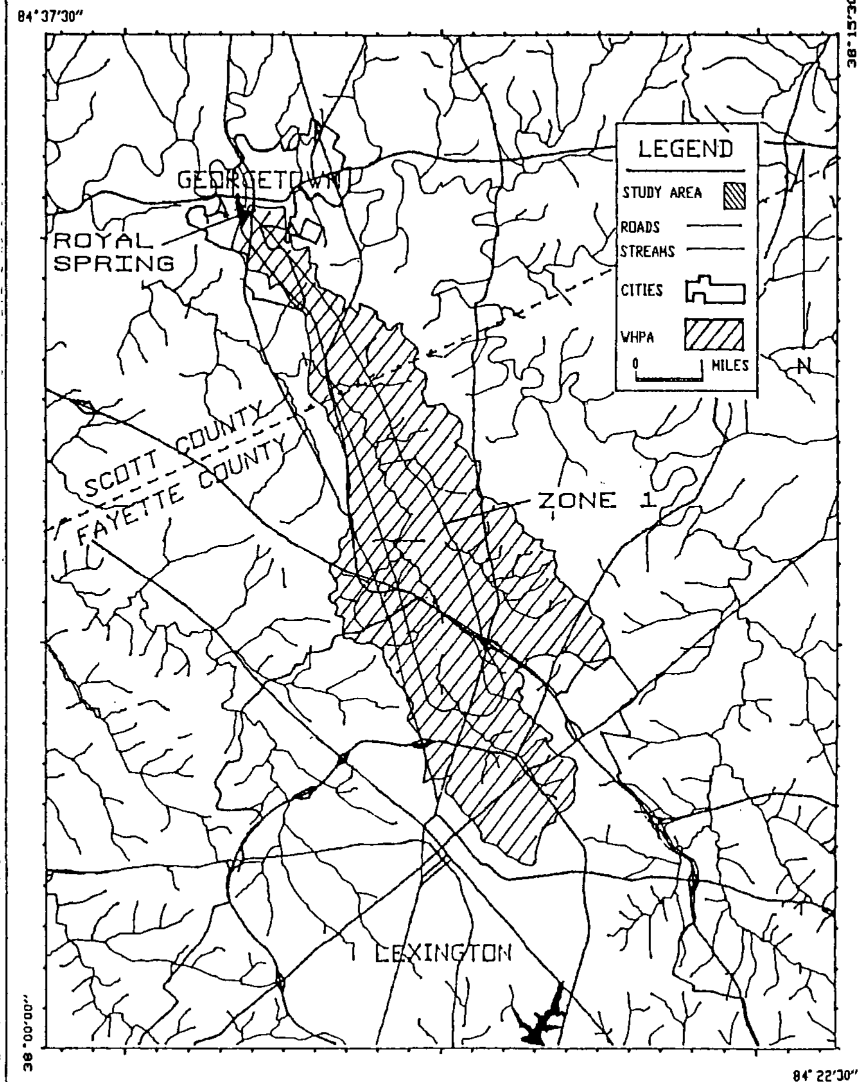


FIGURE 3
ROYAL SPRINGS - WELLHEAD PROTECTION AREA



WELLHEAD PROTECTION FOR ROYAL SPRING

As part of the pilot study of Kentucky, Royal Spring was identified as a municipal water supply to be studied. The study included defining the Wellhead Protection Area (WHPA) and identifying potential sources for contamination of the aquifer. Figure 3 shows the Wellhead Protection Area for Royal Spring. Using U.S. Environmental Protection Agency terminology, the Zone Of Contribution (ZOC) for this area is equal to the recharge area to the spring. A second zone has been defined where the Time Of Travel (TOT) is measured in hours and days rather than months and years common in non-karst regions.

The boundaries of the area were determined by using Thrailkill's approach to not only include the groundwater basin but also the "catchment area" for the groundwater basin. By reference to Figure 2, one can see the other groundwater basins that bounded the Royal Spring groundwater basin. The area bounded by the drainage divide for Cane Run represents the catchment area. Two deviations from this occur, one at the southern end of the WHPA where Russell Cave Spring robs part of the drainage of Cane Run, and the other at the northern end of the WHPA where Cane Run abruptly turns to the west and the groundwater basin no longer underlies the drainage basin of Cane Run (Figure 2).

A zone (Zone 1) within the WHPA (Figure 3) that defines the groundwater basin represents a more vulnerable region of the recharge basin, because within this zone more than 150 openings to the main conduit have been identified by Thrailkill and include the swallets that have been used for tracing the basin. Maximum travel times from the farthest swallet have been as low as 141 hours, but calculations of flow velocities are in the order of 4 km/hr, which would transfer to as little as four hours of travel time for the Newtown Swallet that is approximately 15 kilometers from the spring. The remaining zone represents the rest of the area within the ZOC and falls into the interbasin area.

The other major part of the Wellhead Protection method is the identification of potential contamination sources. Sources identified and located on maps and tabulated for the Royal Spring WHPA (Fitzmaurice, 1990) are: subsurface percolation from sanitary systems, injection wells, landfills, open dumps (including illegal dumping), residential (or local) disposal, graveyards, aboveground storage tanks, underground storage tanks, containers, pipelines, material transport and transfer, operations, animal feeding operations, production and other wells, monitoring and exploration wells, and salt-water intrusion/brackish water upcoming. The information obtained on these potential contamination sources varied for Fayette and Scott Counties. Fayette County is the more populated of the two counties and more potential sources are present and thus more information available. The LFUCG has computerized much of their information and have a program that converts street addresses to latitude/longitude which made it possible to plot information on maps. These data are included on maps in Fitzmaurice (1990).

COLDSTREAM SMALL AREA HYDROGEOLOGY

The topography ranges from 980 feet at the southern end to 885 feet at the point where Cane Run leaves the property. The floodplain of Cane Run is 910 feet at the southern end of the site and 890 feet at the northern end, with a distance of approximately 9,900 feet or a floodplain gradient of 20 feet/9,900 feet (0.002). The total area lies within the upper region of the surface drainage basin of Cane Run. Presently, developed areas (the two subdivisions and the industrial park) are connected to the sanitary sewer system of LFUCG, but surface run off is directed to the natural or constructed drainage ways within the area.

Karst features within the area are two swallets (one considered an estavella by Thrailkill), one active spring, one inactive spring (tufa deposits observed at this spring), sinking stream (most of Cane Run is dry except for major precipitation events and spring flow terminates in one of the swallets), and several sinkholes.

The geologic units present in this area are the various members of the Lexington Limestone of Ordovician age. The Lexington Limestone includes argillaceous units generally less than 6 meters thick and highly variable. Within the area of interest the lower unit of the Lexington Limestone is the Grier Limestone and above that is the Tanglewood Limestone. Interbedded within this unit are the argillaceous units, some include interbedded shale.

Royal Spring emerges from the Grier Limestone Member about five meters below the contact with the Tanglewood Limestone. Within the area, one of the argillaceous units is the two meter-thick Cane Run bed, which is mapped about 10 meters above the Tanglewood-Grier contact. Thrailkill believes that the main conduit for Royal Spring may be stratigraphically controlled within this interval and that the Cane Run bed does not perch the subsurface conduit because most or all of the dye traced swallets within the groundwater basin penetrate the unit.

Surface flow in Cane Run is intermittent on the Coldstream Small Area, but a part of the stream has been observed to flow between precipitation events. The source of the water is a spring shown on Figure 5. Because of the construction in the area, it is not possible to determine if the source of the spring water is related to the storm water system of LFUCG. This water flows on the surface for a short distance entering Cane Run and sinking in the channel at the swallet identified by Thrailkill as the Coldstream Swallet (Figure 5). Near the point where Cane Run leaves the Coldstream Small Area, discharge from the University of Kentucky farm operation flows into the channel.

COLDSTREAM SMALL AREA PLAN

Land use of the 1770.9 acre area as listed in the 1988 Comprehensive Plan is shown in Table 1. The Coldstream Small Area plan changes are also presented in Table 1.

FIGURE 4

COLDSTREAM SMALL AREA PLAN

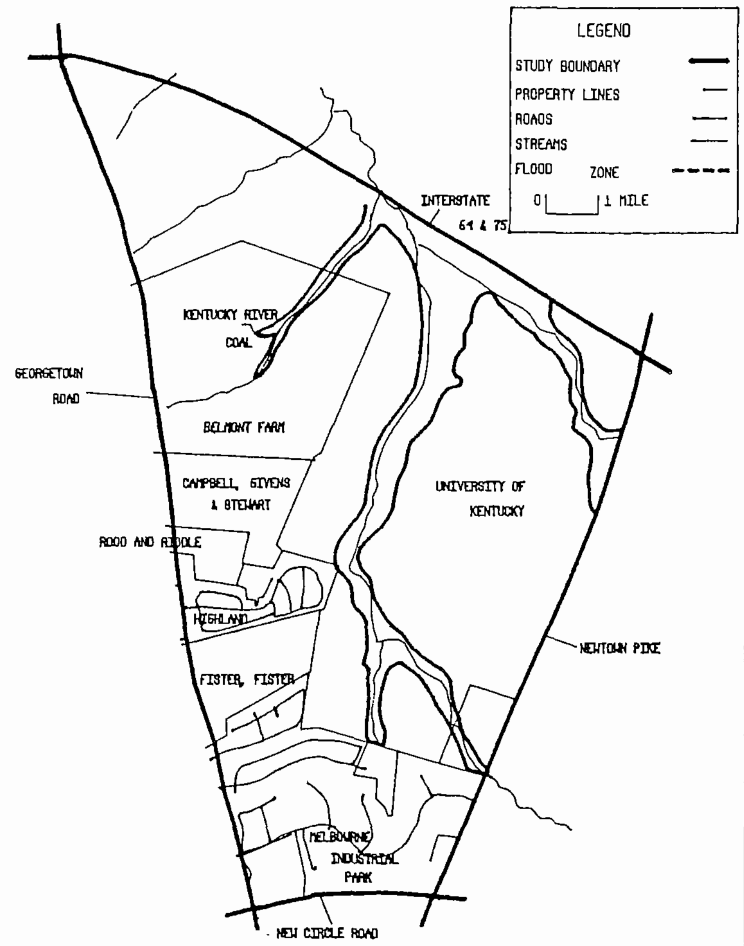


FIGURE 5

COLDSTREAM SMALL AREA - TOPOGRAPHY

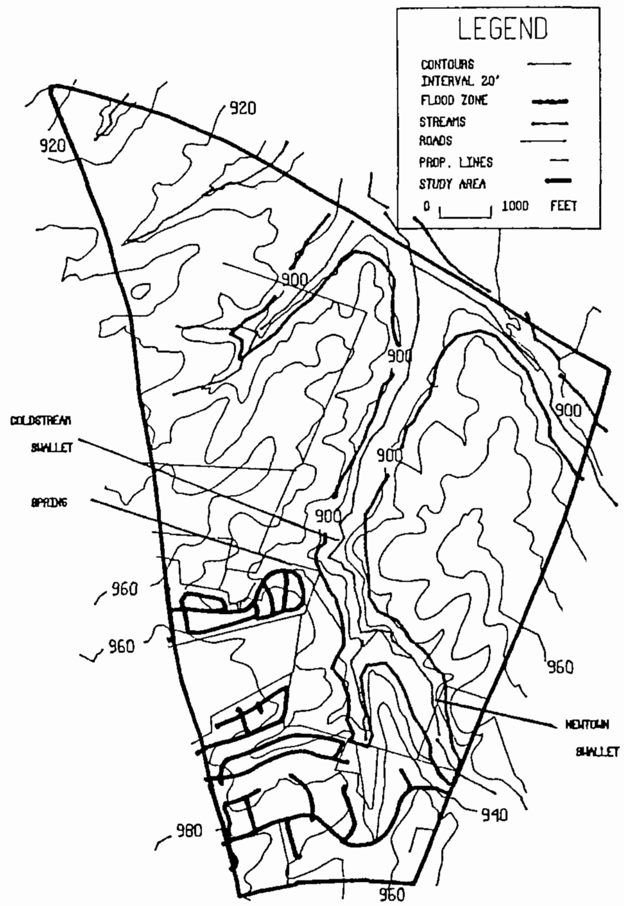


TABLE 1

EXISTING AND PROPOSED LAND USE FOR COLDSTREAM SMALL AREA

	1988 PLAN			1991 PLAN	
LAND USE	EXISTING	PROPOSED	TOTAL	PROPOSED	TOTAL
LOW DENSITY RESIDENTIAL	117.6	69.0	186.6	178.5	296.1
MEDIUM DENSITY RESIDENTIAL	0.0	307.5	307.5	139.9	139.9
RETAIL TRADE & PERSONAL SERVICES	0.6	9.8	10.4	25.5	26.1
PROFESSIONAL SERVICES	27.8	22.9	50.7	61.0	65.3
HIGHWAY ORIENTED COMMERCIAL	3.8	0.0	3.8	0.0	3.8
SEMI-PUBLIC USES	5.6	0.0	5.6	0.0	5.6
PARKS	16.8	0.0	16.8	0.0	16.8
OTHER PUBLIC USES	984.9	0.0	984.9	235.3	235.3
OFFICE, INDUSTRIAL RESEARCH PARK				781.3	781.3
WAREHOUSE & WHOLESALE	146.8	0.0	146.8	0.0	146.8
CIRCULATION	54.4	0.0	57.4	0.0	57.4
UTILITIES & COMMUNICATIONS	2.2	0.0	2.2	0.0	2.2
TOTAL	1363.5	407.4	1770.9	1421.5	1770.9

Figure 4 identifies the major land use of the Coldstream Small Area, and Figure 5 illustrates the topography and geological features. The major change in the area will be the development of the 984.9 acres, listed as Other Public Uses, owned by the University of Kentucky. It is the University's intent to develop a research park in this area. One site has already been developed by Hughes Aircraft on the property, and the University has plans for similar high-technology industries to locate on the property and be related to the University of Kentucky through faculty and student interaction with the new industries.

The Committee has recommended the following changes to the 1988 Comprehensive Plan that are shown in Table 1. Major changes are the addition of 178.5 low density residential, 139.9 medium density residential, and the addition of a new land use identified as Office, Industrial Research Park that would have 781.3 acres.

While the Committee Report acknowledges that much of the Coldstream Small Area falls in the recharge area for Royal Spring, no specific recommendations were made to protect the spring from water quality changes that will be caused by the increased development. The Georgetown Water Board presented to the Committee the policies they plan to follow relative to development in the recharge area under their jurisdiction (Table 2).

TABLE 2

AQUIFER RECHARGE PROTECTION POLICIES FOR ROYAL SPRING

The Urban Service Boundary should not be extended into the Royal Spring Aquifer recharge area for urban development.
No additional properties within the Aquifer Recharge Area **should be planned industrial.**
Existing lands planned industrial but not yet zoned should be redesignated "environmentally sensitive light industry". This category does not allow hazardous materials users/generators.
For **lands already zoned industrial,** new hazardous materials users are prohibited, and existing hazardous materials users may expand only if the design and operation of the site would protect the aquifer from contamination.
Design standards are given for **safe storage and spill containment.**
No new underground storage tanks should be located in the Aquifer Recharge Area.
Rural residential development on septic systems should be clustered outside the recharge area where feasible.
The recharge area has the **highest priority** in the proposed **TDR/PDR program.**
The **LFUCG** is encouraged to adopt similar protection policies.

STATUS OF KNOWLEDGE OF THE SITE

The long term studies of John Thrailkill and his students have identified the swallets that are directly linked to the Royal Spring. Even though this information has been known for the last ten years, very little practical application of the knowledge toward planning resulted.

The DRASTIC analysis illustrates that the Coldstream Small Area falls in the highest and second highest categories of the DRASTIC Index indicating high vulnerability to groundwater contamination. Although the study results are inconsistent on an area basis because it did not highlight the most vulnerable areas for most of the area studied, it did identify the most vulnerable area that must be protected in the Coldstream Small Area (200+ index area).

The Wellhead Protection study identified the recharge area (ZOC) for Royal Spring and based on previous work did identify Zone 1 (200+ index which is a groundwater basin identified by Thrailkill) as the most vulnerable area within the Wellhead Protection Area that must be protected. Once Georgetown officials became aware of this map and related study, they began to include the information in their planning process. It was the

WHPA map and Georgetown's desire to be included in the Coldstream Small Area Planning Committee that brought the issue to the table.

Systematic stream hydrographs have not been developed for Cane Run or the segment flowing across the Coldstream Small Area. Cane Run receives flow from a spring on the site and loses water to two swallets and fractures in the bottom of the Cane Run channel. Cane Run is an intermittent stream that will experience increased discharge as development proceeds.

The Sinkhole Ordinance (described by Dinger and Rebhman, 1986) enacted in Fayette County is an important application of geological knowledge but is not directed toward protection of groundwater quality. In fact, the ordinance allows the direction of surface runoff toward sinkholes but does exclude the construction of dwellings within them.

MANAGEMENT OPTIONS FOR THE COLDSTREAM SMALL AREA

The proposal to rezone this area has provided an opportunity for the analysis of current laws and policies relative to the protection of groundwater resources utilized as a municipal water supply. While the city officials of Georgetown, Kentucky are cognizant of the many potential sources of contamination to their water supply and the fact that it is highly stressed during periods of drought, other sources of water are being sought but will not be in place in the near future, thus this supply is very important to them. The problem is more complex because Georgetown does not control the whole WHPA and therefore cannot enforce land use restrictions to protect the aquifer.

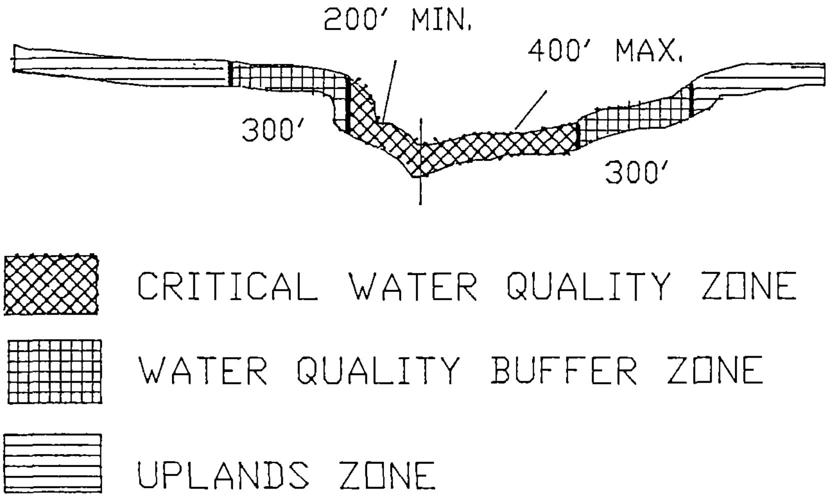
State officials recognize the dilemma faced by Georgetown, but feel they have no regulations that would allow them to effect the development of a protection policy. They were able to inject the State into a problem during the drought of 1988 and cause a single user to stop pumping a large volume of water from the conduit supplying Royal Spring, thus allowing the water to be used for the citizens of Georgetown.

Few examples exist to use as a model for developing a protection strategy for this area. Austin, Texas is currently experimenting with two policies to protect a karst aquifer that provides an important recreational spring (Butler, 1987). East Marlborough and West Whiteland Townships, Chester County Pennsylvania have adopted zoning ordinances to protect a carbonate aquifer (Jaffe and Dinovo, 1987). Both of these utilize resource protection and source control methods to protect the aquifer from potential contamination.

As part of the Coldstream Small Area Plan, a green space (resource protection) to roughly parallel the 100 year floodplain for Cane Run on the property was proposed. This would be an area where no construction other than utility lines and limited roads cross the channel. This approach has been used in Austin, Texas as a protection strategy for the recharge zones of the Edwards Aquifer (a karst aquifer). They have established three zones as illustrated in Figure 6 based on stream hydrograph and water quality data collected for the area. The Pennsylvania example limits the number of dwelling structures per acre and controls the source of potentially contaminating substances such as home heating oil.

FIGURE 6

MAJOR WATERWAY WATER QUALITY ZONES



Application of the water quality zones illustrated in Figure 6 to the Coldstream Small Area would be similar to the designation of the green space that is under consideration at the present time. The Critical Water Quality Zone (Figure 6) is a zone that does not allow any improvement other than minor roads but is generally used as a park area. The Water Quality Buffer Zone is a zone with limited development activity. For example, very low density residential or light industry with limited toxic substances used by them. The Upland Zone does not have any restrictions. If this were overlaid on the sinkhole ordinance in the Coldstream Small Area, this would be a beginning of a protection strategy for Royal Spring.

To make any zoning strategy cost effective, baseline information on the physical properties within the area should be collected before development takes place. This information should include the following: 1) stream flow from all channels providing water to Cane Run, 2) water entering and leaving the Coldstream Small Area, 3) channel gain or loss within the Cane Run channel on the property, and 4) water quality of water entering Cane Run from all sources (springs, surface water discharges from developed areas, and overland flow). Information of flow conditions will allow for the exact location of the boundaries of the green space.

Likewise, consideration of the kinds of land uses that are best suited for the area should be defined. It is understood that sanitary sewers are to be provided for all occupants of the new developments. Special consideration of the quantity and quality of surface water leaving the developed properties must be given so that precautions can be taken so that the quality of the water entering Cane Run does not deteriorate.

CONCLUSIONS

The use of scientific information for the purpose of planning is not easy to document because it becomes part of the policy-making process and this process is not generally documented by retrospective analysis. In this case, Royal Spring is the largest spring in the Commonwealth and is used as a municipal water supply and the Kentucky Department of Environmental Protection and Natural Resources became interested in protection strategies for the spring. Their support of the DRASTIC and Wellhead Protection studies focused attention on the spring and it is fortuitus that the LFUCG began to study the Coldstream Small Area to develop a plan for rezoning the area within the same time frame. From this study the following conclusions can be made:

1. The Wellhead Protection analysis had more of an impact on the planning process because it defined the Zone Of Contribution for the Royal Spring.
2. The zone identified within the WHPA (Zone 1) is a very vulnerable part of the ZOC and occupies a large portion of the Coldstream Small Area Plan.
3. The previous work by Thrailkill, while known to exist for some time, was not directly applied to the planning process until the present time.
4. Because approximately 80% of the ZOC is outside the jurisdiction of the Georgetown City Council, protection strategies for the entire WHPA cannot be enforced.
5. There are no state or federal laws that give the Georgetown City Council authority to protect the ZOC outside of Scott County.
6. Protection of the ZOC within the Coldstream Small Area will depend on the building permits issued by the LFUCG. The statement that LFUCG recognizes that the portion of the Cane Run drainage basin on the property is in recharge area for Royal Spring is a positive sign, and their willingness to define a green space along the stream channel that includes the two swallets that are directly linked to Royal Spring is a form of aquifer protection.
7. Baseline data must be collected to document the potential changes in the quantity and quality of water flowing into Cane Run before, during and after development if a protection strategy is to be cost effective.

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BIOGRAPHICAL SKETCH

Lyle V. A. Sendlein received his BS and AM in geological engineering and geology, respectively, from Washington University in Saint Louis, Missouri in 1958 and 1960. He received his Ph.D. from Iowa State University in geology and soil engineering in 1964. He taught and conducted research for seventeen years at Iowa State University. In 1977 he assumed the directorship of the Coal Research Center at Southern Illinois University at Carbondale and in 1982 assumed his current position as the director of the Institute for Mining and Minerals Research at the University of Kentucky. He is also a Professor in the Department of Geological Sciences and is currently directing graduate student research and conducting research in hydrogeology. His areas of research include aquifer definition in both the coal fields and the karst regions of Kentucky and contaminant transport in these hydrogeological domains related to mining, waste disposal and agricultural practices.

ANALYSIS OF DRASTIC AND WELLHEAD PROTECTION METHODS APPLIED TO A KARST SETTING

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Question 1.

Because DRASTIC drastically under-estimated the vulnerability of those portions of the karst basins dominated by conduit flow and recharged directly by swallets in sinking streams and open sinkholes, how can it be utilized as a predictive hydrologic mapping tool in karst? Aren't there serious limitations and dangers in relying on DRASTIC in karsts without performing tracer studies and doing field reconnaissance for direct recharge sites such as swallets and open sinkholes? Won't dye-tracing give you more information (and more accurate information) about aquifer vulnerability?

Question 1a.

"Because DRASTIC drastically under-estimated the vulnerability of those portions of the karst basins dominated by conduit flow and recharged directly by swallets in sinking streams and open sinkholes, how can it be utilized as a predictive hydrologic mapping tool in karst?"

The key words in this question are "predictive hydrologic mapping tool". DRASTIC is not a hydrologic mapping tool. The method is designed to assist in the planning process, which would address large areas and is not designed for site specific use. For the areas mapped in this study, high DRASTIC Index values resulted which would alert public officials (including planners) that this area is vulnerable to groundwater contamination from surface activities. As indicated in the paper, problems arose when karst groundwater basins determined by dye traces were introduced into the analysis because the deep groundwater table caused by drainage through the main conduit resulted in a lower DRASTIC Index value for those areas within the groundwater basins.

Question 1b.

"Aren't there serious limitations and dangers in relying on DRASTIC in karsts without performing tracer studies and doing field reconnaissance for direct recharge sites such as swallets and open sinkholes?"

If the analysis produces high DRASTIC Index values, the purpose of the analysis has been accomplished, even though specific karst features have not been identified. I do believe that if one has knowledge of specific karst features that would directly transport contaminants to the groundwater within the conduit system, they should be added to the map.

Question 1c.

"Won't dye-tracing give you more information (and more accurate information) about aquifer vulnerability?"

Of course dye-tracing will give site specific as well as general information about the karst groundwater system. This would have to be the next step in an analysis of an area should one decide to proceed with the planning of a specific land use knowing that the DRASTIC Index value is high, which indicates a high vulnerability for contamination of the groundwater.

Question 2.

I have found that although DRASTIC is a powerful and practical mapping tool in terranes consisting of many rock types, it is useless for distinguishing slight differences in vulnerability if the entire area to be evaluated in a karst. DRASTIC wasn't designed to evaluate differences between contiguous karst areas. For Hart County, Kentucky, I found direct interpretation of the published geologic maps to be most reliable. Your comment is welcome.

I agree with this statement. For the public official or planner who may not know anything about geology and karst, a DRASTIC map of a karst area would alert the planner to the high vulnerability for contamination of the groundwater by surface activities. A qualified hydrogeologist can take a published geologic map and probably glean more valuable information about the groundwater flow systems in the area than a lay person. But I would remind all that this method was not developed for geologists, but for public officials and planners.

USE OF DYES FOR TRACING GROUND WATER:

ASPECTS OF REGULATION

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ABSTRACT

An affirmative EPA policy on tracers is needed, and I urge that steps be taken to establish one as part of legislation or administrative regulations. Suggested wording for part of a statute is included here, but if legislation is unnecessary and a policy can be established administratively, it should be so done. Tracing is a technique uniquely able to test and predict flow velocity and flow direction of water and pollutants. When properly used in the settings where they work best -- karst aquifers, fractured rocks, and coarsely granular sediments -- tracers can give reliable answers that can not be obtained in any other way. The substances traditionally used for tracing are benign and harmless in the concentrations employed by most knowledgeable investigators. The use of tracers is justifiable on the basis of their safety, ease of use, utility, reliability, and unique ability to secure results obtainable in no other way. Currently, EPA has no policy on the use or non-use of tracers, but their use has been endorsed by the Agency in several publications and reports. Some states routinely employ tracers; a few regulate them; a few forbid their use. Most states ignore them and have no policy on their use.

INTRODUCTION

Tracing agents (tracers) are added to a medium, usually water, to enable the medium to be recognized at a remote location. Their arrival concentration and arrival time can be used in calculations and decisions. The most common tracers are substances added (such as fluorescent dyes) or physical changes (such as temperature) made in the medium. This paper is concerned primarily with tracers added to water and used to study its movement and velocity as they relate to environmental problems.

Tracers 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 for high-reliability prediction of contaminant flow-paths and flow-directions. They can be essential for the calibration of computer models of water flow and pollutant movement, and they greatly enhance the validity of such models. Tracing is cost-efficient and is often the only way to obtain essential data.

Two major types of tracers are used most commonly: water-soluble fluorescent dyes that have been shown to be non-toxic (Field et al., 1992) and ions of non-toxic elements or compounds. Specific tracers are named in a subsequent paragraph.

The use of tracers as an essential aid in the design of monitoring systems in karst terranes is critically reviewed by Quinlan (1989, 1990). Most karst terranes are areas underlain by limestone and/or dolomite and characterized by sinkholes, sinking streams, caves, and springs. Approximately 20% of the U.S. (40% of the eastern U.S. that includes the upper Mississippi Valley) is underlain by carbonate rock and is some type of karst.

An anonymous wag has whimsically stated that "One well-designed tracer test, properly done, and correctly interpreted is worth 1000 expert opinions . . . or 100 computer simulations of ground-water flow in karst terranes." He believes that this is because most computer models are incapable of giving valid results for flow in most karst terranes -- for various technical reasons not germane to this paper.

I do not deny that great advances have been made (and will be made) in understanding and predicting ground-water flow with computer models, and I freely acknowledge their unique capabilities. Nevertheless, models have recognized limitations (Scarrow, 1989), about which two points must be made:

1. Hard data (real numbers, based on measurements) have greater reliability than soft, imaginary data (calculated values derived from computer models or simulations). Given a choice between believing the results of a well designed, properly executed, and carefully interpreted tracer test or the answers given by a computer model which disagree with the tracer test, the tracer results should be given greater weight -- for the same reason that we do not debate the existence of gravity. The tangible results of each are obvious.

2. Computer models are no better than the validity of the assumptions made in their design, the reliability (representativeness) of the parameters used for data input, and the correctness of the analysis and manipulation of data.

Their validity must be evaluated with a sensitivity analysis and by comparison with hard data.

I make these points because tracers and models do not substitute for one another. Tracer results take priority over model results, however, but each can complement the other, doing what the other can not. Although I enthusiastically endorse the use of computer models for analyzing karst aquifers when done successfully (Teutsch, 1989; Teutsch and Sauter, 1991; Sauter, 1991a, 1991b), I demand the rigor imposed by J. C. Griffiths, the doyen of quantitative geologists, on the methodology of geologic investigation (Drew, 1990, p. 3-10).

DESCRIPTION OF PROBLEM

The use of tracers for solving environmental problems has become well established in recent years. In some settings, it is not possible to define important environmental parameters without first conducting comprehensive tracing studies.

In an unrealistic effort to protect human health and the environment from any possible harm, some state agencies have unrealistically restricted the use of tracers, regardless of the information to be gained. For example, some state agencies have interpreted tracers used in ground-water investigations to be pollutants or contaminants and therefore violations of their "non-degradation policy." Strictly speaking, injection of dye or other non-toxic tracers into a well (or other input source, e.g., a sinkhole), no matter how noble the reason for doing so, makes it possible to construe the well to be a Class V injection well and thus subject to State and Federal regulations governing its use. (If the tracer were toxic, the well would be a Class IV injection well.)

Part of the problem is due to the misperceptions of many individuals about fluorescent dyes and other substances used for environmental tracing studies, especially with regard to their suspected toxicity. This is the fallacious logic used: Fluorescent dyes are synthetic substances (i.e., they do not occur naturally); therefore, they must be a pollutant and harmful to human health and the environment. This misperception is exacerbated by the lack of comprehensive toxicity test data on the dyes. The fallacious argument continues: Since fluorescent dyes must be harmful, so probably must be all other tracing agents. Granted, some tracers are hazardous (e.g., tritium), but assumptions such as these cited are absurd and unscientific, and they decrease our ability to gather data needed to better protect ground-water quality.

Clearly, there is a need for a Federal policy endorsing the use of tracers in ground water if the various misperceptions regarding their use are to be dispelled. Such a policy does not need to be very extensive or detailed, but it does need to be flexible.

DISCUSSION

Well-meaning as the above interpretation of regulations for injection wells and the concern regarding suspected tracer toxicity may be, it is not justifiable in terms of potential benefits for environmental protection, original 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 into it for disposal. And like boats, dyes and ions generally used for tracing ground water are benign and harmless in the concentrations commonly employed (Field et al., 1992; Smart, 1984).

A further analogy describing the use of tracers 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 much smaller 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. If it is felt necessary to give the states regulatory authority in tracing, I suggest that regulation be through licensing of individuals rather than on a study-by-study basis.

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. Several major EPA documents endorse the use of tracers in ground-water studies (US EPA, 1986, p. 64; 1987a, p. 2-12, 2-22 2-23; 1987b, p. 127-148; Quinlan, 1989). Guidance manuals for tracing techniques exist and have been sponsored by EPA (Davis et al., 1985; Mull et al., 1988; Quinlan, 1991) 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 Ground Water Association (Aley et al., in prep.). 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 (Kilpatrick and Wilson, 1989; Wilson et al., 1986; Kilpatrick and Cobb, 1985; Hubbard et al., 1982).

International interest in and recognition of tracing as a useful tool in the kit of the hydrogeologist is suggested by the content of proceedings volumes of the five International Symposiums on Underground Water Tracing held during the past 25 years and by the synthesis of Gaspar (1987), and by a recent meeting on granular aquifers (Moltyaner, 1990). The sixth International Symposium, to be sponsored by the Association for Tracer Hydrology, will be held in Germany in 1992.

It is desirable and necessary that proposed Federal ground-water legislation include a section that sanctions the use of tracing agents, prevents them from being regulated as pollutants or contaminants, and strongly encourages their use in predicting the possible consequences of pollutant release and dispersal. Alternatively, and until legislation is passed, a policy endorsing the judicious use of tracers should be promulgated.

RECOMMENDED SOLUTION

I suggest that legislation or policy concerning ground-water protection include wording (originally developed by Quinlan and Field, 1991) to the effect that:

> The use of organic and/or inorganic tracing agents employed in investigations of ground-water and surface-water hydrology is a desirable activity and shall not be construed as the addition of a pollutant or contaminant to the water when deliberately introduced as a tracer. Useful tracing agents include, but are not limited to, the following: dyes such as fluorescein (Colour Index [CI] Acid Yellow 73), eosin (CI Acid Red 87), Rhodamine WT (CI Acid Red 388), Sulpho Rhodamine G (CI Acid Red 50), Sulpho Rhodamine B (CI Acid Red 52), optical brighteners (fluorescent whitening agents with various CI numbers), Diphenyl Brilliant Flavine 7GFF (CI Direct Yellow 96), Lissamine Flavine FF (CI Acid Yellow 7), pyranine (CI Solvent Green 7); ions such as lithium, chloride, bromide, iodide, nitrate, and sulfate; fluorinated organic anions; *Lycopodium* spores; bacteriophages; noble gases; other gases such as sulfur hexafluoride; stable isotopes; particulates; hot water; and cold water.
>
> Sewage effluent and waste products from industrial activity, waste disposal, or spills may function as a tracing agent, but they are not suitable for deliberate addition to water as a tracer.
>
> Investigations regarding the vadose zone in fractured and karstic terranes can employ the use of smoke and various gases (e.g., sulfur hexafluoride). These can be used for determining subsurface interconnections above the potentiometric surface when light nonaqueous phase liquids (LNAPLS) and other highly volatile contaminants are residing in the vadose zone.
>
> The U.S. Environmental Protection Agency, U.S. Geological Survey, American Society for Testing and Materials, and professional societies of hydrologists, hydrogeologists, and engineers are authorized to establish voluntary

protocols for the selection and use of tracing agents.

Tracing should be conducted only by those individuals who can demonstrate evidence of substantial training and/or experience in the use of tracer agents. The training may have been acquired through academic study at a university, courses offered by professional societies, or by experience gained while working under the supervision of an experienced water-tracing professional.

CONCLUSIONS

The importance of establishing a set of regulations for the use of tracers in the environment, particularly the use of fluorescent dyes in ground water cannot be overemphasized. Their utility in defining the true flow paths and flow rates are invaluable, especially in pollution studies.

Failure to develop coherent regulations will continue to result in confusion regarding the alleged toxicity of the dyes and other tracer substances, their use by professionals versus amateurs, and their importance in the realm of ground-water quality monitoring. It is essential that the Federal government support the appropriate use of water-tracing substances so that much of the confusion regarding their proper use can be overcome.

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BIOGRAPHICAL SKETCH

Dr. James F. Quinlan is a consultant specializing in the hydrology of karst aquifers and techniques for tracing ground water with fluorescent dyes. He has more than 36 years of experience in karsts of 26 states and 23 countries. For 16 years he directed a hydrologic research program for the National Park Service at Mammoth Cave National Park, Kentucky. He has written or co-written more than 170 publications on ground-water monitoring, evaluation of waste disposal sites, and spill response in karst terranes. In 1985 he and Ralph Ewers received the Burwell Award from the Geological Society of America for a pioneering paper on ground-water monitoring in karst terranes. He is a chairman of an ASTM Task Group on ground-water monitoring in karst and other fractured rocks, has been a Director of the Association of Ground Water Scientists and Engineers, has organized numerous scientific meetings and, since 1986, has annually co-taught an N.G.W.A. short course on practical karst hydrogeology.

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USE OF DYES FOR TRACING GROUND WATER: ASPECTS OF REGULATION

By: James F. Quinlan

Both Missouri and Kentucky are in the process of establishing dye-trace data bases. This will require sharing of tracing results. Do you, as a consultant, support this type of additional regulation?

Assuming the data base is available to all, yes, absolutely. Everyone benefits.

Has there been extensive acute and chronic testing of tracers?

Yes, but there is never as much as one might wish for. The best review of this topic is by Pete Smart in the NSS Bulletin (1984. 46(2):21-33).

I have heard that it is dangerous to use Rhodamine WT because it forms nitrosamine compounds which are carcinogenic. Would you please comment on this issue.

The concern about nitrosamines from Rhodamine WT was raised by Abidi (1982, Water Research, v. 16, p. 199-204). She reported that diethylnitrosamine (NDEA) formed in river waters treated with nitrite ions (NO_2^-) and either Rhodamine WT or B. Nitrite is an unstable intermediate ion formed in the reduction of nitrate ions or the oxidation of ammonium and organic nitrogen. Nitrite ions are found, in small concentrations, primarily in waters heavily impacted by animal wastes such as the discharge from trout-rearing ponds or sewage wastes. Nitrite levels are very low in most ground waters, even those with elevated levels of nitrate ions.

Steinheimer and Johnson (1986, U.S.G.S. Water Supply Paper 2290, p. 37-49) did a very detailed and careful follow-up on Abidi's work and found that in four different river waters "nitrosamine formation did not occur in the river samples at concentrations typically encountered during dye injection studies." They then concluded "under customary dye-study practices, NDEA resulting from the use of Rhodamine WT does not constitute an environmental hazard."

No diagnostic tool is without risk, however. An x-ray examination of a broken bone carries a measurable risk of inducing a cancer, for example. The advantage of diagnosing and setting the bone correctly outweighs the risk and diagnostic tool. It should only be used when there is a need for the information it will yield. The danger associated with long-term chronic pollution of a water supply will usually outweigh any risk associated with an improbable, transient, potential exposure to a reaction product of the dye. The dye is not the problem here, it is the conditions that necessitate the dye trace.

This paper is included within the Proceedings of the National Cave Conservation Association

THE EFFECTS OF RECHARGE BASIN LAND-USE PRACTICES ON WATER QUALITY AT MAMMOTH CAVE NATIONAL PARK, KENTUCKY

Joe Meiman
Mammoth Cave National Park
Mammoth Cave, Kentucky

ABSTRACT

A water quality monitoring program was designed at Mammoth Cave National Park to determine if there exists any influence on the water quality of the Mammoth Cave karst aquifer within the park from various land-use practices of the recharge area. These land uses primarily include: heavy agriculture (row crops and livestock), logging, oil and gas production, and residential areas. The program, initiated in March 1990 and extending through September 1992, samples two rivers, and eight springs recharged by lands with varying land-use. Monthly non-conditional synoptic sampling monitors 36 parameters, including site discharge. The first 19 months of data demonstrate a strong correlation between drainage basin land-use and water quality. Contaminant entrainment mechanisms and relative pollutant input rates can be discerned when the mass-flux of selected parameters is calculated. By use of these data, effective resource management decisions can, and are being made to conserve and protect the irreplaceable natural resources of Mammoth Cave National Park.

INTRODUCTION

For what purpose do we monitor the quality of water at Mammoth Cave National Park? Aside from pure stoichiometric data to satisfy our curiosity of the water's chemical composition, spatially and temporally, the fundamental mission of this monitoring program is to better understand, and thus better manage, the aquatic natural resources of the park. During the three year course of this program, data will be collected and interpreted to provide information on the current state of the surface and subsurface water of the park. This data set will be used as a datum from which to compare past and future studies. As the author is not a biologist, no claims, speculations, conjectures, or theories pertaining to the present health or future of the aquatic ecosystems will be made. However, before trained personnel can accurately assess the condition of the park's aquatic life, a broad database of the physical and chemical properties must be available.

Although this phase of the monitoring program is far from complete, there appear to exist a few trends and correlations which deserve mention. The following pages will concern the first nineteen rounds of monthly sampling.

BRIEF DESCRIPTION OF MONITORING PROGRAM

The monitoring program is largely based upon synoptic samplings. Synoptic, as defined by Webster, is "relating to or displaying conditions as they exist simultaneously over a broad area". Although the water quality monitoring program includes two different synoptic approaches, conditional and non-conditional, the later comprises by far the bulk of monitoring activities for the first years of the study. The program also includes topical sampling which provides a detailed evaluation of a particular flow condition, contaminant, basin or river reach.

Choosing synoptic stations within a karst aquifer differs greatly from the same task preformed on a surface drainage. In a surface drainage one can choose sites based upon stream reaches (every 20 miles for example) to improve spatial distribution, or install a station exactly where a known pollutant source is located. The monitoring sites of this program were chosen with respect to land-use practices of the various recharge basins. These practices range from the naturally wooded park-land groundwater basins where human influence has been absent for at least 50 years, to highly agricultural lands with a share of urban use and oil and gas exploration.

The ten non-conditional synoptic stations are sampled, regardless of flow or weather conditions, on the 10th of each month for the duration of the study (Figure 1). The sites are sampled during a single day by one field crew. The need for repetitive sampling (each month for three years) at each non-conditional synoptic station arises from the considerable temporal variability of karst and surface water quality. This variability is largely a result of sudden changes in discharge, and seasonal availability of contaminant sources. Over the course of the study each end of the flow continuum (base and flood conditions) and each growing season will have been encountered several times, as by program design.

The primary use of conditional synoptic surveys is to provide a finer degree of spatial resolution to the descriptions of discrete water quality and flow conditions than would be attainable from the non-conditional synoptic station network. One of the goals of the conditional synoptic surveys is to identify relatively short reaches of drainage basins which have demonstrated (by data from the non-conditional synoptic station network) chronic water quality problems.

Although the program is designed to allow the park to determine the effects of land-use practices on water quality after three years of sampling, we can, at this juncture, observe various traits which may be attributed to types of land-use in the recharge areas. Each time a sample is extracted, discharge at the site is recorded. Parameter concentration, coupled with discharge will yield flow-weighted values. These values will allow us to determine the mass flux (loading) of a particular parameter at various flow conditions. These data will allow us to better determine contaminant source as it pertains to constituent availability, release, and entrainment into the water, and mechanisms of transfer from the surface to the subsurface.

A DISCUSSION CONCERNING MASS FLUX AND FLOOD PULSES

It would be difficult to continue this discussion without first examining mass flux and flood pulses. Mass flux is simply the amount (mass) of a particular parameter passing a point in a given time interval (flux). A flood pulse is the portion of water propagated along a channel and/or conduit as result of a recharge event, most commonly, rainfall. With an understanding of flood pulse movement, one might better understand mass flux signatures of various contaminants.

MASS FLUX

If a contaminant is released into a stream of water at a constant rate, and at some point downstream its concentration and the stream's flow can be measured, the mass flux of the contaminant can be calculated (Figure 1). If flow (Discharge 1) decreases or increases, the contaminant's concentration will proportionally increase or decrease, respectively (Concentration). That is, at times of high flow the contaminant will experience a greater amount of dilution. The resultant mass flux signature (mass flux over time) of a constant source release will consist of a relatively low amplitude disturbance (Mass Flux 1). One may think of this mass flux signature as a type of destructive wave form interference.

Suppose a contaminant is released into a stream only when specific hydrologic conditions are met, a rainfall event of a certain intensity and volume for example. Therefore, if flow (Discharge 2) increases, contaminant concentration also increases (Concentration) as these stores are displaced into the streams during flood pulse activity. That is during the times of peak

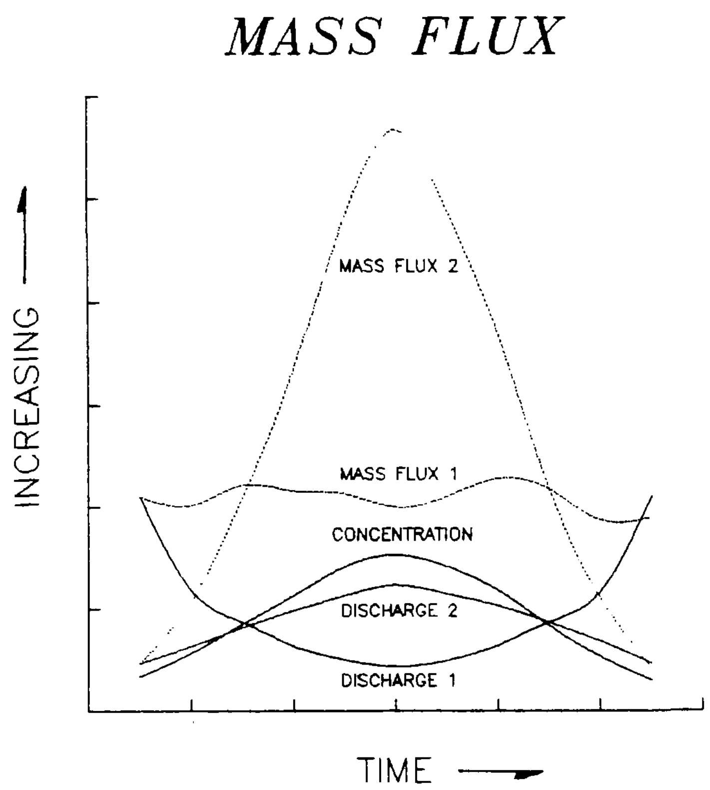

Figure 1. Hypothetical mass flux signatures.

flow, peak (or near peak) concentrations also occur. The resultant mass flux signature of a precipitation-triggered release will be of relatively high amplitude, perhaps several orders of magnitude greater than the pre-pulse mass flux (Mass Flux 2). This mass flux signature may be likened to constructive wave form interference.

FLOOD PULSES

Flood pulses in the Mammoth Cave area may raise a basin's discharge a couple liters per second following minor rainfall, to several thousand liters per second after major rainfall. Research by Meiman (1988 and 1989) has demonstrated that a flood pulse is comprised of two chemically and physically distinct components: displaced stores and freshly input recharge. The former, which usually occurs as the leading edge of a flood pulse and is characterized by high specific conductances, can be thought of as easily displaced vadose storage. Freshly input recharge, which comprises the bulk of a flood pulse, is characterized by low conductances, as there is little time for interaction between its waters and ionic sources. These relationships are also manifested in water temperature, as displaced stores, with longer residence times, will reflect the antecedent system temperature, and freshly input recharge correlative to surface temperature (Meiman, 1988 and 1989).

Consider the flood pulse displayed in Figure 2. This pulse, documented over approximately 42 hours in the fall of 1987 at a sinking creek of the Turnhole Spring groundwater basin, clearly indicates the arrival of three highly conductive sources during the course of flood pulse activity. The majority of flow generated from this rainfall event was of the low-conductance, run-off variety. If water provenance suddenly changes, one may expect to see a similar change in water quality with respect to available water-borne constituents. Hallberg, et al (1985) identified an acute, albeit brief, water quality degradation associated with this run-off component in the karst of northeastern Iowa.

WATER QUALITY, EXPRESSED BY MASS FLUX, AS IT RELATES TO FLOOD PULSES

Important water quality information may be gained if knowledge of flood pulses is combined with mass flux signatures. As rainfall occurs, flood pulses are generated and propagated through the karst aquifer. Just as stage may suddenly vault from its base condition, water quality may also undergo rapid and drastic change as a flood pulse passes. If a significant amount of

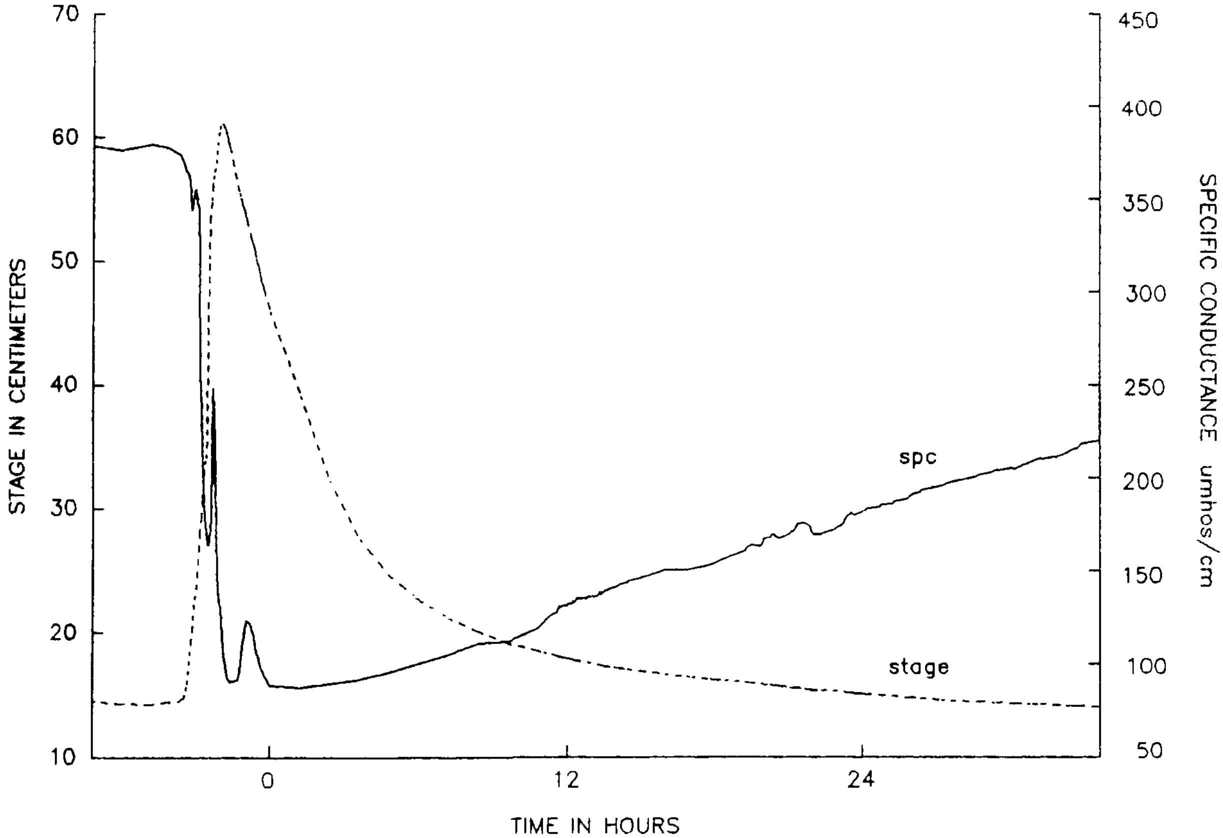

Figure 2. Stage and specific conductance responses of a flood pulse in a surface stream, Turnhole Spring groundwater basin.

constituents are released by the precipitation event (entrained in run-off), the mass flux of these elements may rise tremendously.

Contemplate the two hypothetical mass flux signatures of Figure 3. It is vital to note the relative, unitless scales of the two graphs. Although the X-axis scales are equal, the Y-axis of 3a is 1/8th that of 3b. Also note that "Time 0" indicates the advent of precipitation. The same discharge hydrograph is employed for both graphs. Remember, data used in these graphs are hypothetical. Numbers were derived by a noting the timing, duration, and wave-form characteristics of years of continuous data (stage, specific conductance, water temperature and discharge) and months of water quality data. Mass fluxes are actual products of discharges and concentrations.

If a constant source parameter is monitored through a flood pulse, oil-field brine chlorides from a leaking well casing for example, a response similar to that of Graph 3A might be expected. Following an initial upward spike in concentration, perhaps caused by a flushing of vadose stores, chloride concentration is diluted as the pulse's freshly input recharge component dominates the flow. The resultant mass flux signature, although not without structure, displays relatively low amplitude disturbance.

If the same discharge is used with a precipitation triggered release parameter, certain pesticide residues for instance, a totally different mass flux signature results (Graph 3b). Again, note the Y-axes scales. The effects of constructive wave-form

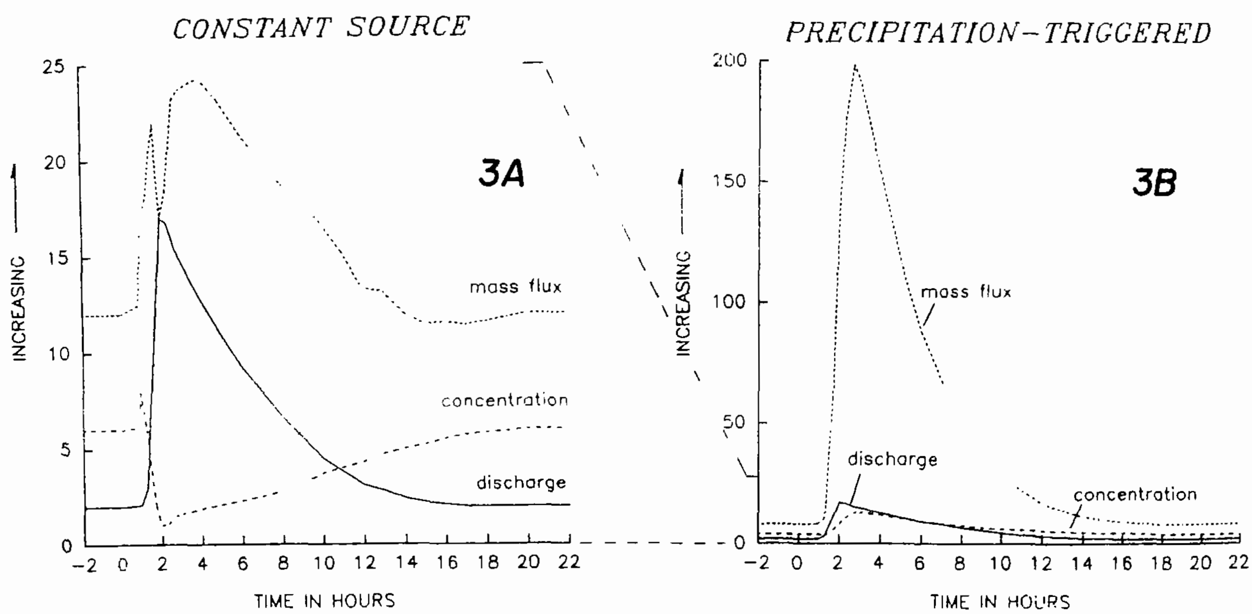

Figure 3. Hypothetical mass flux signatures of constant source release (3A) and precipitation-triggered release (3B).

interference are noticeable as a high amplitude mass flux signature is generated. The passage of the leading edge of the pulse may be reflected in a sharp drop in pesticide residue concentration, as long-residing stores are displaced. This effect will be far overshadowed by the arrival of the freshly input recharge. Not only does this flow component comprise the majority of the flood pulse discharge, it also contains the bulk of surface run-off with entrained herbicides. The resultant mass flux signature may be several orders of magnitude higher than pre-pulse values.

There are many factors that may control the shape of the mass flux signature: availability of constituents, entrainment method, transfer mechanism from surface to subsurface, rainfall volume and areal distribution, time since last rainfall, and conduit condition, to name a few. It should be noted that the conduit condition used in this example is highly vadose. A different signature, especially with respect to temporal lags of concentrations and discharge peaks, will occur when dealing with a phreatic conduit system (Meiman 1988, 1989). Current research at Mammoth Cave specifically addresses flood-pulse water quality.

Perhaps by close examination of mass flux signatures of fecal coliform bacteria, dominant waste sources, human or animal, may be discerned. Human waste, for the most part, should behave as a constant release source. Human waste is injected directly into the aquifer via leach fields, leaking septic tanks, or dry-wells, at a relatively constant rate. A constant mass flux signature should result. Animals, not nearly intelligent enough to defecate down wells, will deposit waste on the surface. Without rainfall (or a major snow-melt), this waste will not be transferred into the aquifer. Following a significant recharge event, animal waste will be washed into the aquifer, producing a mass flux signature characterized by a very high amplitude disturbance.

RESULTS OF MARCH 1990 THROUGH SEPTEMBER 1991

The following data (based upon non-conditional samples) run from March 1990 through September 1991. The summer months, which comprise a disproportionally large percentage of the data, will skew the data toward low-flow conditions. Non-conditional synoptic sampling covers a wide spectrum of flow conditions, ranging from flood-pulses to flow-reversals. If river water is back-flooded into the spring, it is considered to be representative of the spring's water at that moment in time. The sample will be taken and analyzed regardless of water provenance (river or cave derived). The aquatic communities of the spring and related conduit must live in the waters, regardless the source, therefore the sample is representative of their environment.

The presentation of water quality data in the following discussion will be in two forms: statistical graphs (bar and whisker) and XY graphs depicting trends at selected sites of selected parameters. The four selected sampling sites for this document are: Light agriculture (Pike Spring, PSPS), Park/heavy agriculture (Echo River Spring, ERES), Heavy agriculture (Turnhole Spring area, THNS), and Park lands (Buffalo Creek Spring, BCGR).

DISCHARGE

Discharge depends, of course, upon precipitation events. The summer and fall months are traditionally characterized by low discharge, with higher discharge through the winter and spring (Figure 4). Overall the largest discharges during sampling occurred on April 10, 1991. On this date flood-pulse activity was high as the aquifer quickly responded to the rains of the previous day. This sampling round is of specific importance as samples were extracted near peak discharge times of the flood-pulses. Although other rounds saw relatively high discharges, samples were, as dictated by monitoring program, taken either well before or well after pulse peaks.

During the first nineteen months of the study a major back-flooding event was sampled on June 10, 1990. At this time all

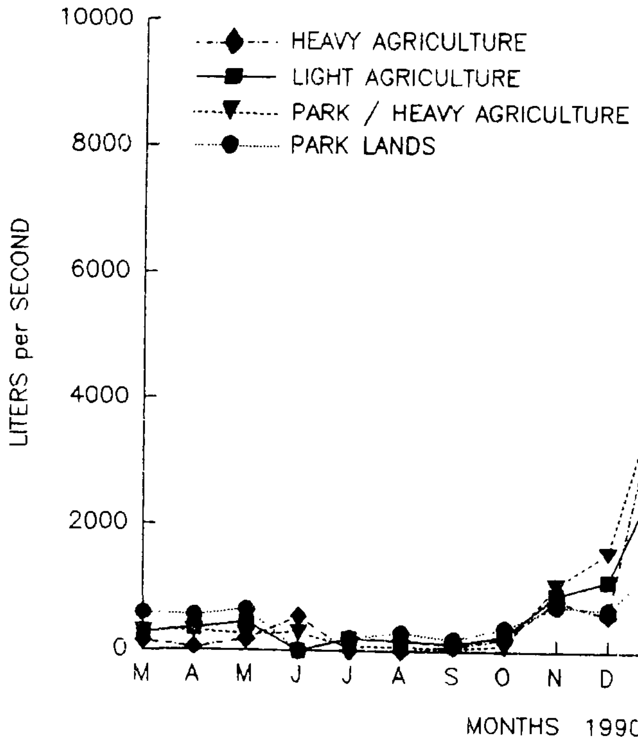

Figure 4. Discharge values, March 1990 through September 1991, of four monitoring sites.

springs, with the exception of THNS, were in a state of flow reversal - water from the Green River flowing back into the aquifer.

Notice that Echo River Spring is referred to as "Park/heavy agriculture". During times of high discharge, flow from the heavy agriculture basin is shunted through a high-level overflow route into the Echo River basin, which is normally recharged by park lands. When this route is activated, water quality in Echo River may, nearly instantaneously, degrade. Research in the next year will document the conditions needed to conduct flow through the overflow route.

TURBIDITY

Turbidity, correlative to the amount of suspended sediment in the water, is highly variable through all non-conditional synoptic sites (Figure 5). As expected, basins dominated by agricultural land-use, discharge more turbid waters than those dominated by undisturbed forests. Generally one would expect the higher turbidities associated with areas of high soil loss. Although the Turnhole basin (Heavy agriculture) contains by far the greatest area of tilled crop-lands, its turbidities, albeit high, were not the highest recorded; that honor goes to the light agriculture Pike Spring basin. Although containing far fewer

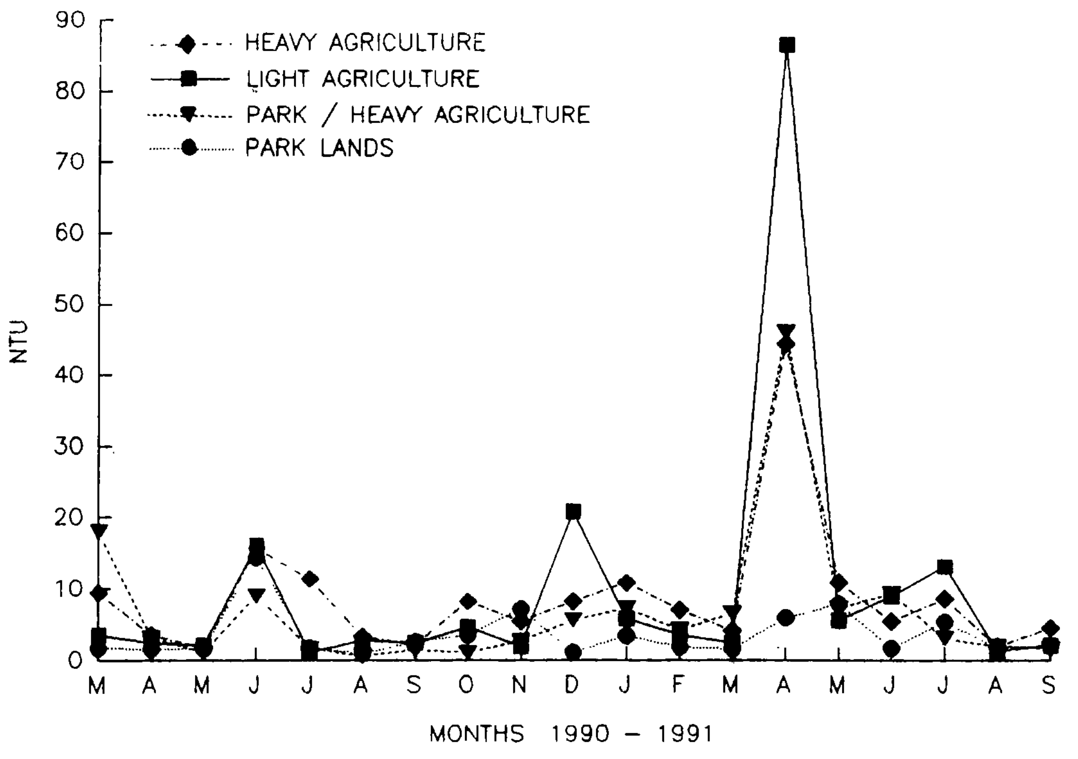

Figure 5. Turbidity values, March 1990 through September 1991, of four monitoring sites.

acres of tilled land, the rugged topography of the Pike Spring basin amplifies soil loss when disturbed.

Displayed in turbidity are the back-flooding and overflow described in the preceding section. The back-flooding event of June 1990 is evident in turbidity, as back-flooded springs display turbidities close to that of the Green River. The Echo River basin (Park/heavy agriculture) exhibits low turbidities, associated with low to moderate discharges when the spring is recharge by park lands, and high turbidities when the overflow route from the heavy agriculture basin is activated.

CHLORIDE

Chloride may be indicative of animal/human waste and oil field brines. Figure 6a shows the chloride concentration trends of the four selected sites, while Figure 6b demonstrates the mass flux of the chloride ions. Both graphs exhibit interesting data. Figure 6a shows elevated concentrations of chloride in the heavy agriculture basin.

Oil field brines seem the prime suspect for two reasons: presence of associated brine ions, and mass flux signatures. Within the headwaters of the basin, and adjacent to the Park City oil-field, is Parker Cave. On a low-flow conditional synoptic survey (September, 1990), Parker River (a stream passage within Parker Cave) had chloride levels of 1476.1 ppm. Further down-basin, Mill Hole chloride was 59.9 ppm, as the Parker River water was diluted. At the basin's terminal spring, chloride was further diluted to 31.6 ppm. Bromide and sulphate, also suggestive of brines, were found decreasing at similar rates at the same sites. Similar results were reported by Meiman (1989), and Quinlan and Rowe (1978).

The mass flux signature of chloride may also indicate brine contamination instead of animal waste. Figure 6a displays a variable, yet predictable pattern of chloride concentrations. One may assume that the chloride source is of relatively constant delivery as chloride concentrations are higher during low flow periods of summer and early autumn, and lower, more dilute, during the high flow periods of winter and spring. Figure 6b indicates an apparently dramatic increase on the mass flux of chloride during months of high discharge. This increase, some eight times the mass flux of low discharge periods, may not be as severe as it may seem. This variation may be normal, even for this relatively constant source parameter. One might expect a much greater (several orders of magnitude) rise in the mass flux signature if a source is released by run-off from a precipitation event.

A certain portion of chloride can be considered as a natural, background concentration. Observe the chloride trends of the Park land basin. Not only are chloride concentrations low

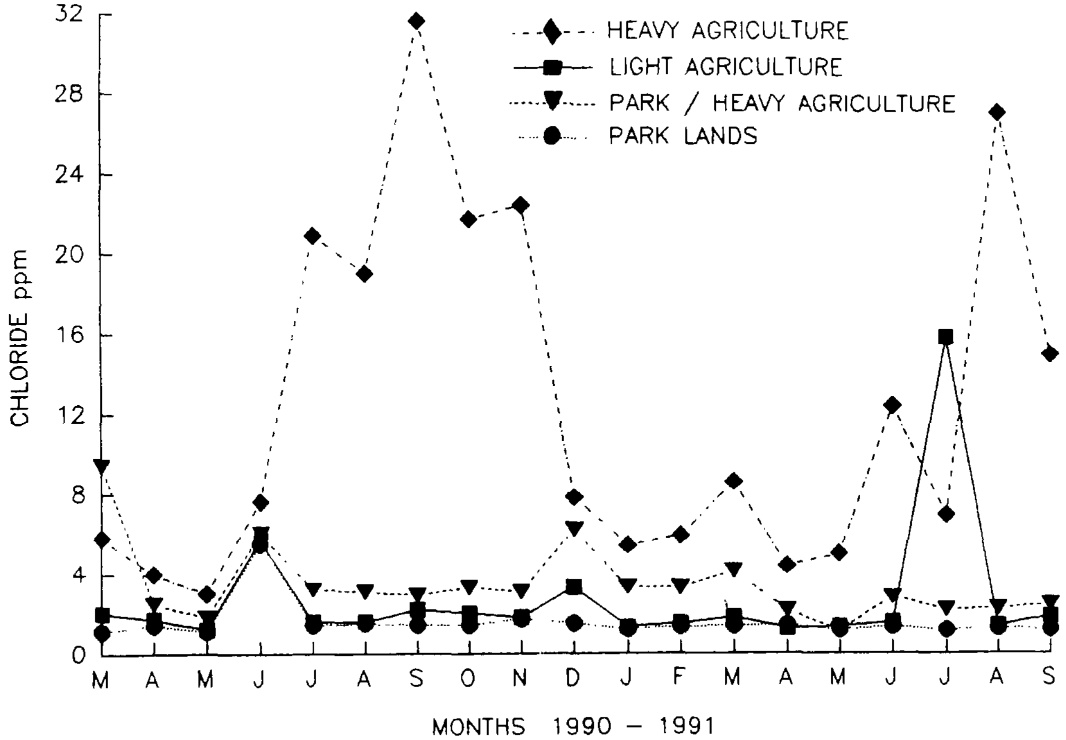

Figure 6a. Chloride concentrations, March 1990 through September 1991, of four monitoring sites.

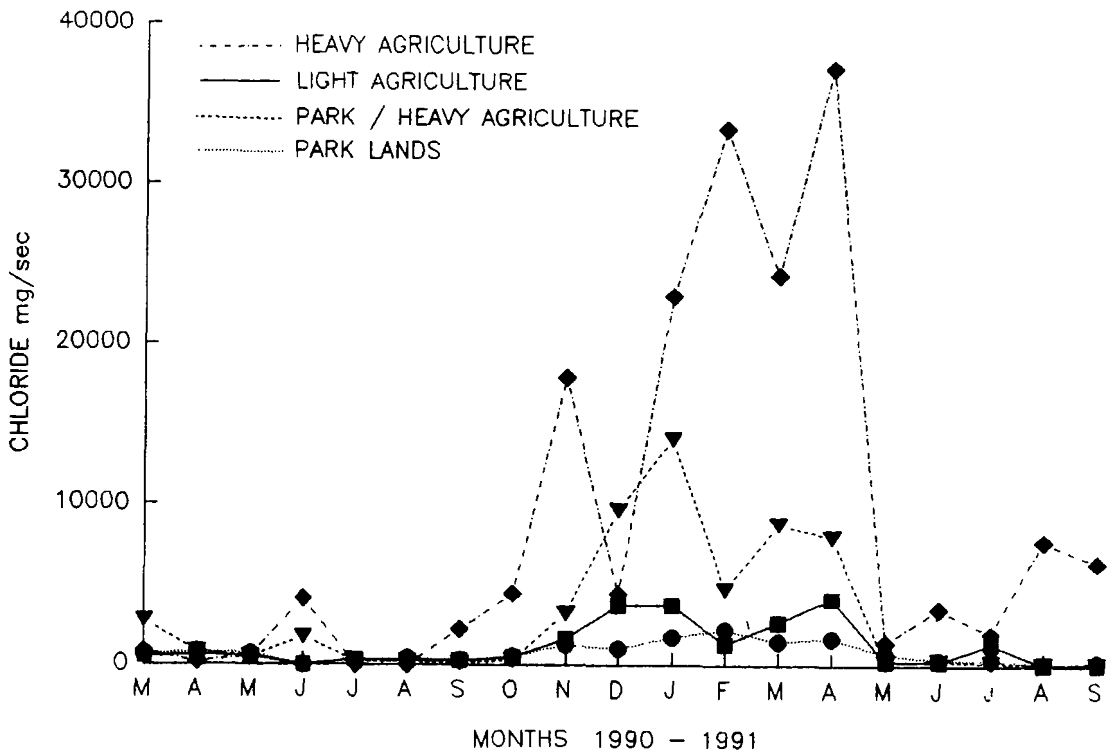

Figure 6b. Chloride mass flux, March 1990 through September 1991, of four sampling sites.

(Figure 6a), they remain at approximately the same mass flux throughout the year (Figure 6b). Upon closer examination, notice the slight increase in mass flux through the winter months. Although seemingly small and insignificant, the relative changes between seasonal mass fluxes in the Park land and the Heavy agriculture basins are very similar. This trend may be an inherent wave-form signature due to the vast increase in discharge.

Road salts as a potential chloride source must be recognized. Although the use of road salts have been prohibited within the park since 1987, they are used throughout several of the park's groundwater basins. The amount of road salt contributing to chloride levels found in parks waters is yet unknown. Since the first month of sampling there has not been a significant snow fall to warrant the use of much salt. Perhaps this unnatural source will be manifested in a "unique" mass flux signature, representative of seasonal application and recharge.

FECAL COLIFORM

Fecal coliform bacteria is found at all sampling sites (Figure 7). These bacteria are common in wastes of all healthy warm-blooded animals. By far, basins with high occurrences of dairies, feed-lots and urban areas are characterized by high levels of fecal coliform.

A certain amount of fecal coliform can be attributed to wildlife. Note that the Park land basin (BCGR), representative of pristine conditions, contains a fair amount of fecal coliform bacteria (mean of 67 colonies per 100 ml), and discharges a relatively stable 1.5 million colonies per second (not shown). Although the latter number may appear high, a single gram of feces may contain tens of millions to tens of billions of cells (Feachem, et al., 1983).

The heavy agriculture basin (THNS), with hundreds of homes without proper waste treatment facilities, and scores of dairies and feed-lots where live-stock waste flows as sinking-creeks into the aquifer yielded the highest overall fecal coliform levels. Feachem et al. (1983) reports that although fecal coliform density per gram of feces of man and livestock are comparable, a human may excrete 150 grams of feces per day compared to 15 to 20 kilograms for a cow. The highest flow weighted value, greater than 535 million colonies/second, was observed at this spring on April 10, 1991. As samples are taken on a set monthly date, regardless of weather or flow conditions, over the three-year period of monitoring some flood pulses are likely to be sampled. April 1991 was such an occurrence. It is important to note the relative temporal position within the flood pulse from which the sample was taken. A great amount of variance in parameter concentration may exist throughout high discharge periods of a flood pulse. It is not possible to tell from one sample its

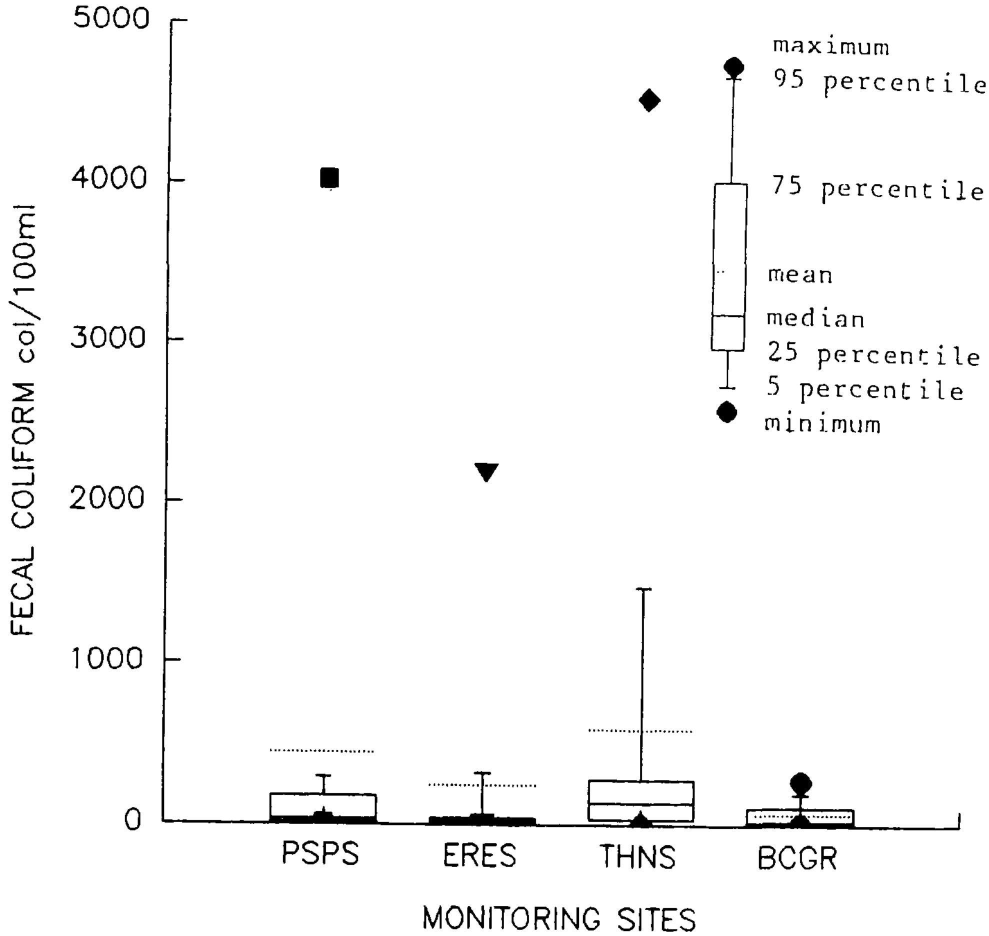

Figure 7. Statistical examination of fecal coliform bacteria levels, March 1990 through September 1991, of four monitoring sites.

temporal relationship to concentration or mass flux peaks of a particular parameter.

The chance occurrence of flood pulse activity coinciding with a predetermined sampling date tends to create a large variance in reported concentrations and mass fluxes of fecal coliform bacteria. A high variance may indicate, as in the heavy agriculture basin, the presence of large amounts of animal waste stored at the surface, awaiting release by a rainfall event.

Notice that low values dominate the data set in the park land/heavy agriculture basin (ERES). Occasionally these low levels of fecal coliform are interrupted by brief periods of very high concentrations, as expressed in the elevated mean and maximum values. Recall the overflow route mentioned earlier. During high-stage times, a portion of the bacteria-laden waters of the heavy agriculture basin are shunted into the relatively clean park basin.

TRIAZINE-CLASS HERBICIDES

The monitoring program indicates the presence of triazine-class herbicides (greater than 1 part per billion) within the surface and ground waters of the park. The occurrence of these compounds generally coincided with the peak application period. To avoid costly organic laboratory testing for these compounds, the program employs assay screening tests. Although gas chromatography analysis would indeed be desired, laboratory costs of a couple of sampling rounds would destroy the monitoring budget. Assay-screening can not be thought of as a quantitative analysis. It is used primarily as a "hit-or-miss" technique, with semi-quantitative values (ie, greater than 1 ppb).

The spatial and temporal occurrence of triazine-class herbicides reflect land-use, herbicide application periods, and perhaps the mechanism of transfer between the surface and subsurface (Figure 8). With the exception of a back-flooding of triazine-tainted river water, the only groundwater sampling site in which triazines were found was the heavy agriculture basin (THNS) spring. Additionally, triazines were only found in months (June 1990, June and July 1991) following peak application periods within the basin. Triazines are also found at both river sites following peak application.

For the remainder of the year no triazine-class herbicide residues were found in the sampled springs. Although rapid

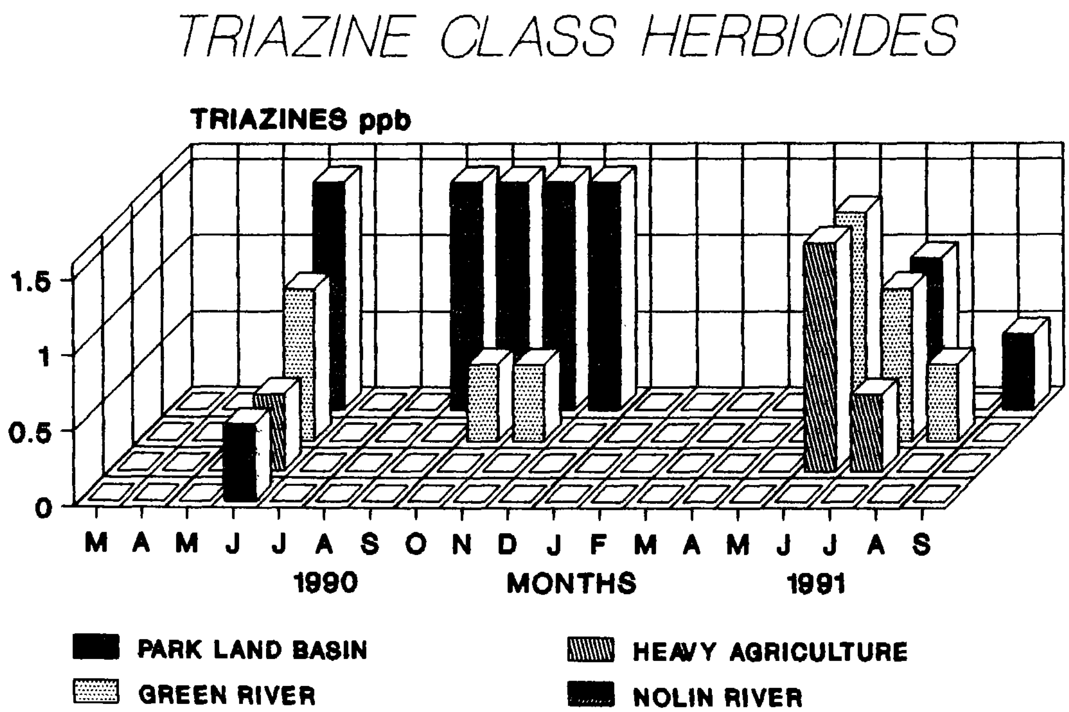

Figure 8. Temporal occurrences of triazine-class herbicides using immuno-assay methods, March 1990 through September 1991, at four monitoring sites.

transport of these residues through the karst system is expected, one may not assume that all, or even the majority of these compounds that will move through the aquifer have done so. Research in Iowa by Hallberg et al. (1985) found that although large amounts of herbicides are quickly transported through the karst system via run-off following rainfall, the bulk of these materials are slowly released through infiltration in low concentrations. It would not be surprising to see a similar pattern of pesticide transfer through the Mammoth Cave aquifer.

Aside from occurrence following peak application periods, triazine-class herbicides were found in both the Green and Nolin rivers in the fall of 1990, and possibly in the fall of 1991. Two scenarios may be possible: 1) There was a late application of these compounds in the fall, or, 2) The residues were slowly transferred through a less permeable media (clastic strata).

As triazines are applied as pre-plant or pre-emergence herbicides, there is no reason to believe that there were a late applications, as crops that receive triazines (corn, and to a lesser degree, soybeans) were near harvest.

River flood plains, with associated unconsolidated fluvial deposits, are favored lands for row-crop production. It may be possible that these persistent compounds may: 1) become entrained in run-off shortly after application, and 2) slowly infiltrate through the fluvial materials and leached into the river following fall rains. The "half-life" of these compounds (3-12 months) is certainly sufficient to cause such persistency.

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BIOGRAPHIC SKETCH

Joe Meiman, a Kentuckian, is currently the Hydrologist at Mammoth Cave National Park. His B.S. in Geology (1985) and M.S. in Geology/Hydrogeology (1989) were earned at Eastern Kentucky University. At the present he is kept busy by directing hydrologic research at the park, which includes: water quality monitoring, dye-tracing, three-dimensional schematic karst aquifer modeling, research and development of monitoring equipment, and the general quest of scientific knowledge.

The Effects of Recharge Basin Land-use Practices on Water Quality at Mammoth Cave National Park, Kentucky

Joe Meiman

Your study is interesting and has some novel, admirable interpretations of the data. Having said that, however, I am surprised that samples were not also collected frequently (perhaps at 1- to 4-hour intervals) and analyzed at several sites throughout several storm events. This is more important than monthly non-conditional synoptic sampling--if you want to understand what is happening in the groundwater basins. The 3-year monthly sampling program is incapable of having any significant statistical validity because it is not tied into analysis of storm events and has too short a duration. There already exist hundreds of analysis of samples collected weekly to monthly at many of your sites by Jack Hess (Ph.D. dissertation, 1974), the U.S. Geological Survey, and others. Before your sampling began, the National Park Service was repeatedly advised in writing, by specialists in monitoring water quality in karst terranes of the existence of such data, and that sampling of storm events could yield more useful data and insights than monthly sampling. Why has the Park Service ignored sampling and characterization of storm events?

As I stated in the presentation, monthly non-conditional synoptic sampling is only a portion of the Park's water quality monitoring program. Indeed, storm event sampling will comprise a large measure of the monitoring during the coming year, as was discussed. I am currently using storm-event (flood-pulse) data, recorded as often as every two minutes, in our present study. As work in this area continues, we will report our results when sufficient flood-pulse water quality data have been collected. I am aware of the excellent study by Dr. Hess, who primarily focused upon carbonate chemistry variations during flood-pulse activity. We have expanded our scope of parameters to include bacteria, nutrients and brine-associated ions. Synoptic data generated by the studies of the U.S. Geological Survey during the 1950's and 1960's are rather spatially limited. We have increased sampling sites to capture data related to the myriad of lands uses within the recharge basins. Past works serve as skeletal datums of which data of the current study will be compared. Temporal comparison is the essence of monitoring. Be assured that the National Park Service has not entered this program without careful consideration of past work in the region, or the dynamic nature of the karst aquifer.

THE SINKHOLE COLLAPSE OF THE LEWISTON, MINNESOTA WASTE WATER TREATMENT FACILITY LAGOON

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ABSTRACT

On February 20, 1991, city workers discovered a sinkhole collapse in the Lewiston, MN waste water treatment facility (WWTF) lagoons. The collapse apparently occurred during the preceding few days and drained an estimated 7.7 million gallons of partially treated effluent into the local ground water system. A temporary dike was constructed to isolate the sinkhole from the rest of the lagoon. Subsequent, ad hoc testing for coliform bacteria and nitrates did not detect evidence of effluent from the lagoon in nearby residential wells. Following a shallow (20 foot penetration) geophysical investigation using ground penetrating radar and an electromagnetic survey, the city decided to fill the sinkhole and to erect a dike around the collapse. The collapse was repaired in May, 1991 and the lagoon returned to full operation.

The 1991 Lewiston collapse follows the nearby, 1974 and 1976 collapses of the Altura, Minnesota WWTF lagoon (Liesch, 1977; Alexander and Book, 1984). Two of the 7 to 10 WWTF lagoons constructed on the Ordovician Prairie du Chien Group carbonates in the southeastern Minnesota karst terrain have catastrophically failed in less than 20 years. That corresponds to a failure rate of over 20% for these million dollar WWTFs -- so far. The federal programs that cost-shared the bulk of the construction expenses for these WWTFs no longer exist. The cost of potential damages, remediation, and/or replacement of these WWTFs falls directly on the state and local units of government.

INTRODUCTION

Southeastern Minnesota is an active karst area. Geomorphic features associated with the karst include sinkholes, enlarged joints, numerous springs, disappearing streams, cave systems, and dry valleys. There are problems with ground-water quality ranging from occasional high levels of selected parameters to chronic sub-standard drinking water conditions in the hydrogeologically sensitive area.

The region is characterized by farms, small towns, and a few moderate-sized cities. Many of the community centers have waste treatment facilities that consist of a series of settling ponds, or lagoons. Lewiston, Minnesota is one of the small towns located within this karst region. This paper documents the failure of one of Lewiston's ponds due to the instantaneous collapse of a sinkhole.

PHYSICAL SETTING

Topography

Lewiston is located in southeast Minnesota, in Winona County (Figure 1). The region surrounding Lewiston is characterized by gently rolling hills and swales with local relief of about 20 m (Figure 1). Sinkholes and dry valleys are evident at the surface. A very thin soil, ranging in thickness from 0 to about 15 m, covers the bedrock. The source material for the soil is both residuum and/or glacial tills and loess.

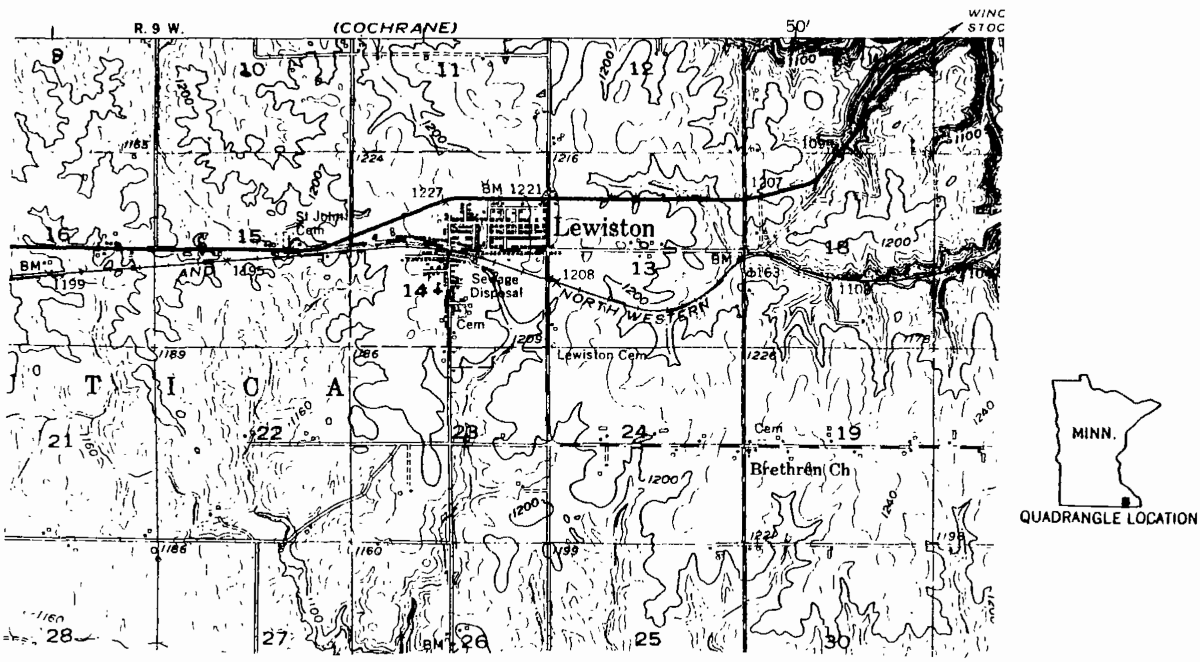


Figure 1. Portion of the topographic map in the vicinity of Lewiston, Lewiston, Minnesota Quadrangle.

Climate

At present, the area has a temperate climate, with a mean annual temperature of 7.6^0, and an annual precipitation of 75 cm (NOAA, 1978). This region has experienced climatic changes, most recently, the changes that resulted in Pleistocene glaciation.

Geology

The region is underlain by a series of lower Ordovician and Cambrian sandstone and carbonate units (Figure 2). The units were deposited in nearshore to shallow-sea environments, and exhibit typical vertical and lateral facies changes associated with sedimentation during Transgression and regression of the shallow sea. The strata dip gently to the southwest towards the center of the Hollandale Embayment (Mossler and Book, 1984). The units of most concern are the Jordan Sandstone and the Oneota Dolomite.

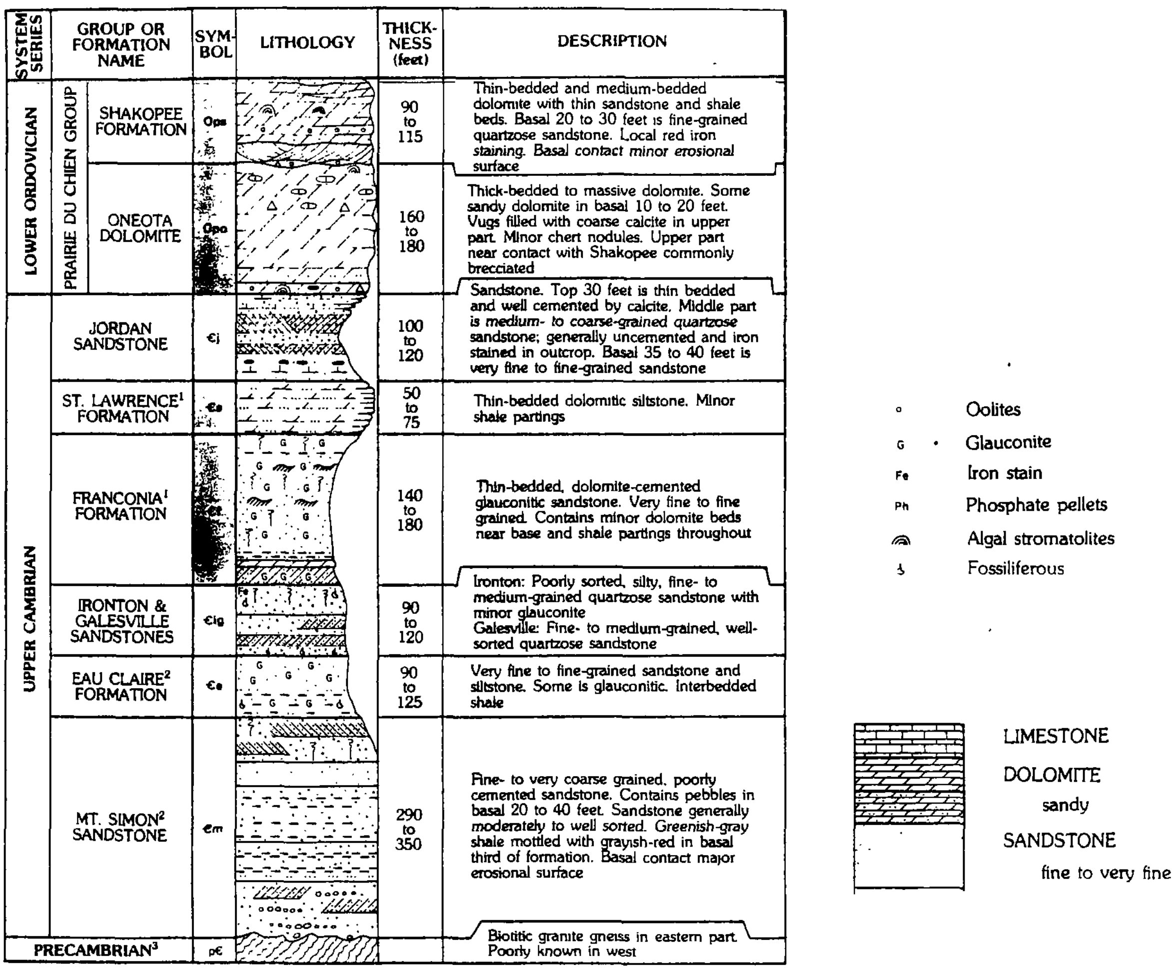


Figure 2. Generalized stratigraphic section in the Lewiston, Minnesota area. Adopted from Mossler and Book (1984).

The Jordan Sandstone is Cambrian in age and averages 30 m in thickness. The Jordan is a massive, upward-grading, fine- to coarse-grained friable sandstone. Upward in the unit, it becomes progressively more indurated with carbonate and siliceous cements, first forming lenses and concretions and then well-bedded, highly lithified strata.

The Ordovician Prairie du Chien Group conformably overlies the Jordan (Figure 2). The Prairie du Chien is composed of the Oneota Dolomite and upper Shakopee Formation. The Oneota Dolomite is about 60 m thick, and is fine- to medium-grained, thick- to thin-bedded to massive, with calcite-filled vugs in the upper portion, and minor chert nodules throughout the unit (Mossler and Book, 1984). Both drill cores and outcrops reveal that the dolomite is highly jointed and has undergone extensive solution. The dolomite is vuggy to cavernous particularly in the upper portion.

The Shakopee Formation is subdivided into the lower New Richmond Sandstone member and the upper Willow River Dolomite member. The latter is not present in the area and is not discussed further. The New Richmond Sandstone of the Shakopee Formation is a fine- to medium-grained quartzose sandstone with infrequent interbedded medium-grained arenaceous carbonate beds. This sandstone unit averages about 6 m in thickness, is friable, extensively jointed, easily eroded, and does not form many outcrops.

Hydrology

Surface flow is in small headwater channels of the Whitewater and Root Rivers. The channels are characterized by meander development and easily erodible banks. Some surface run off flows into sinkholes. Regional ground-water flow is east-northeast toward the Mississippi River. Local ground-water flow is toward discharge points such as small tributaries or springs.

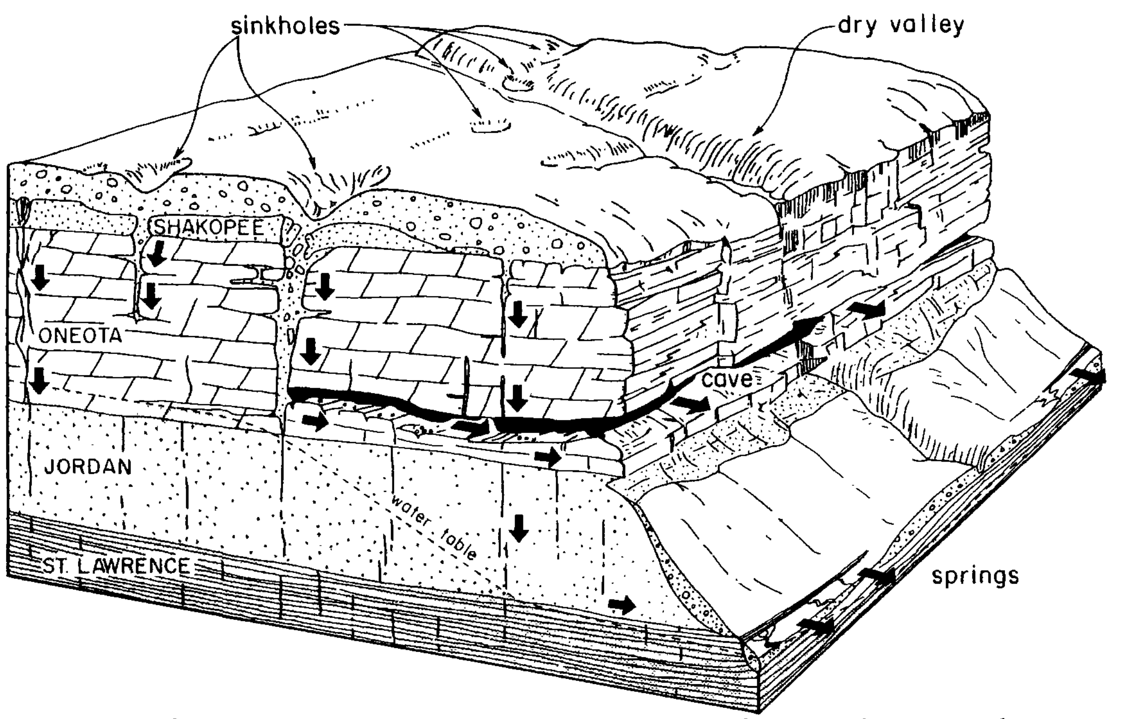
Joints are common throughout the Jordan Sandstone and springs in the well-lithified portions tend to discharge directly from joints. In the more friable lower part, springs are often a combination of discrete flow from joints and diffuse flow from numerous seeps. The Jordan is a major source of water for wells in the area.

Only a few springs, confined to discharge from well-developed joints, have been mapped in the Oneota. Few wells in the area rely solely on the Oneota as a water supply. However, many older wells are open holes through the Oneota. The New Richmond Sandstone member of the Shakopee Formation has a few springs which emerge form the New Richmond/Oneota contact. The New Richmond is not a significant aquifer in the area.

Karst Features

Numerous karst features such as sinkholes, enlarged joints, springs, dry valleys, and small caves have formed in the Oneota Formation of the Prairie du Chien Group (Figure 3). Sediment-filled solution cavities are common features in outcrops and quarry walls that expose the Oneota Dolomite. The karstification of the Oneota probably began during the Ordovician and has continued intermittently until the present. The region would be classified as fluviokarst according to the scheme used by Sweeting (1973) because both karst and fluvial processes have contributed to the development of the features that are evident, or that are being exhumed.

Sinkholes are by far the most dominant karst feature. Historically, if the holes are left in the natural state, they are either fenced-in and left to be naturally vegetated, or several have been used as backyard landfills. In the past, several have been filled with debris and soil and then used as farm land. Many of the sinkholes in the area have developed catastrophically, often in the spring of the year in response to unusually wet conditions. It appears that the sinkholes develop through the New Richmond Sandstone into the underlying Oneota Dolomite.



Typical relationships in Winona County between karst and topography

Figure 3. Block diagram of karst landforms in southeast Minnesota (Dalgleish and Alexander, 1984).

Lewiston, and the immediate vicinity, were classified as high probability of sinkhole development by Dalgleish and Alexander (1984). The classification was based on the observed density of sinkholes, together with information on the bedrock geology, surficial geology, and hydrogeology. Dalgleish and Alexander (1984) conclude that the carbonate bedrock is the primary control on sinkhole formation. Secondary controls include the type and thickness of the overburden, and the depth of the water table. Areas where the Oneota Dolomite is overlain by the sandstone member of the Shakopee Formation, such as in the vicinity of Lewiston, are the most susceptible to sinkhole development. Fractures in the noncalcareous sandstone act as conduits to preferentially direct surface water into the Oneota.

THE SINKHOLE COLLAPSE OF THE LEWISTON WASTE WATER TREATMENT FACILITY LAGOON

Background

The Waste Water Treatment Facility at Lewiston, (population ~1300) is constructed in an area that overlies the New Richmond Sandstone member of the Shakopee Formation and the Oneota Dolomite, and has less than 30 m of regolith. The Waste Water Treatment Facility consists of a series of settling ponds or lagoons which is commonly known as a "natural" treatment system (Figure 4). These types of systems are common in southeast Minnesota. The one at Lewiston is about 20 years old. Exposure of the waste water allows oxygen and sunlight, together with microorganism to "treat" the effluent after initial screening for large-sized solids. Machines or chemicals are not used. The treated water is then discharged to a surface-water channel. This natural purification takes about 6 months.

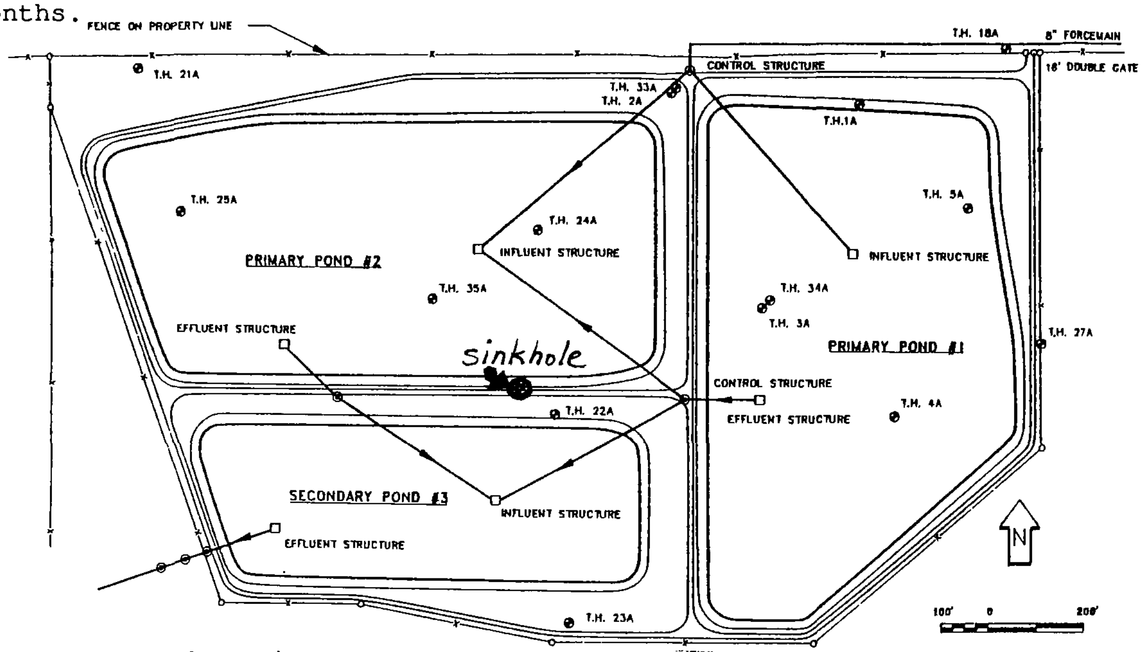


Figure 4. Schematic of the Lewiston WWTF (From Braun, Intertec, 1991)

According to the superintendent of the Lewiston's sewer and water system Lewiston had wanted a mechanical system 20 years ago, but was denied the request by the Minnesota Pollution Control Agency. Supposedly, the decision was based on the consensus that small towns could not afford the upkeep and the operating expenses of a mechanical plant (Rochester Post and Bulletin; February 27, 1991). Today, that opinion is not held by any state government agency. The change of opinion is based not on a town's new-found ability to afford a plant, but because of the ground-water quality problems associated with a karst terrain, and the documented failure of a similar lagoon system in Altura, Minnesota (Alexander and Book, 1984), which is about 10 km northwest of Lewiston.

Sinkhole collapse

On February 20, 1991, it was discovered that a sinkhole had opened on the edge of sewage lagoon Number 2 (Figure 4) at the Waste Water Treatment Facility at Lewiston, Minnesota. The sinkhole collapse caused a break in the dike enclosing the lagoon. The collapse left a hole that was approximately 12 m in diameter and 2-4 m in depth (Figure 5).

Figure 5. Photo of dike-side of sinkhole, on February 20, 1991. View is northwest. Photo courtesy of R. Dunsmoor, Winona County.

It is estimated that the collapse occurred on or about February 14, 1991. According to the records of city workers, approximately 7.7 million gallons of semi-treated sewage effluent were lost from Lagoon Number 2. The loss occurred over several hours to perhaps a day. The effluent entered the ground through a conduit at the bottom of the sinkhole. The waste water had been in the lagoon only about two months, and probably still contained bacteria and/or viruses because it was covered with ice which would prevent sunlight and heat from destroying them.

The sinkhole collapse at Lewiston has striking similarity to the collapse at the Altura Waste Water Treatment Facility as documented by Alexander and Book (1984). Both collapses formed in the Oneota Dolomite where it was overlain with the basal sandstone member of the Shakopee Formation. At both locations, sinkhole collapse was catastrophic. One of major differences is that the failure of the Altura lagoon was due to several sinkholes in the bottom of the lagoon, whereas, the failure of the Lewiston lagoon was due to a single sinkhole collapse near the edge of the lagoon which led to breaching of the dike.

Response to the problem

The water level in Lagoon Number was 2 was lowered and continued to be monitored so that further semi-treated water did not spill over into the sinkhole. A dam was built around the sinkhole in order to prevent surface run off from entering the hole.

Water-quality tests were performed on the city wells and on 11 private wells in the area. The results did not detect contamination from the effluent. Residents were advised to drink water from hot water heaters that had been cooled, until they could have their well tested. They were further advised to chlorinate their wells, and drink bottled water if they had any concerns.

Remediation of the problem

The city hired a consulting firm in early March to determine the size of the sinkhole and propose short term remediation. A shallow (6 m penetration) geophysical investigation used ground-penetrating radar and an electromagnetic survey to determine the limits of the observed sinkhole and possible fractures in the vicinity. An independent proposal to run a dye trace form the sinkhole in order to determine ground-water flow patterns was not adopted by the city. This was due in part to the concern of liability.

By mid May 1991, it was decided by city employees that the best response to the sinkhole collapse was to repair the dike and seal the sinkhole. Beginning May 21, 1991, the site was cleaned-up, and new dike was created about 15 m from the surface expression of the sinkhole. According to city employees, the sinkhole was excavated to within 1 m of bedrock. The hole was then filled and sealed by May 24, 1991.

POTENTIAL FAILURE OF OTHER WASTE WATER TREATMENT FACILITIES

The Minnesota Pollution Control Agency has compiled a list of towns in southeast Minnesota with waste-water pond facilities similar to that at Lewiston. The screening for the list included those sites which are situated over karstic bedrock and had less than 30 m of soil or till above the bedrock. The initial list, which is being refined at this time, includes 14 sites. Of those 14, 10 are considered to have high potential for failure, and 4, low potential. Of the 10 that have high potential for failure, 2 have had failures within the past 20 years. This corresponds to a failure rate of about 20%.

Costs for potential damages, repair and/or remediation for these Waste Water Treatment Facilities is now the responsibility of the local units of government. The Federal programs that cost-shared the bulk of the original construction costs have been severely reduced or phased out. It is recognized that the use of a lagoon system in a karst region can lead to catastrophic failures and potential health concerns. However, the sealing of the sinkholes not only in, or next to, sewage ponds, but in other sensitive areas, is often the remedial method of choice due to economics. A new sewage treatment plant would cost about 1.5 million dollars which is an exorbitant financial burden for a small town. These small towns are forced either to spend millions of dollars on new construction or risk potential liability suits. Neither choice is good.

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BIOGRAPHICAL SKETCHES

Nancy O. Jannik is an Associate Professor in the Department of Geology at Winona State University. She has a B.S. in geology (1976) from the College of William and Mary, a M.S. in geology (1980) from Rutgers University, and a Ph.D. in geosciences/hydrology (1988) from the New Mexico Institute of Mining and Technology. Her current research involves determining aquifer characteristics, aquifer interactions, and the inter-relationships between surface and ground water in the karst terrain of southeastern Minnesota. Her full address is as follows.

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Erosion and Sedimentation Control Methodologies for Construction Activities over the Edwards Aquifer in Texas

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I. Abstract

Through its implementation of the Edwards Aquifer Program, the Texas Water Commission strives to preserve the quality of water produced from this karst aquifer. The Texas Water Commission (TWC) has rules regarding development over the Edwards Aquifer Recharge Zone which require erosion and sedimentation controls.

During construction-related activities, erosion and sedimentation are the primary mechanisms for aquifer contamination. Temporary erosion and sedimentation control structures are necessary to ensure that suspended sediments in stormwater remain within the construction area. These control structures filter sediments and include hay bales, filter fences, rock berms, and brush berms.

Sediments that are deposited in streams and lakes impact benthic organisms, reduce storage capacity of lakes, and reduce water quality. Sedimentation also impairs natural groundwater recharge which results in the reduction of base flow in creeks, destruction of seeps and springs, and loss of groundwater supplies.

Erosion occurs when natural vegetation that binds soils together is disturbed. During rainfall events, natural vegetative cover acts as a filter to remove pollutants, and reduces runoff velocity. During construction related activities, natural vegetative cover is disturbed. As a consequence, Total Suspended Solids (TSS) in stormwater runoff is greatly increased.

Stormwater runoff from urban areas has increased levels of nutrients (nitrogen, phosphorus, etc.) from fertilizers, and pesticides. Other pollutants come from asphalt roofing materials, sewage disposal facilities, and other forms of non-point source pollution. Stormwater runoff from roadways

typically has increased levels of lead, total petroleum hydrocarbons, oil & grease, and metals.

The Edwards Aquifer Program requires permanent sedimentation control structures which improve the quality of stormwater runoff and provide protection of water resources. Permanent control structures include sand filtration and sedimentation basins and hazardous materials traps.

II. <u>The Edwards Aquifer Protection Program in Texas</u>

Effective March 21, 1990, the Texas Water Commission (TWC) adopted revised rules requiring the preparation and submission of a Water Pollution Abatement Plan (WPAP) for any regulated developments over the Edwards Aquifer recharge zone in Texas. The purpose of the rules is to regulate activities having potential for polluting the Edwards Aquifer. The activities addressed are those that pose direct threats to water quality. Nothing in the rules is intended to restrict the powers of the TWC or any other governmental entity to prevent, correct, or curtail activities that result or potentially result in pollution of the Edwards Aquifer. Each WPAP must contain, at a minimum, the following information:

1) name, address, and telephone number of owner, agent, developer and contact persons;
2) general location map which identifies the limits of the recharge zone, and the location of the proposed development;
3) site plan with proposed development, flood plain information, location of known wells, existing and proposed drainage patterns, and identification of any known significant recharge features;
4) detailed assessment of area geology which includes: geologic map at site plan scale; stratigraphic columns; narrative description of surface geology, soil profile and units; and, a narrative description of all significant recharge features; and
5) detailed technical report which includes: description of the proposed development; volume and character of wastewater and stormwater expected to be generated from the site; description of methods to be taken to prevent pollution of stormwater originating on-site and upstream of the site; description of methods to be used to prevent pollutants from entering recharge features; and a description of method of disposal of wastewater.

Upon receiving final approval of the WPAP, the applicant must deed record the approval in the County Courthouse. Any modifications to the plans must be submitted to and approved by the TWC prior to construction. The TWC must be notified prior to initiation of construction. If any significant recharge features are identified, all construction activities in the immediate area must be halted upon discovery until protection plans can be developed, reviewed and approved by the TWC.

All organized sewage collection systems must also be approved by the TWC prior to initiation of construction. At a minimum, the plans must comply with the following design criteria.

1) all manholes must be watertight with watertight rings and covers. Alternate means of venting must be provided for every third manhole. All manholes must also be tested for water-tightness prior to use.
2) lift stations must be designed to ensure that bypassing does not occur. All lift stations must have audible and visual alarms, as well as, auto-dial telemetry to notify a series of locations with 24-hour response coverage.
3) all gravity and pressure sewage lines, including private service laterals and manholes, must be inspected and tested to ensure that no leakage occurs after construction.
4) all wastewater systems must be tested every five years to determine the types and locations of structural damage and defects such as offset or open joints, or cracked or crushed lines that could allow exfiltration to occur.

Additional requirements provide that new or increased discharges of treated effluent that would create increased loading of treated wastewater are prohibited over the recharge zone. Land application systems that rely on percolation are prohibited. However, irrigation systems may be allowed and are reviewed on a case-by-case basis.

All WPAP's and sewage collection system plans must include erosion and sedimentation control plans. These plans shall provide designs for control structures that ensure stormwater contaminated during construction activities is not allowed to discharge from the site without treatment. Some of the effects of uncontrolled erosion and sedimentation from construction areas include:

1) changes in habitat and localized natural land uses;
2) increased flood plain depths;
3) reduced volumes of water storage in rivers and lakes resulting in increased down-stream flooding;
4) increased eutrophication rates from excessive nutrient loadings; and
5) loss of top soil.

III. <u>Controlling Erosion and Sedimentation</u>

During construction related activities erosion is the primary concern. The major part of the erosion potential exists between the time when the native vegetation is removed and when the site construction is complete and the vegetation is restored. These impacts can be minimized with the proper design, installation, and maintenance of temporary erosion and sedimentation control structures. In addition to temporary control structures, impacts can be reduced by limiting the amount of disturbed areas by identifying a construction area and limiting access to this area. Temporary erosion controls can be designed and installed within the construction areas to provide four basic functions:

Diversion - by diverting off-site flows around disturbed areas the amount of runoff collecting sediment and the volume of water that must be treated can be minimized. Diversions can also be used to direct runoff from drainage areas to a location where sedimentation can occur more cost effectively.

Ponding - by installing controls that reduce the runoff velocities the sediments are allowed to settle out and can be removed from the pond on a routine basis. Ponding is one of the primary methods of sediment removal used during and after construction.

Filtration - when stormwater flow is routed through a filtration structure, soil particles are trapped on the filter media, and sediment settles out up-gradient of the filter structure. Filter structures can be used to capture the fine portions of the sediment and are used extensively during construction.

Flow spreading - by installing controls that change the runoff characteristics from concentrated to sheet flow, the velocity of the runoff is reduced which increases local sedimentation and minimizes erosion.

The effectiveness of any temporary erosion and sedimentation control structure is dependent upon proper installation and adequate maintenance. The following guideline can be used to approximate the type of structure that is appropriate for various conditions.

Type of Structure	Reach Length	Drainage area	Slope
Hay Bales	< 100 feet	1/2 Acre	0 - 10%
Silt Fence	n/a	2 Acres	0 - 10%
	< 200 feet	2 Acres	10 - 20%
	< 100 feet	1 Acre	20 - 30%
	< 50 feet	1/2 Acre	> 30%
Brush Berm	n/a	1 1/2 Acres	0 - 10%
	< 150 feet	1 Acre	10 - 20%
	< 75 feet	1/2 Acre	20 - 30%
			> 30%
Rock Berm	500 feet	5 Acres	0 - 10%

For drainage areas greater than 5 acres, the design of structural features are reviewed on a case-by-case basis. Approval is based on the anticipated quantity and quality of stormwater, and the location of any critical environmental features downstream.

Hay bales are an inexpensive and readily available control structure; however, they are difficult to maintain, must be replaced following every rainfall, and are ineffective for any drainage area larger than 0.5 acres. The TWC does not recommend the use of hay bales except in limited areas where construction times will be short and rainfall is not anticipated.

Filter fabric may be required within the rock berms if determined necessary by the TWC. These structures have been proven effective in steep canyons, above permanent springs, pools or other environmentally sensitive areas. This type of structure combines the removal efficiency of silt fences with the structural stability of rock berms.

IV. Design Calculations

Temporary erosion and sedimentation control plans must include calculations to verify that the proposed structures are adequately designed. The calculations involve establishing a storm flow volume for the drainage area for each structure, and an appropriate barrier flow-through rate. The computational procedure for establishing a flow-through rate includes several calculations such as coefficient of roughness, time of concentration, and rainfall intensity.

Using the "Rational Method" ($Q = C * i * A$) the volume "Q" of flow can be calculated for various drainage areas with the following parameters:

1) determine the drainage area "A" for each structure or drainage basin based on the site plan;
2) determine rainfall intensity "i";
3) estimate the runoff coefficient "C" for a two year rainfall storm for each specific soil type and channel slope;
4) compute peak flow rate of runoff "Q" for a two year storm using the rational method;
5) determine frontal area for flow "A_f" at the barrier; and
6) determine the flow through rate at the barrier using the following "$q = Q / A_f$"

Based on the flow through rate "q" the proper structure is chosen from the following Table 1:

Table 1

	Maximum Flow Through Rate (Gal/Min/SqFt)	Trenched or Secured	Recommended Maximum Drainage Area	Recommended Maximum Slope Length
Sheet Flow				
Hay Bales	5	Yes	.5 acre	50 feet for slopes exceeding 10%
Filter Dikes	20	Yes	1 acre	50 feet for slopes exceeding 10%
Brush Berm	40	Yes	2 acre	
Silt Fence	40	Yes	2 acre	
Rock Berm	60	No	5 acres	
Diversion Dike, etc.			5 acres	
Swales			5 acres	
Concentrated Flow				
Stone Outlets	40	Yes	5 acres	
Diversion Dike, etc.			5 acres	
Swales			5 acres	
Rock Berm	60	Wire Mesh Sheath	5 acres	
Slope Drain			5 acres	
Sediment Traps			5 acres	
Sediment Basins			100 acre	
Stream Flow				
Rock Berm		Wire Mesh Sheath	+5 acres	
Sandbag Berm		No	+5 acres	

V. Additional Structures

Temporary erosion and sedimentation control plans should also provide for stabilized construction entrances, equipment and petroleum storage areas, and spoils disposal sites. An inspection program and a maintenance plan and schedule should also be included.

Stabilized construction entrances consist of a stabilized pad of open graded rock located at any point where traffic may be entering or leaving a construction site to a public right-of-way, street, alley, etc. The purpose of a stabilized construction entrance is to reduce or eliminate the tracking or flowing of sediment onto public roadways where they contribute to non-point source pollution.

Equipment storage areas, and temporary fuel and oil storage facilities should be designated on the site plans. All construction vehicles should be parked in protected areas when not in use, and all maintenance of vehicles should occur within these limits. These areas should be placed at topographical highs, or berms should be provided to divert stormwater flow around the site. Structural controls must be placed downgradient of the site to filter stormwater runoff. Special precautions should be taken to prevent the spill of petroleum products from storage facilities or from emergency maintenance activities. General scheduled maintenance is not allowed on-site.

Spoil disposal and material storage areas should also be designated on the site plans. Spoils material from the construction activity, sediment removed from temporary protective structures, and construction material (backfill, sand, base material, etc.) must all be stored in these areas. These areas should be protected from stormwater run-on. Stormwater discharges from the site should be filtered to remove potential pollutants.

Temporary erosion and sedimentation control structures should be inspected routinely to ensure effective control of stormwater. Weekly inspections should be conducted of the entire area, and inspections should be conducted within 24 hours of any rainfall event. Inspection forms should be developed and should be maintained at the construction site.

Typically, the contractor should be required to routinely clean paved surfaces with appropriate power brooms, or vacuum devices to minimize the amount of sediment within the construction area.

A temporary erosion and sedimentation control maintenance plan should be developed prior to initiation of construction. These maintenance plans should provide a schedule for the removal and replacement of temporary structures that are no longer effective. Provisions should be made for the removal of silt, and sediments following rainfalls. Troublesome areas should be identified and evaluated for improved structural controls.

VI. Cooperative Agreements

The TWC also enters into Cooperative Agreements with other State agencies to ensure that the agencies involved in development activities can be leaders in environmental protection.

In September 1991, the Texas Department of Transportation and the TWC entered into a cooperative agreement regarding the construction of a major highway intersection over the Edwards Aquifer. The agreement required that the Texas Department of Transportation comply with all rules and requirements for the submission of a Water Pollution Abatement Plan prior to construction. The agreement also provided additional specific requirements applicable only to the development at this location, and included the following.

1) Provide a detailed temporary erosion and sedimentation control maintenance plan and schedule that would establish guidelines for inspection of structures, replacement of inadequate or damaged structures, and increased protection of any impacted areas.
2) Restore and maintain vegetative cover prior to completion of construction.
3) Restrict construction vehicle access from unprotected areas, provide routine street cleaning and employee parking in stabilized areas.
4) Designate contact persons for the TWC, Texas Department of Transportation, and the contractor.
5) Develop an interagency training program.
6) Develop a remediation plan and schedule in the event an impacted area is identified.

The TWC also encourages input from other entities such as municipalities, river authorities, conservation districts, and environmental groups. Citizen monitoring can also provide valuable information regarding impacted areas and problem areas within existing developments.

VII. Estimated Annual Pollutant Load

The amount of total suspended solids, total phosphorus, and oil and grease that may be discharged from a site before and after development can be approximated. The annual pollutant load is the pollutant concentration times the annual runoff volume. The following calculation can be used to estimate the annual pollutant load for a tract of land:

$$L = A \times RF \times R_v \times 0.2266 \times C, \text{ where:}$$

L = Annual pollutant load in pounds
A = Area of tract in acres
RF = Annual rainfall amount in inches
R_v = runoff to rainfall ratio
$R_v = 0.05 + (0.009 * IC)$ where:
IC = impervious cover for site in percent
C = Pollutant concentration in milligrams per liter

	Background "C"	Developed "C"
Total Suspended Solids	48	130
Total Phosphorus	0.08	0.26
Oil and Grease	0	15

The area used in the annual pollutant load calculation for total suspended solids and total phosphorus should represent only that portion of the tract which is to be developed, not the entire site area. Typically, this is the limits of construction for the area. For oil and grease, the load calculation should only use the paved area.

VIII. Permanent Erosion and Sedimentation Control

Sedimentation basins can be effective at removing suspended solids from developed areas but are very limited in effectiveness of removing dissolved solids. The basic design of this type of basin is to capture and isolate a percentage of the rainfall or a certain volume of stormwater, and discharge the flow over a vegetative strip over a prolonged period of about 24 hours. A design to capture the first 1" of stormwater flow will generally be able to remove about 50% of the total suspended solids, 15%-20% of the total phosphorus and about 8% of the oil and grease content of the stormwater.

Sand filtration basins have been used in the Austin and Central Texas area to effectively remove suspended solids, bacteria, and other pollutants from developed areas. The basic design of these structures is to capture and isolate up to the first inch of stormwater runoff and release the flow through a porous sand media over a prolonged period of about 24 hours. A design to capture the first 1" of stormwater flow will remove approximately 55%-70% of the total suspended solids, 25%-33% of the total phosphorus and about 23%-30% of the oil and grease content of the stormwater.

Sand filtration and sedimentation basins are often designed together with the sedimentation basin discharging into the sand filtration basin for optimum efficiency.

A formal maintenance plan and schedule is required to be submitted to the TWC for review, possible modification, and approval prior to completion of construction. The plan must include a responsible party and the anticipated cleaning and monitoring schedule. The plan must also include a disposal site for any material removed from the basins and an anticipated removal schedule.

Plans should be developed to ensure that a copy of the WPAP and approval letter is maintained at the construction site. The requirements contained within the design plans and the approval letter from the TWC should be reviewed with the contractor.

Other factors to consider during the development design phase include land grading, stormwater outfall designs, grade stabilizing structures, ponds and retarding structures, streambank protection, and storm sewer design.

IX. Other Options

Street sweeping and parking lot sweeping activities can be used to reduce levels of pollutant run-off from developed areas. The effectiveness of street sweeping and vacuuming is difficult to measure and is a function of how frequently it is performed, and the percent of site that can be cleaned. Consideration must be given to the pollutant load removed by non-sweeping processes such as wind and biochemical processes. The following efficiencies for sweeping activities are appropriate:

Parameter	Broom Sweeper Efficiency (%)	Vacuum Sweeper Efficiency (%)
Total Solids	55	93
Total Phosphorus	40	74
Total Nitrogen	42	77
Chemical Oxygen Demand	31	63
Biochemical Oxygen Demand	43	77
Lead	35	76
Zinc	47	85

An important consideration in reviewing street sweeping options is that 78% of the solids load is contained within an area six inches from the curb, and that 97% of the solids load is within 40 inches from the curb. Some street sweeping devices only redistribute the solids and actually remove only about half of the solids. A significant percentage of the solid particles are also of sizes which are not effectively removed by conventional street sweepers.

X. Hazardous Spill Consideration

Spills of hazardous materials from roadways and other areas where hazardous materials are stored, transported, or used create potentially major impacts to surface and groundwater quality. Hazardous materials traps (HMT's) can be at stormwater outfalls and other locations that will protect our states' water resources from spill contamination.

HMT design typically incorporates concrete basins with a minimum storage volume of 10,000 gallons. The design allows for the first 10,000 gallons of runoff to be directed to a concrete basin where it can be isolated. The average truck carries less than 10,000 gallons of fuel, petroleum product or waste, sewage, or other hazardous material. Any flow greater than the capacity of the basin would initiate a siphon flow which would drain the basin. In theory, any hazardous material spill will be of a volume less than 10,000 gallons and would therefore be isolated in the concrete basin. A series of valves could then be opened or closed by emergency crews which would divert future flows around the basin until the hazardous material can be removed.

Most designs for these basins are ineffective during rainfall events since it is not practical to contain all rainfall or try to differentiate between contaminated and uncontaminated stormwater flows.

A formal maintenance plan and schedule is required to be submitted to the TWC for review, possible modification, and approval prior to completion of construction. The plan must include a responsible party for monitoring the siphons and the hazardous materials traps. The plan must also provide for notification and training of local emergency response personnel.

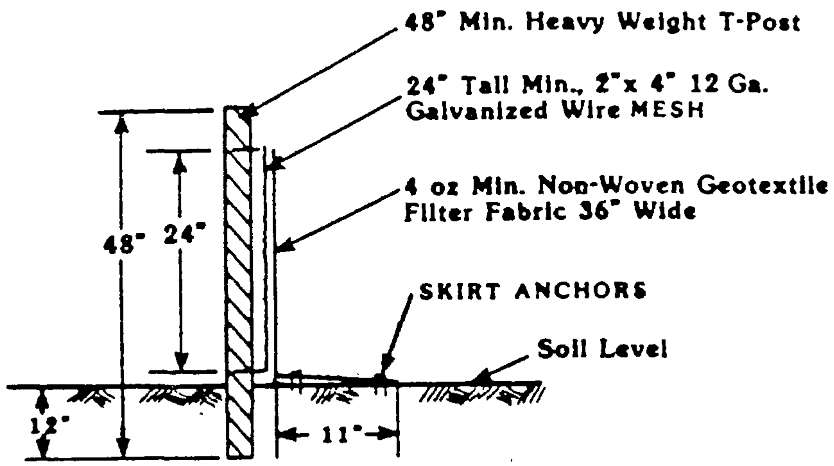
XI. Summary

The TWC reviews water pollution abatement plans and sewage collection systems prior to initiation of construction to ensure adequate protection of environmental features. As a part of this process, the TWC requires temporary erosion and sedimentation control structures be designed, approved, and installed prior to initiation of construction over the Edwards aquifer. The TWC also requires adequate maintenance and routine inspection of these structures.

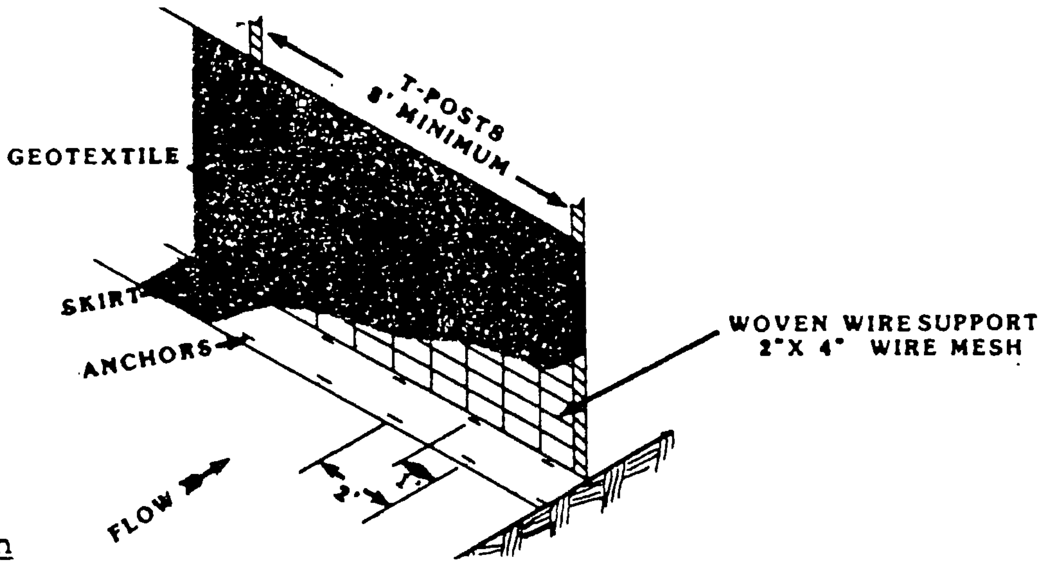
Erosion and sedimentation from construction areas is a major factor leading toward contamination of karst aquifers. By the proper design, installation, and maintenance of temporary erosion and sedimentation control structures the impacts of construction can be minimized.

Permanent sedimentation and filtration basins, and hazardous materials traps can be used to mitigate the impacts from developed areas. The amount of pollutants from developed areas can be estimated and using appropriate removal rates for sedimentation and filtration structures the anticipated additional loading to receiving waters can be estimated during the design phase of developments.

SILT FENCE



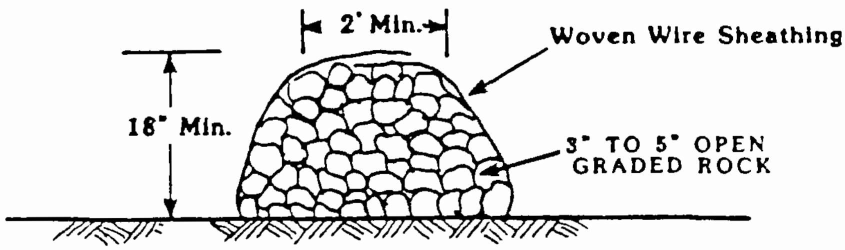
Cross-Section



Installation

- Layout the silt fence following as closely as possible to the contour.
- Clear the ground of debris, rocks, plants (including grasses taller than 2") so that a smooth surface can be utilized for anchoring the skirt.
- Drive the heavy duty T posts at least 12 inches in to the ground and at a slight angle towards the flow.
- Attach the 2" x 4" 12 Gauge welded wire mesh to the T posts with 11 1/2 gauge galvanized T post clips. The top of the wire should be 24" above ground level. The welded wire mesh shall be overlapped 6" and tied at least 6 times with hog rings.
- The silt fence will be installed with a skirt a minimum of 11" wide placed on the uphill side of the fence in a direction towards the anticipated runoff. The fabric should overlap the top of the wire by 1".
- Anchor the silt fence on alternating 2' centers front and back so that there is only 1' between anchor points.
- Geotextile splices should be a minimum of 18" wide attached in at least 6 places. Splices in concentrated flow areas will not be accepted.

ROCK BERM



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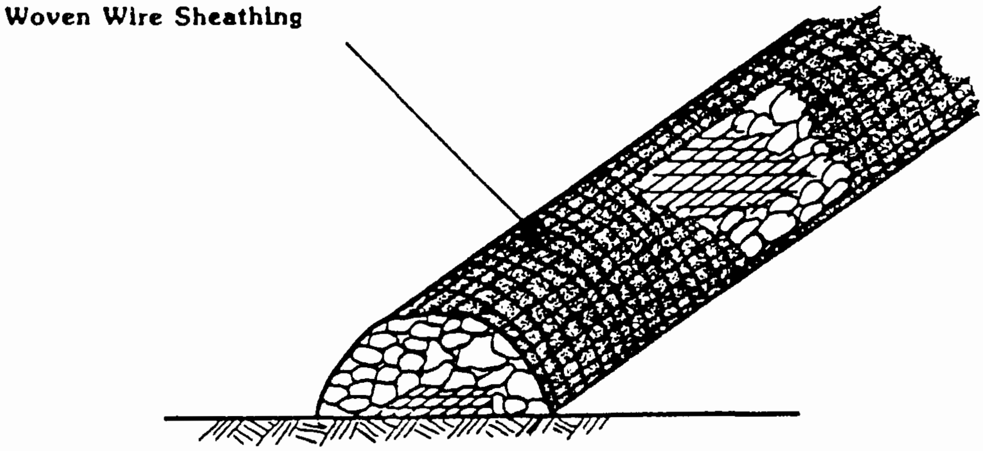
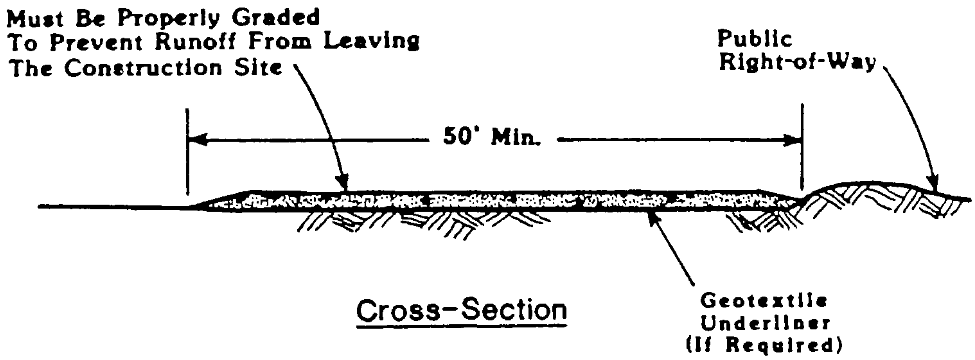


Installation

- Layout the rock berm following as closely as possible to the contour.
- Clear the area of debris, rocks or plants that will interfere with installation.
- Place woven wire fabric on the ground along the proposed installation with enough overlap to completely encircle the finished size of the berm.
- Place the rock along the center of the wire to the designated height.
- Wrap the structure with the previously placed wire mesh secure enough so that when walked across the structure retains it's shape.
- Secure with tie wire.

STABILIZED CONSTRUCTION ENTRANCE



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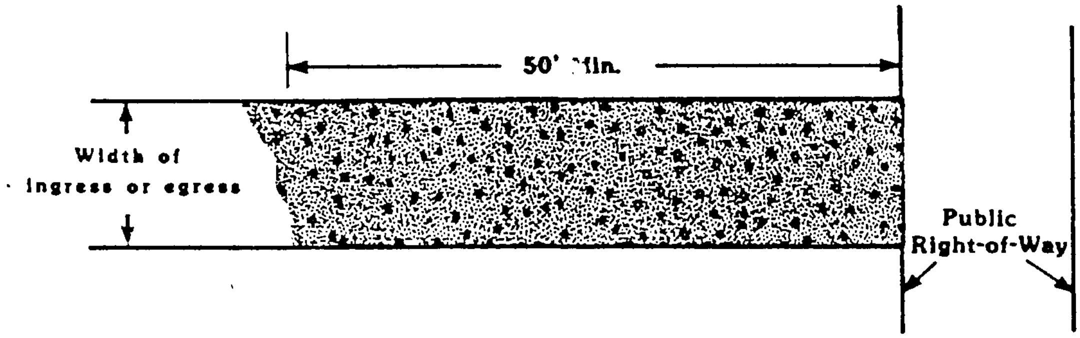

Plan View

Installation

- Clear the area of debris, rocks or plants that will interfere with installation.
- Grade the area for the entrance to flow back on to the construction site. Runoff from the S.C.E. onto a public street will not be accepted.
- Place geotextile fabric if required.
- Place rock as required.

XII. References

Regional Resource Division; Northern Division Planning District Commission, 1979, Guidebook for Screening Urban Non-point Pollution Management Strategies

Hydroscience Inc.; 1977, Water Quality Management Planning Methodology for Urban and Industrial Stormwater Needs

Lower Colorado River Authority (LCRA); Environmental Quality Division, January 1991, LCRA Lake Travis Non-point Source Pollution Control Ordinance

City of Austin; 1988, Erosion and Sedimentation Control Manual

City of Austin; Drainage Criteria Manual

City of Austin; Design Guidelines for Water Quality Control Basins

Erosion and Sedimentation Control Methodologies for Construction Activities over the Edwards Aquifer in Texas

Hank B. Smith, P.E.

Texas Water Commission
Austin, Texas

Question - On environmental geology field trips in the Austin area, I have noticed severe erosion problems in the stream channels as a result of ephemeral storm runoff caused by urbanization. What is the Texas Water Board doing about this problem?

Answer -

The Texas Water Commission (TWC) has rules regarding development over the Edwards Aquifer Recharge Zone which require temporary erosion and sedimentation controls. These control structures filter sediments and include hay bales, filter fences, rock berms, and brush berms.

The major part of the erosion potential exists between the time when the native vegetation is removed and when the site construction is complete and the vegetation is stabilized. Such impact can be minimized with the proper design, installation, and maintenance of temporary erosion and sedimentation control structures. Additionally, negative impact can be reduced by limiting the amount of disturbed areas by identifying a construction area and limiting access to this area.

Contractors are required to routinely clean paved surfaces with appropriate power brooms, or vacuum devices to minimize the amount of sediment within the construction area.

The Edwards Aquifer Program requires permanent sedimentation control structures which improve the quality of stormwater runoff and provide protection of water resources. Sedimentation basins can be effective at removing suspended solids from developed areas, and sand filtration basins have been used in the Austin and Central Texas area to effectively remove suspended solids, bacteria, and other pollutants from developed areas.

Case Histories of Several Approaches to Stormwater Management in Urbanized Karst Terrain, Southwest Missouri

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Abstract

Because sinkholes in karst terrain are dynamic components of surface-ground water interactions, they present particular challenges in the design of urban stormwater systems. Sinkholes may function as either drains or ponds for surface runoff, and their hydrologic behavior may change markedly with long term urban growth, as impacts of earlier development and off-site influences are added to natural drainage patterns at proposed construction sites. Comparison of three sites in Springfield, Missouri, in the Springfield Plateau Karst, illustrates the effects of contrasting approaches to stormwater design.

A wooded tract with unmodified sinkholes was cleared and developed for residential use. Discharge of stormwater was directed into sinkholes, and erosion control consisted of hydro-mulching and sedimentation fences in sinkhole areas. East of this location are two parcels which differ in removal of vegetation and off-site drainage relationships. Stormwater design at these sites was adapted for modifications made to sinkholes during railroad and highway construction several decades earlier. Sediment fencing, hydro-mulching, and detention berms augment infiltration, restrict erosion, retard discharge to sinkholes, and incorporate off-site considerations.

Ongoing observations of stormwater behavior indicate problems of flooding and sediment control at the western site but minimal disruption of existing drainage patterns at the eastern sites. Difficulties in ascertaining all potential sources of groundwater contamination underscore the need for stormwater regulations which regard sinkholes as integral components of

entire karst drainage systems.

Introduction

Springfield, the county seat of Greene County, is located in the Springfield Karst Plain in southwest Missouri (Figure 1). Dominant bedrock underlying this plain consists of horizontal strata of the Mississippian Burlington-Keokuk Limestone, a light gray, coarsely crystalline, fossiliferous unit.

Because the Burlington-Keokuk Limestone is susceptible to solution, karst features are extensive throughout the county. Figure 2 illustrates areas of extensive sinkhole development. Two major concentrations of sinkholes are found within the city limits of Springfield. These areas are located in northwest Springfield near the airport and in a part of east Springfield referred to as the East Cherry Street Sinkhole Area.

These two sinkhole regions are situated on topographic highs, near surface water drainage divides. Apparent controls on sinkhole occurrence in these areas include facies differences within the Burlington-Keokuk Limestone and intersections of large scale lineaments. Stormwater drainage of unaltered sinkholes ranges from rapid drainage to long-term ponding.

Past Stormwater Management Practices

Springfield and its Metropolitan Statistical Area comprise the fastest growing urban region in the state of Missouri. Rapid population growth has created additional stresses to existing land use, including reduction of open areas and increased stormwater runoff. Control of stormwater in sinkhole areas has utilized several standard engineering approaches: concrete lined channels draining into sinkholes; installation of drain pipes into the sinkhole "eye"; filling of sinkholes; elaborate drains or pumps removing stormwater from one sinkhole and discharging into another drainage basin or sinkhole; enlargement of sinkhole eyes by excavation to increase drainage capacity.

Several problems may arise in using these standard practices:

1. Runoff may exceed drainage capacity of sinkholes.
2. Sediment and debris may plug sinkholes and associated karst conduits.
3. Sinkholes are not isolated entities, but are integral parts of dynamic drainage systems. Stormwater directed into one sinkhole may resurface in another part of the drainage.
4. Direct disposal of stormwater into a karst aquifer may degrade water quality by introducing contaminants. Existing practices may violate 1988 federal regulations governing urban stormwater quality.

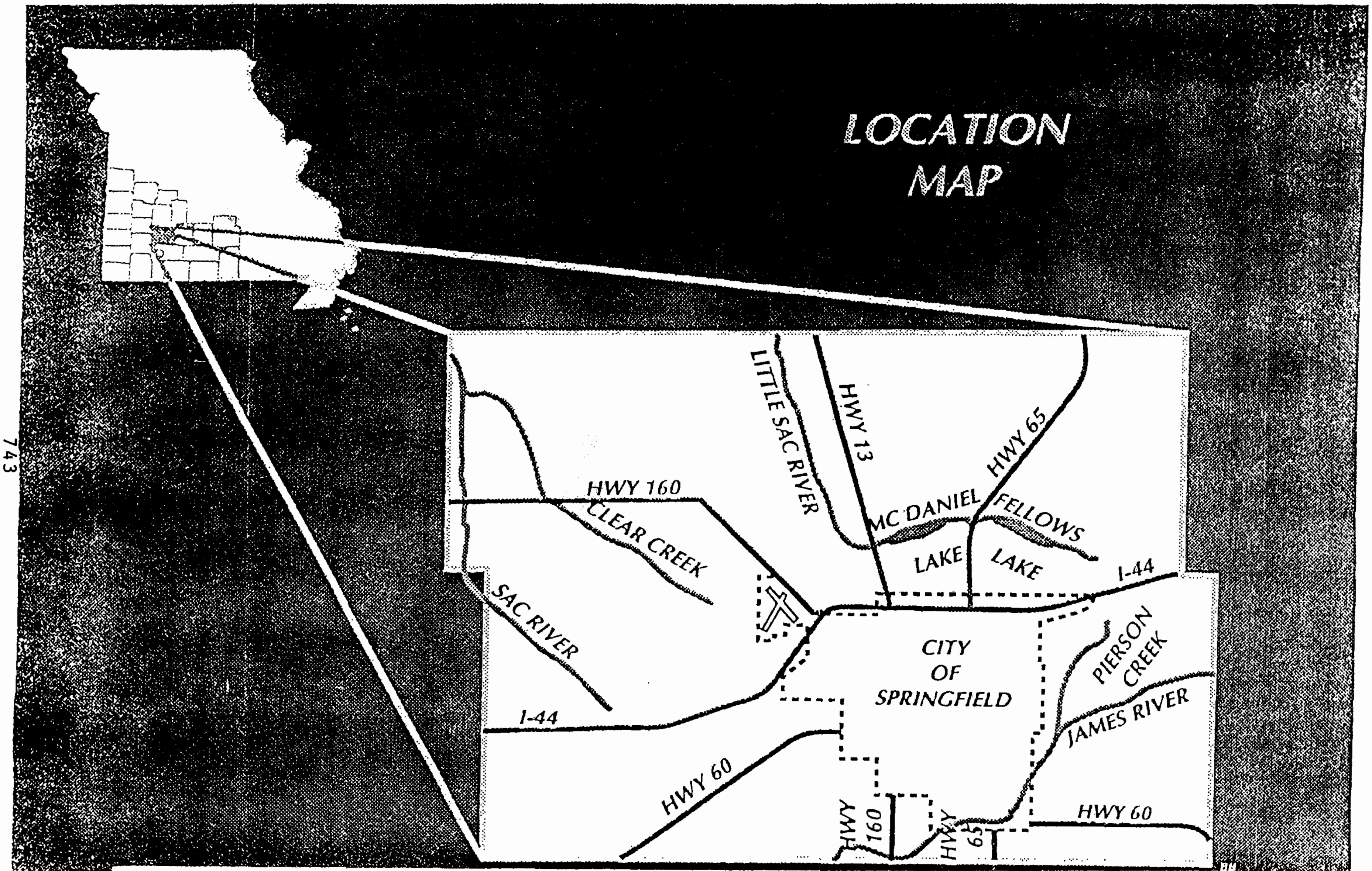

Figure 1. Location map of Springfield and Greene County, Missouri.

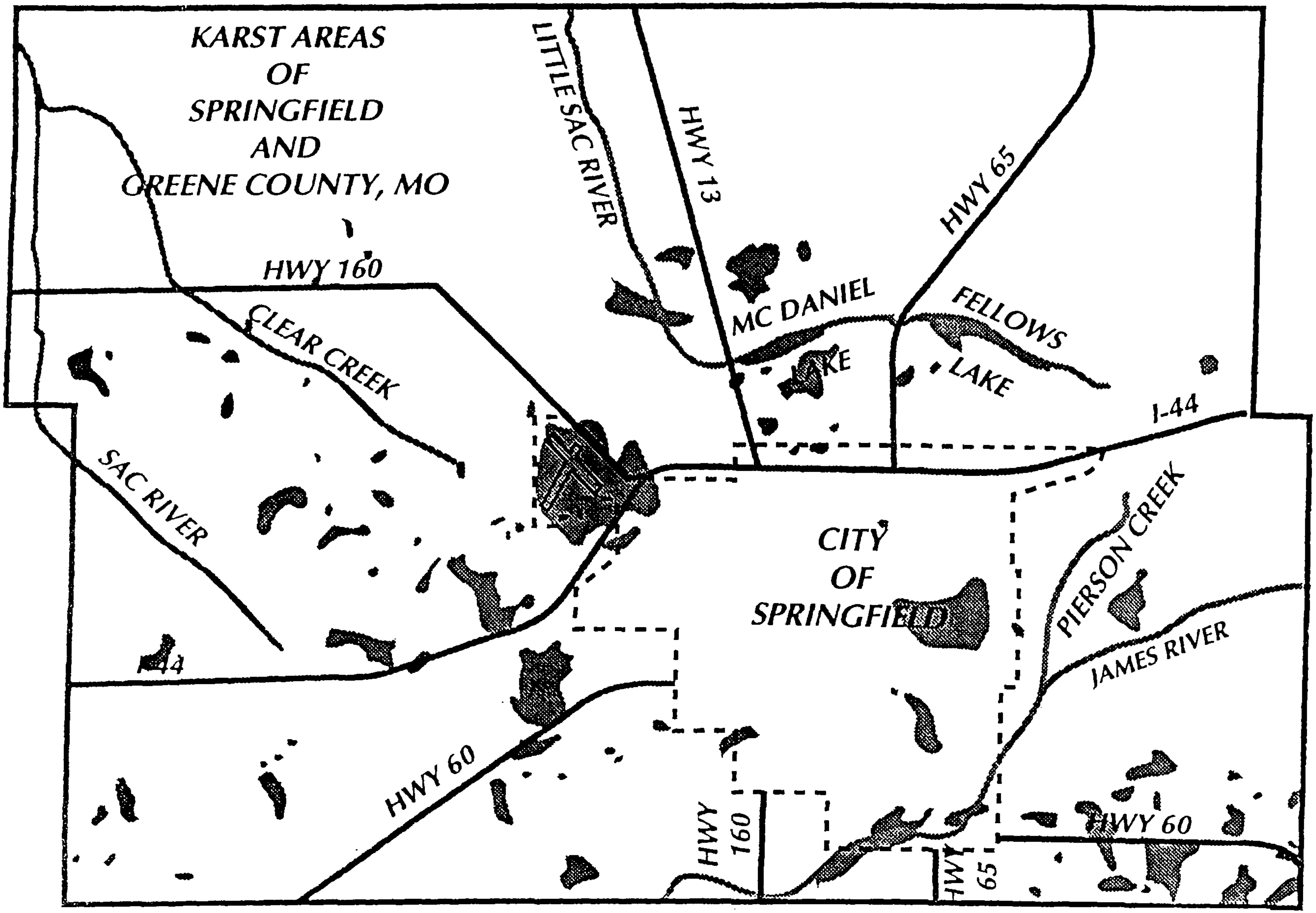

Figure 2. Major karst areas of Springfield and Greene County, Missouri. Two areas of extensive sinkhole development within the city are located at the Springfield Airport and near East Cherry Street. Scale is approximately 1"=3.5 mi.

These problems have been recognized in previous studies and master plans for stormwater control in the Springfield area. In the 1950's a comprehensive stormwater master plan addressed existing practices and recommended discontinuing the use of sinkholes as stormwater drains. Because of the total costs necessary to accomplish its goals, the master plan was never adopted (Turner, 1989).

In the 1970's William Hayes, Environmental Geologist for the City of Springfield, prepared a report sponsored jointly by the city and the Federal Department of Housing and Urban Development (Hayes, 1977). This report discussed the origin of local sinkholes, their drainage areas and hydrologic characteristics, and geologic factors controlling their alignment. Although the study addressed the adverse effects of urban development on sinkholes and karst groundwater systems, it emphasized many standard structural controls for containment and disposal of stormwater.

Having recognized the inadequacy of existing designs to control flooding and the need to accommodate increased runoff from future development, the City of Springfield adopted an ordinance (effective June 19, 1989) to protect sinkholes and prevent flooding. This ordinance, based on similar regulations enacted by Lexington, Kentucky, requires a permit for any proposed construction within a sinkhole drainage area. Application for the permit must include a hydrogeologic report evaluating the sensitivity of soil to erosion; relationship of the sinkhole to the overall drainage area; existing sinkhole drainage characteristics; capability of the sinkhole to accept stormwater; type of work and equipment to be used, impacts from proposed activities, and requirements to protect the sinkhole during construction.

Although the ordinance attempts a balance between urban growth and protection of natural drainage of sinkholes, several serious ambiguities exist. Description of enforcement procedures is indistinct. The ordinance does not address the piecemeal nature of development, in which earlier construction on adjacent sites may affect drainage at the site being considered. Lack of recognition of sinkholes as integral parts of dynamic hydrologic systems may result in problems with on-site/off-site drainage.

Case Histories of Recent Urban Developments

Three sites were analyzed to examine the effectiveness of contrasting design approaches to stormwater management. These sites differ in vegetation, on-site/off-site considerations, and types of development proposed. All three sites are located within the East Cherry Street Sinkhole Area previously described. Figure 3 illustrates the locations of these sites relative to important hydrologic and cultural features. Dye tracing (Hayes and Vineyard, 1969; Hayes, 1977; Aley and Thomson, 1981; and Thomson, 1986) has indicated that the East Cherry Sinkhole

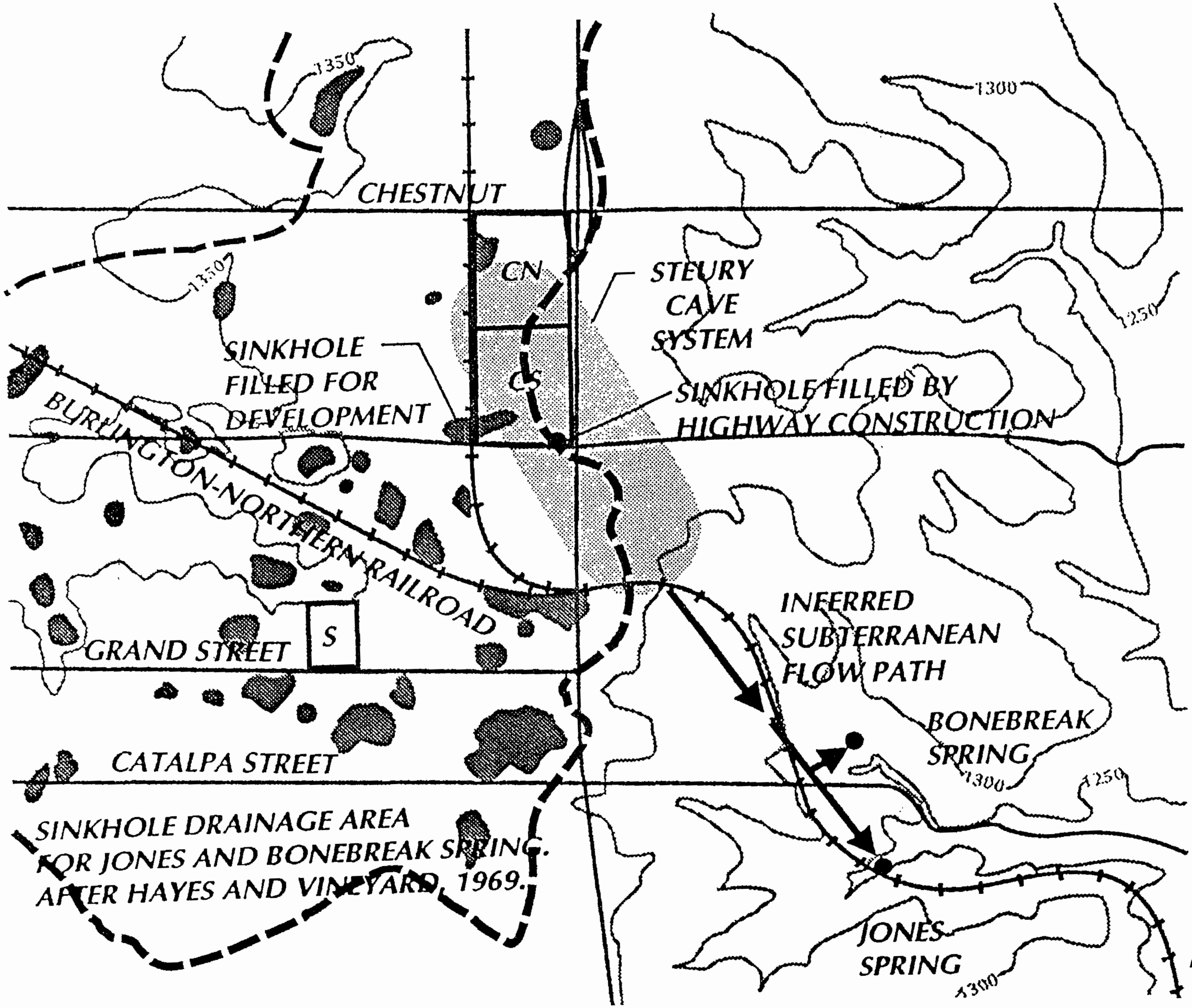

Figure 3. Map of East Cherry Street Sinkhole Area (dashed line), showing locations of study sites relative to hydrologic and cultural features. S = Sundance Estates; CN = Chestnut Plaza Phase I; and CS = Chestnut Plaza Phase II. Scale is approximately 1"= 3 mi.

drainage provides recharge to Jones and Bonebreak Springs. These springs in turn discharge to Pierson Creek, a major tributary which joins the James River approximately 3 miles upstream from the intake of the Blackman Treatment Plant municipal water supply. Sinkholes on two of the sites, Chestnut Plaza North and South, drain by way of the Steury Cave system to Jones and Bonebreak Springs.

Urban development in the East Cherry Street Area began in the early 1900's. Residential development, usually on large parcels, continued into the 1950's and 1960's, when industrial and commercial uses were targeted for this area. By the late 1980's few open areas remained. These undeveloped parcels are attractive because of their proximity to downtown Springfield and major regional transportation routes. During the past several decades, this area has experienced sinkhole flooding, exacerbated by increased runoff accompanying development.

Sundance Estates Subdivision

Sundance Estates Subdivision is located on East Grand Street, southwest of the other two study areas (Figure 3). This ten-acre was a densely wooded tract with one farm house and several outbuildings. Existing structures were all located near Grand Street.

Sinkholes on this tract are not apparent on either the U.S.G.S. (10-foot contour interval) or the City of Springfield (5-foot contour interval) topographic maps. Preliminary plans for the subdivision were approved in May 1989 based on the possible existence of one sinkhole near Grand Street. Evidence of other sinkholes on the site was not encountered until design topographic surveys were conducted.

As a result of this new evidence, the city specified several additional design considerations for the site. Buildings were prohibited within the floodplains of individual sinkholes. Construction of basements was prohibited, additional reinforcement was required for the foundations, and wider easements between houses were required for possible future enhancement of flood control. The city, the developer, and the design consultant agreed to several other precautionary measures: installation of silt fences around sinkholes; planting of vegetation to control erosion; prohibited use of heavy equipment in the sinkholes; clearing of vegetation by backhoes situated outside the sinkhole rim and extending only the bucket into the sinkholes. Figure 4 illustrates the site layout for the subdivision and the location of the sinkholes.

Prior to development, observations made during moderate to heavy storm events indicated that the sinkholes were free draining. During a 4 to 5 inch rainfall in approximately 10 to 12 hours (May 1989), ponding was not observed in the large sinkhole (S-1) and only very moist soil conditions were evident in several

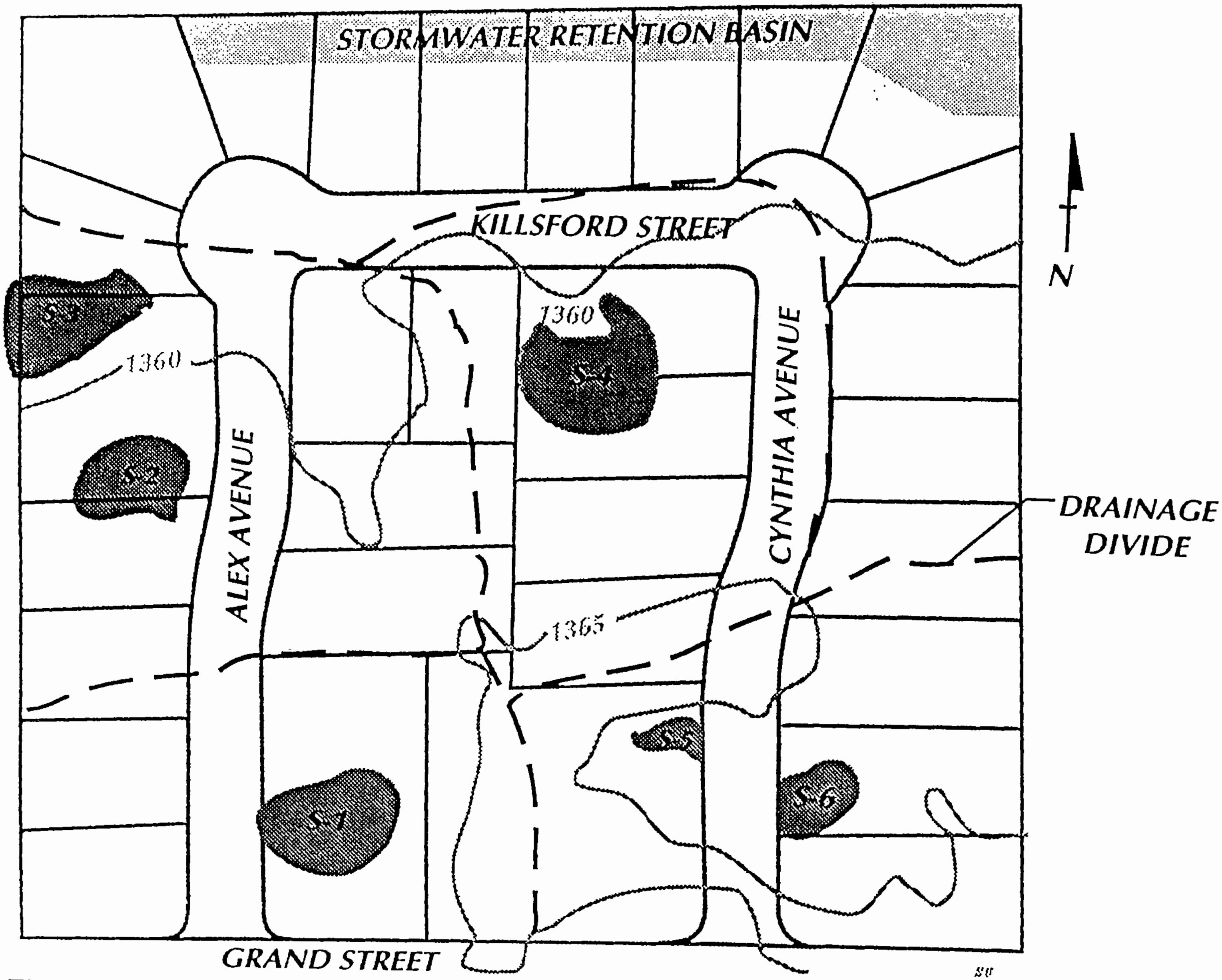

Figure 4. Sundance Estates subdivision, showing locations of sinkholes S-1 through S-6. Scale is approximately 1"=100'.

other sinkholes (S-4 and S-3). Stormwater design used the Rational Method to determine peak flows and detention volumes necessary for each sinkhole. Total drainage to each sinkhole is small, and for the proposed land-use, calculated volumes accommodate this additional runoff. Table 1 lists a summary of sinkhole drainage characteristics. A retention basin was designed at the north end of the property in order to prevent potential off-site flooding. Recommended stormwater design measures were preventive and not remedial in nature.

Problems at the site began after clearing the sinkholes of debris. Cleared sinkholes were hydro-mulched but were not watered to facilitate germination of the seed. Work was performed during the late fall, and temperatures dropped below freezing before germination. Sediment and seed were washed into the centers of the sinkholes, and sinkholes remained void of vegetation for several months. Contrary to recommendations, the contractor used heavy equipment in sinkhole areas for grading, thus compacting the soil structure. Lack of silt fence maintenance allowed sediment to wash into the sinkholes. Heavy equipment used in the large retention area compacted soils and reduced infiltration capacity. One house, constructed between sinkholes S-2 and S-3, encroaches on sinkhole floodplains, and a second house is located within the floodplain of sinkhole S-1. Instead of enforcing design agreements, the city placed responsibility on the design consultant to oversee all phases of construction.

Sundance Estates Subdivision is almost completely developed, and all sinkholes have adequate vegetative cover. During 1989 and 1990, flooding occurred repeatedly at the site. Moderate rainfall (1 to 2-inch rain in 6 hours) caused ponding of water in the sinkholes, with water remaining up to 3 or 4 days in all sinkholes. Because of the flooding and possible structural hazards, HUD has refused loans for this subdivision on lots with sinkholes or adjacent to sinkholes. Further evaluation of the site during 1991 has been limited because of sparse rainfall in the Springfield area.

Chestnut Plaza Phase I (North)

Chestnut Plaza Phase I, the first site affected by the new sinkhole ordinance, is an approximately 30 acre tract located at the corner of East Chestnut Expressway and U.S. Highway 65 (Figure 3). Chestnut Plaza Phase I is planned for commercial development, which is anticipated to include fast food restaurants, a hotel, retail stores, and light industry or manufacturing.

During the 1930's this site was used as pasture land. Review of 1936 aerial photos reveals that sinkholes at the site were clear of any trees and that the railroad and the highway had not been constructed. Shortly after this time, construction of the Burlington Northern railroad bisected the large sinkhole on

Sinkhole Drainage Characteristics

SINKHOLE	SINKHOLE AREA	SINKHOLE DRAINAGE VOLUME	SINKHOLE DRAINAGE AREA	CALCULATED RUNOFF VOLUME
S-1	0.12 ACRES	5,800 CU. FT.	1.60 ACRES	2,400 CU. FT.
S-2	0.09 ACRES	7,240 CU. FT.	1.87 ACRES	2,800 CU. FT.
S-3	0.11 ACRES			
S-4	0.03 ACRES	2,600 CU. FT.	1.53 ACRES	2,300 CU. FT.
S-5	0.01 ACRES	6,964 CU. FT.	1.57 ACRES	2,400 CU. FT.
S-6	0.05 ACRES			

Table 1

Sinkhole Drainage Characteristics

SINKHOLE	SINKHOLE AREA	SINKHOLE DRAINAGE VOLUME	SINKHOLE DRAINAGE AREA	CALCULATED RUNOFF VOLUME
CN-1	3.6 ACRES	535,000 CU. FT.	27.5 ACRES	100,000 CU. FT.
CN-2	0.81 ACRES	60,000 CU. FT.	3.9 ACRES	15,000 CU. FT.
CN-3	216 FT^2	N/A	N/A	N/A
CN-4	0.15 ACRES	N/A	N/A	N/A
CS-1A	5 ACRES	800,000 CU. FT.	50 ACRES	741,000 CU. FT.
CS-1B	10 ACRES	1,600,000 CU. FT.	50 ACRES	1,263,000 CU. FT.
CS-2	0.68 ACRES	50,000 CU. FT.	10 ACRES	130,000 CU. FT.

Table 2

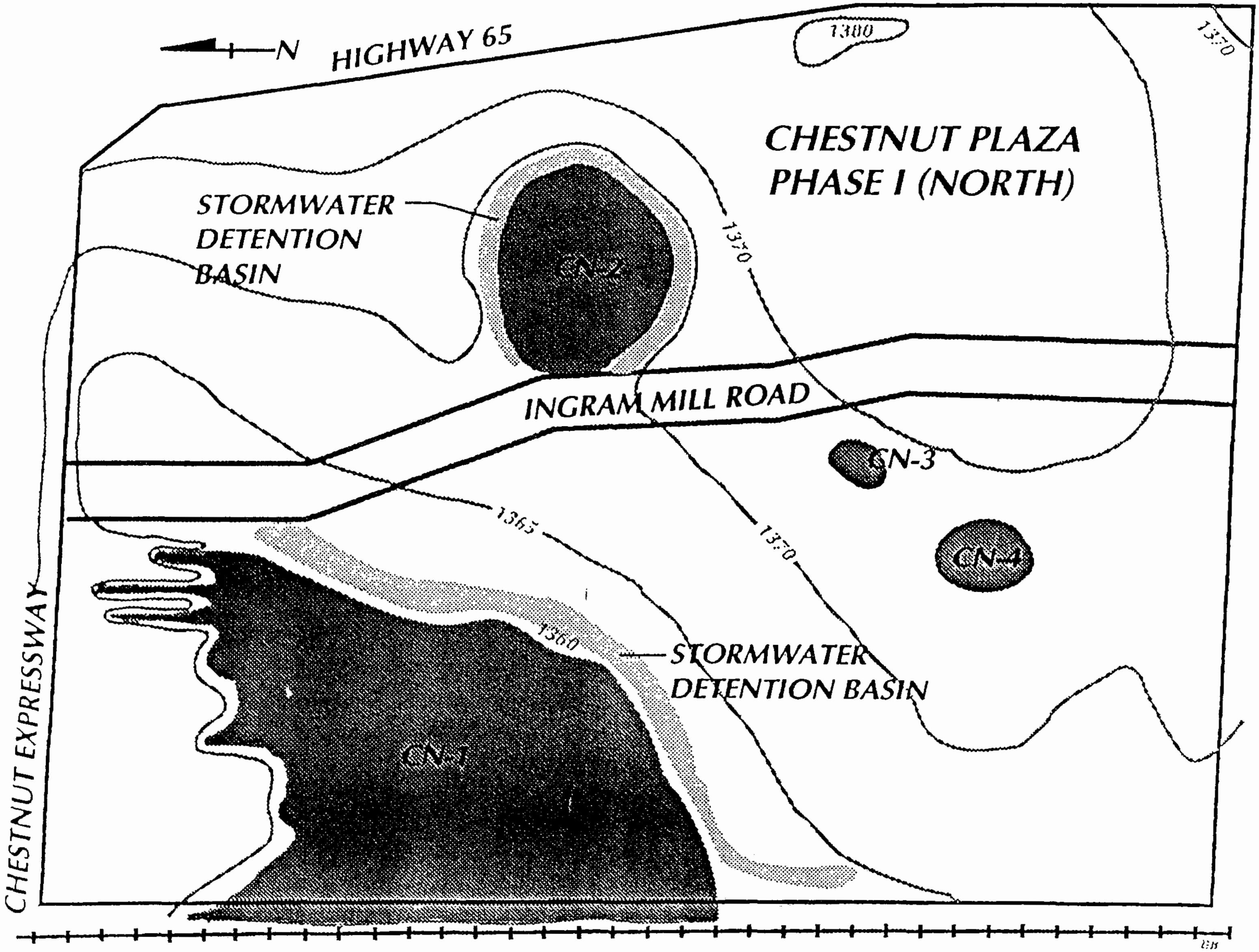

Figure 5. Chestnut Plaza Phase I (north), showing location of sinkholes CN-1 through CN-4.
Scale is approximately 1"=160'

the site. Later construction of East Chestnut Street covered another part of the sinkhole.

During construction of the railroad and Chestnut Street, excavated soil and other material were dumped into the sinkhole. When Chestnut Expressway was widened, excavated soil, pavement, and other materials were dumped into the sinkhole.

Alteration of the large sinkhole severely restricted its drainage capacity. The sinkhole floods and ponds water for several days after moderate rainfalls. Observations made during a 4 to 5-inch rainfall in January, 1990, revealed that stormwater runoff rose to the 1354 elevation within the sinkhole. This level is below the established floodplain elevation of 1359, and flood waters drained after 5 to 6 days. This large sinkhole also receives off-site drainage from a 30 to 40 acre site to the north.

Because sinkholes at this site had already been altered, it was imperative that development would not further degrade drainage of the sinkholes. Although it contained a great amount of fill material, the larger sinkhole still had the capacity and drainage to accommodate increased runoff if some protective measures were taken into consideration.

Design of the stormwater detention areas utilized the Rational Method for calculating stormwater runoff and peak flows (Table 2). FEMA had already established floodplain limits in the larger sinkhole. To protect the sinkhole and allow for future development, stormwater detention berms were designed around the two larger sinkholes (CN-1 and CN-2 in Figure 5). No building is allowed in the sinkhole areas. To prevent soil compaction by heavy equipment, clearing is permitted by hand only.

Additional stormwater detention will be required for individual sites along the northwest side of sinkhole CN-1 as development occurs along Chestnut Expressway. Minor filling has been approved for the area adjacent to Chestnut Expressway. Although two smaller sinkholes, CN-3 and CN-4, are not important in the overall drainage of the site, extreme thickness of their soils and lack of solid bedrock beneath them could lead to potential collapse. For these reasons CN-3 and CN-4 have been designated "non-buildable" and will remain in their natural state.

Chestnut Plaza Phase I was approved for construction during the summer of 1990. Prior to any construction or clearing at the site, sediment control methods were established. These methods included silt fences around sinkhole areas and also outside the perimeters of their established floodplains. Completion of detention berms around sinkholes CN-1 and CN-2 and establishment of vegetation in these berms was required before construction could begin.

Recommended protective measures have been established, and the street and one warehouse have been constructed. Additional construction is expected within the next several months. Observations made during a 2 to 3 inch, 12 hour storm indicate confinement of drainage to the sinkhole and berms and infiltration of stormwater within 3 to 4 days. Additional monitoring of runoff and sampling of stormwater are planned for the site.

Chestnut Plaza Phase II

Chestnut Plaza Phase II is located at the intersection of East Cherry Street and U.S. Highway 65 and south of Chestnut Plaza Phase I (Figure 6). This site encompasses approximately 30 acres and surface runoff drains southwest to an off-site sinkhole (CS-1B). Sinkholes CS-1A and CS-1B are two halves of an originally continuous sinkhole which was bisected by construction of the Burlington Northern Railroad. Other modifications include partial filling of sinkhole CS-1B for industrial development during the 1970's.

Figure 6 illustrates the complexities of the drainage system. Drainage area shown in this figure is approximately 100 acres. All runoff is directed toward an outlet constructed in sinkhole CS-1B. When this modification was made in 1972, the city's environmental geologist stated that the Chestnut Plaza Phase II site was "undevelopable" (Giles, 1990, personal communication).

Initially sinkhole SC-1B was able to accept runoff from 1970's and early 1980's construction. However, because design modification included only a metal grate to stop debris, the underlying cavern system has filled with sediment and is greatly reduced in drainage capacity (Thomson, 1990, personal communication). Plans for Chestnut Plaza Phase II had to address this problem.

Figure 7 illustrates the design for Chestnut Plaza Phase II, including the sinkhole area and proposed stormwater controls. Because the developer desired the largest possible area for building, stormwater detention basins such as those at Chestnut Plaza Phase I were not used. Instead, a two tier stormwater network was designed to provide maximum buildable area while controlling stormwater runoff.

Because FEMA had not established a floodplain for the site, floodplain information was calculated using the Soil Conservation Service TR-55 Stormwater Method. Pre-existing major alterations of sinkhole drainage and sensitivity of the adjacent site to flooding suggested the use of the TR-55 method, which allows more detailed analysis of stormwater behavior. This method utilizes soil characteristics, land use, hydrologic characteristics, and rainfall distribution and duration to develop runoff hydrographs. Calculated floodplain elevation is 1353.5 feet and volume is

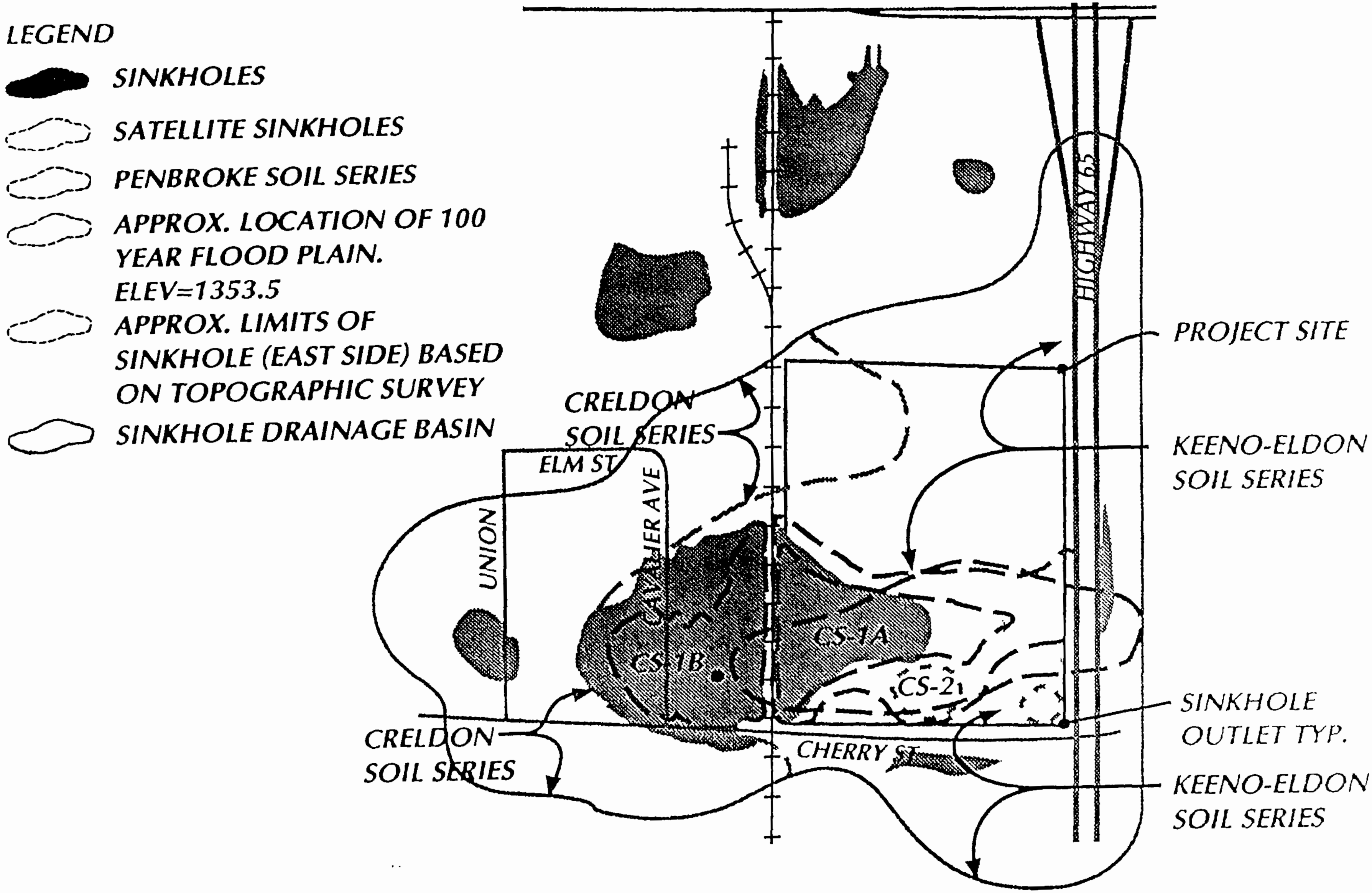

Figure 6. Map of Chestnut Plaza area, showing drainage basin for large bisected sinkhole (CS-1A and CS-1B). Scale is approximately 1"=500'.

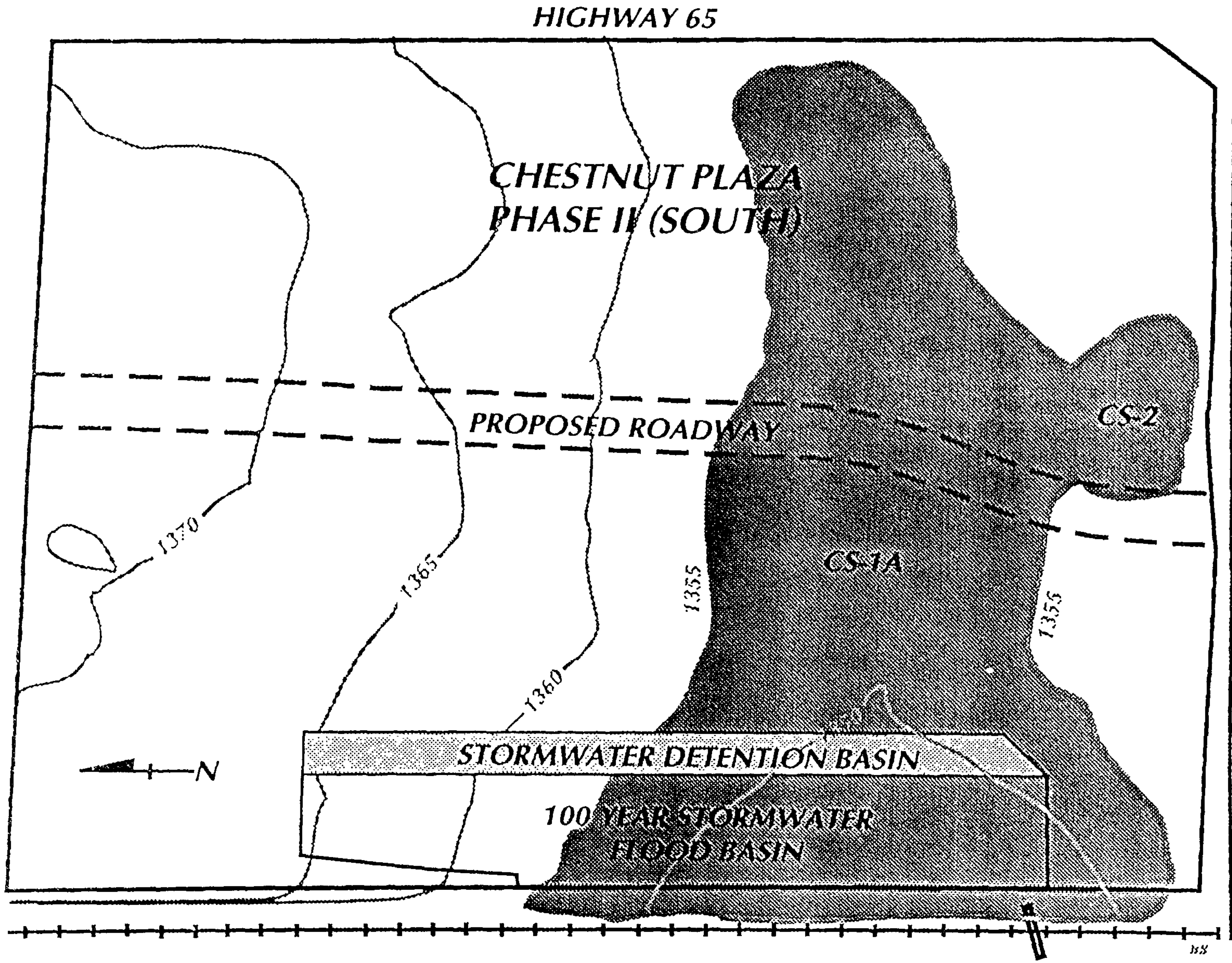

Figure 7. Chestnut Plaza Phase II (south), showing sinkholes CS-1A and CS-1B. Scale is approximately 1"=170'.

estimated at 1,000,000 cubic feet for the 100 year flood storage basin (Table 2).

In addition to the 100 year flood volume, drainage volume for projected complet development was computed using the Rational Method, and a detention basin was designed for this volume. As in Chestnut Plaza Phase I, both basins will be grass lined and sediment control measures will be used during and after construction.

Outflow to the off-site drain was determined from field measurement, and values obtained were further reduced to accommodate runoff from future development. Drainage design for the site west of Chestnut Plaza Phase II allowed a discharge of 93 cfs from sinkhole CS-1B during a 10 year storm (Hayes and Thomson, 1973). Mapping of the Steury Cave system suggests that sedimentation has reduced this discharge capacity by 90 to 95 percent (Thomson, 1990, personal communication). Field observations of sinkhole CS-1B made during a 2 inch, 8 hour rainfall support this estimate. Off-site discharge to CS-1B from Chestnut Plaza II was reduced accordingly by 95 percent to a design outflow of 5 cfs.

Preliminary design for Chestnut Plaza Phase II was approved in the summer of 1990. Vegetation has been cleared at the site, and silt fences have been placed at sinkhole CN-1B. Construction is expected to begin in 1992. Runoff and water quality will be monitored before and during construction.

Conclusions

Case histories of three sites in Springfield, Missouri illustrate results of contrasting approaches to stormwater design and regulation in urban karst terrain. Residential development planned for a 10-acre wooded tract (Sundance Estates) required minimal preventive measures to protect free-draining sinkholes from construction activities and to maintain the drainage capacities of those sinkholes. Lack of adherence to design recommendations has resulted in repeated flooding of sinkholes at this site.

Two adjoining commercial parcels are located in an area which experienced extensive alteration of drainage during development of neighboring sites. Because of existing flood hazards and inflow of stormwater from an adjacent site, design for the northern parcel (Chestnut Plaza Phase I) included land use restrictions, sediment control, and construction of detention basins which approximate natural landforms. Close communication among the city, the developer, and the design consultant ensured completion of stormwater measures as planned. Observations made during recent storms suggest that the design maintains adequate drainage. Future studies will examine the impact of the detention basins on stormwater quality.

Because of off-site releases of stormwater and a desire to maximize commercial land use, design for the southern parcel (Chestnut Plaza Phase II) includes a two-tier stormwater network as well as vegetative and sediment control measures. Construction at this site is incomplete, and continued monitoring of runoff and stormwater quality are planned.

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TITLE: **CASE HISTORIES OF SEVERAL APPROACHES TO STORMWATER MANAGEMENT IN AN URBANIZED KARST TERRAIN, SOUTHWEST MISSOURI**

AUTHORS: Wendell L. Barner - ICF Kaiser Engineers
Patricia Miller - California State University, Sacramento

Question: Is dye-tracing a requirement (or is it done) prior to zoning or development?

Response: The sinkhole ordinance adopted in 1989 does not specifically require or mention dye tracing. However, the ordinance does specify substantial "state of the art" field studies and evaluation of the sinkhole system. Therefore, dye tracing may be performed by the developer or the consultant to determine sinkhole drainage characteristics and impacts on the groundwater system from the proposed development. Other investigators have performed dye tracing in the Springfield area, and their results are on file with the Missouri Geological Survey, as well as with local governing agencies. Many traces in the Springfield area were executed prior to, during, or after modifications to sinkholes for stormwater drainage.

CONTAMINANT INVESTIGATION IN A KARST REGION

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Abstract

A series of springs in the karst region of north central Kentucky appeared to have been contaminated. These springs are within a half mile of two sinkholes, which were filled in as permitted landfills for inert waste and then developed into a medium industrial park. A preremedial site inspection was performed under the authority of the Superfund laws in late 1989. A preliminary site visit included site reconnaissance and geologic field work to locate the springs. A review of historical aerial photos aided in the planning of the investigation program, which consisted of magnetic and soil gas surveys and environmental soil and water sampling. The magnetic survey indicated the presence of buried ferrous objects. The soil gas survey points were laid out incorporating this information. Soil sample locations were selected based on the results of the soil gas survey. Seventeen surface and subsurface soil samples were taken. Eleven water samples were taken from various springs, rivers, and the local public water supply. The analytical results of the soil samples taken over the sinkholes matched 20 compounds also found in the water and sediment samples taken from the spring. The location of the springs roughly coincided with the strike of the major fracture systems reported in the literature. The success of this investigation emphasizes the importance of proper geologic consideration for contaminant monitoring in karst regions.

Introduction

This investigation was part of the phased investigative approach that is employed by the USEPA to determine whether sites warrant being placed on the National Priorities List. The purpose of this particular investigation was to determine the nature of the contaminants present at the site and to determine if a release of these substances has occurred or is likely to occur (USEPA, 1991). Additionally, the potential pathways for contaminant migration were investigated. This paper describes the methodology that was employed to meet the stated purpose.

It is important to note several limitations, both organizational and geologic, that were key in formulating the approach to this investigation. This Screening Site Inspection (SSI) was done under the authority of the USEPA, Region IV, as part of a preremedial Superfund investigation. These SSIs are limited in scope and concentrate on source characterization and identification of potential pathways for contaminant migration. Permanent groundwater monitoring wells are not part of normal SSIs. Temporary groundwater monitoring wells were not considered for two reasons. First, the site is located adjacent to a bluff that overlooks Valley Creek and Valley Creek reservoir (the closest surface water bodies). The top of the bluff is approximately

65 feet above the water level in Valley Creek reservoir. This physically precluded the installation of the temporary wells by the investigation personnel. Second, in mature karst areas, such as around Elizabethtown, Kentucky, the majority of the groundwater, from 60% to 80%, would be expected to flow in the dissolution features in the limestone (conduit flow), not as diffuse flow through the limestone itself (Quinlin and Ewers, 1985). These dissolution features cannot be readily encountered when drilling. A much more reliable approach towards monitoring potentially contaminated groundwater is to inventory and monitor springs in the areas as they are the discharge points for the groundwater (Quinlin and Ewers, 1985). This idea is the basis for the technical approach taken in this investigation.

Background

From 1969 to 1977, a landfill was permitted to receive inert waste to be disposed of in a series of sinkholes located approximately 1/2 mile southeast of Elizabethtown. The site is approximately ten acres in size (1800 feet by 1000 feet) and contains at least three sinkholes. The site is bounded approximately on the west by Interstate 65 and on the north by the valley of Valley Creek (Figure 1). This creek flows to the west through Elizabethtown.

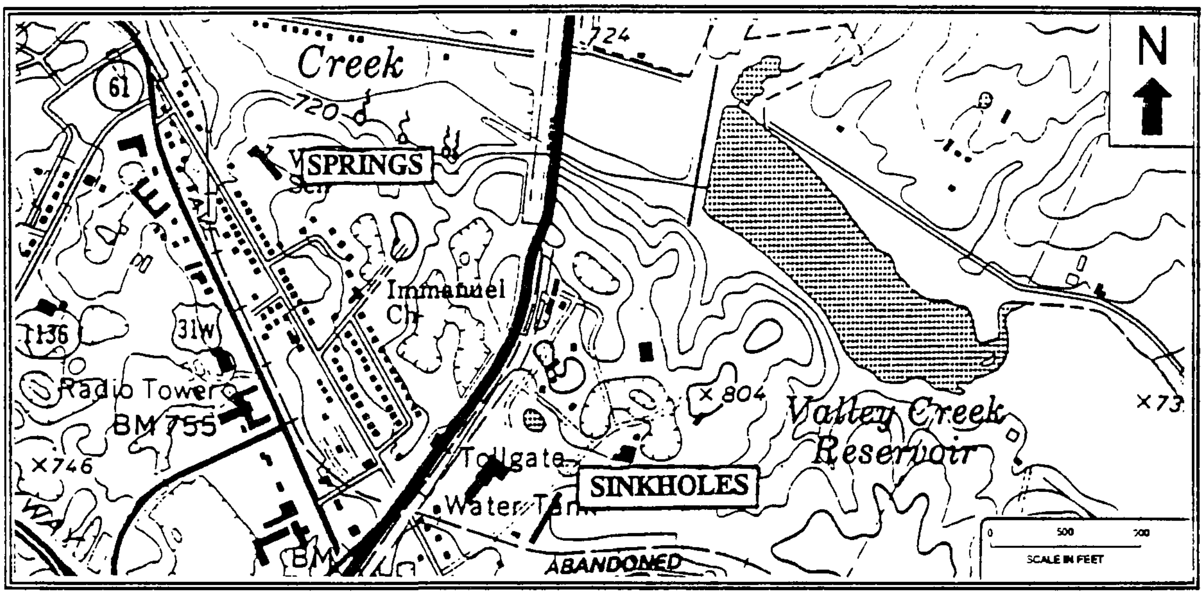

Figure 1. Study Area
Elizabethtown, Kentucky

The permitted wastes for this landfill included Neoprene, cardboard, wood, paper, foundry sand, bricks, wood palettes, and empty barrels. An inspection in 1972 by the Kentucky Department of Environmental Protection indicated that the landfill was receiving liquid wastes and domestic garbage in violation of its permit. Regarding the liquid wastes, a galvanizing company reportedly had a pipeline for the disposal of spent sulfuric acid into one of the sinkholes. USEPA files raise the question of the disposal method of 1200 drums of dross zinc ash by the galvanizing company. These files also raise the possibility that a tanker truck was buried in the largest of the sinkholes (USEPA, 1991).

Contamination of a spring on a farm immediately across I-65 from the landfill was noted in the early 1970s. Investigations by both State and Federal agencies indicated the presence of elevated values for zinc, nickel, chlorinated compounds, and other organic compounds as well as low pH values for the spring water. The landfill was officially closed in 1977. This investigation took place in 1989 (USEPA, 1991).

Geology

The Elizabethtown area is in the Interior Low Plateaus physiographic province (Fenneman, 1938). The Pennyroyal Plain is a gently rolling mature plain that slopes towards the southwest and is characterized by such karst features as sinkholes, springs, and sinking streams (Sauer, 1928). Valley Creek, the only perennial river in this area, flows along the northern boundary of the study area.

The discussion of the geology of this area will be limited to describing the surficial soils and the carbonates of Mississippian age which control the development of the karst features. The alluvium in this area is confined to Valley Creek and its feeder streams. The bedrock of the area is covered by a blanket of residuum of clay and chert fragments. This layer is of variable thickness; it fills in the dissolution and collapse features of the underlying limestone (Mull et al, 1988).

The primary limestones of the area are the Ste. Genevieve and the underlying St. Louis. These formations are separated by a relatively thin bed of limestone and silicified fossils known as the Lost River Chert. The Ste. Genevieve is typified by occasionally shaly oolitic limestone and dolomite beds varying from 0.5 to 4.0 feet in thickness (Mull et al, 1988). The St. Louis is the most significant formation with regards to the development of the dissolution features. The upper portion of the formation consists of primarily massive to thin-bedded limestone and dolomite. In the lower portion of the formation, there is an increase in the amount of gypsum and anhydrite. Noger and Kepferle (1985) believe that the presence of these soluble minerals is what lead to the extensive development of the mature karst features seen in the study area.

Locally, the confirmation of the rock units was not possible due to the limited outcrop in the area and the lack of subsurface or bedrock drilling. More important is the hydrogeology of the site as typified by the filled-in sinkholes, the adjacent springs and the elevation difference between the land around the sinkhole and the water table, as represented by the springs and Valley Creek. As seen in Figure 1, the landfill was located in the largest of the three sinkholes east of Interstate-65 and south of Valley Creek. This plateau is bordered on the north by the alluvial valley of Valley Creek and rises approximately 65 feet above the valley floor. Across the Interstate to the west are five springs that originate from the base of a hill and flow over the alluvium to Valley Creek.

The groundwater flow patterns of this area have been the subject of considerable study. A recent work by the USGS (Mull et al, 1988) describes dye tracing studies that were performed in the Elizabethtown area. Their studies include the documentation of groundwater flow direction towards Valley Creek and the suggestion that shallow groundwater flow follows a major fracture joint set that trends N40W. The springs in this study are approximately along this trend from the landfill/sinkhole.

Technical Approach

The basis of the technical approach for this investigation is a combination of modified procedures for monitoring groundwater quality in karst areas presented by Quinlin and Ewers (1985) and the standard investigation methods used for preremedial investigations conducted under authority of the USEPA. The essence of Quinlin's and Ewer's recommendations is that the majority of the groundwater flow in karst areas is in the dissolution features. The probability of intersecting these dissolution features with a monitoring well is very low. Therefore, the only effective method for groundwater monitoring is to locate all the springs and seeps along riverbanks and hillsides, plus other accessible points of intersection with the water table, and monitor those locations. Dye tracing studies should be an integral part of every sinkhole investigation, but they were not possible in this case, becuse the sinkhole had been filled in. There was no surface expression of the dissolution or collapse features.

This investigation consisted of the following parts:

1. Site reconnaissance

2. Project planning

3. Magnetic survey

4. Soil gas survey

5. Sampling of environmental media to include:
    a. surface soil
    b. subsurface soil
    c. spring water
    d. spring sediment
    e. municipal supply
    f. appropriate background for each media

6. Data reduction and report preparation

Site Reconnaissance

After reviewing the file, a trip was arranged to Elizabethtown to view the site, perform field geology studies, and talk to local officials. The field work was done in October, 1988, following a dry summer. The topographic map indicated the location of the sinkholes prior to being filled. The area is now a small industrial park with the main sinkhole area being a storage area for large cement drainage pipes. Across the Interstate, four springs were located at the base of the hill. A spring house was noted at one of the springs. Also noted was a dry spring. Two large sinkhole depressions were noted at the top of the hill, above the springs. Contact was made with the Elizabethtown Water Department to arrange permission to take water samples from the municipal springs and wells later in the year. The local office of the Department of Transportation was contacted regarding the availability of historical aerial photos that might aid in the planning of the investigation. While this office had some coverage, the investigators were referred to the headquarters office in Lexington for the most complete historical coverage of the area.

Project Planning

With the information provided by the file material and the field reconnaissance, the project plans were formulated. Because the sinkholes were used for illegal dumping and the contents and depth of burial were not documented, a magnetic survey would provide an nonintrusive method of investigation. Groundpenetrating radar would have been the first choice, but it was ruled out due to the high clay content of the soils. Additionally, the file materials indicated that a tanker truck may have been buried in the largest sinkhole. A magnetic survey would give some indication of its possible location, as well as provide some measure of safety for the subsurface soil samples to be taken later. The file material indicated that chlorinated compounds had been detected in the adjacent springs. It was assumed that these compounds were leaching from the sinkhole contents and that a soil gas survey would detect the presence of these compounds. This would ensure that soil samples were taken in the most appropriate location and allow the efficient investigation of such a large area. The presence of chlorinated compounds, specifically vinyl chloride, demanded that the sampling of the spring water and sediment be planned as a Level B exercise. Appropriate access was requested and the field work was performed in early December, 1988.

Magnetic Survey

The magnetic survey was conducted with a proton precision magnetometer. Survey grids were laid out above two sinkholes with 28 stations for the eastern sinkhole and 20 stations for the northern sinkhole.

(Access permission was not granted in time to perform a survey over the southern sinkhole). All appropriate calibrations were performed and standard methodologies were followed. The results of the eastern sinkhole indicated two positive anomalies in the center of the sinkhole and four negative anomalies around the edges of the grid. These results may be interpreted as indicating the presence of buried man-made, ferrous objects in the center of the sinkhole. The two positive anomalies align in such a way that they may be interpreted as indicating the presence of the suspected tanker truck. The results of the northern grid only indicated one positive anomaly along the southwest corner of the grid. This was interpreted as indicating buried man-made, ferrous objects only in the southwest area, with the rest being nonferrous fill material (USEPA, 1991).

Soil Gas Survey

The soil gas survey was conducted using a photoionization detector (HNu), which is designed for the direct reading of the presence of volatile organic compounds. Soil gas probes were installed to an approximate depth of three feet in eleven locations in the eastern sinkhole and five locations in the northern sinkhole. On the basis of this survey, locations of the soil samples were selected. A third survey was attempted in the area of the southern sinkhole. Access problems delayed the performance of work in this area. The weather became extremely unfavorable with temperatures in the 20s, blowing snow, and winds in excess of 15 miles per hour. An attempt was made to perform a soil gas survey, but the author believes that more favorable conditions would have allowed a more successful attempt.

Taking of Environmental Samples

A total of 17 soil and 11 water samples were taken during this investigation. The breakdown of the sample types follows:
* A total of nine surface and subsurface soil samples were taken from the three sinkholes.
* A total of four water and two sediment samples were taken from the five springs located at the base of the hill across the interstate. Lack of water or lack of sediment, combined with temperatures in the mid-teens, precluded the completion of the entire planned sampling program in this area. The cold temperatures did allow the downgrading to modified Level D due to the lack of volatilization as indicated by the air monitoring equipment.
* Six water samples were taken from the Elizabethtown municipal water system.
* A water sample and a sediment sample were taken as background from Valley Creek Reservoir located northeast of the site. A total of two surface and two subsurface soil samples were taken as background samples: one set onsite near the sinkholes and one set on the farm where the springs were located.

All samples were taken observing USEPA Region IV sampling protocol.

Analytical Results

Over 150 compounds, including volatiles, semivolatiles, pesticides, PCBs, acids/base/neutrals, and inorganics were analyzed as part of the USEPA-approved Contract Lab Program. There were numerous compounds present above detection limits. Some were present only in the soil samples taken above the sinkholes. Some were present only in the samples taken at the springs. The set of compounds that were detected in both locations numbered 20: 11 organic compounds (Table 1) and nine inorganic compounds (Table 2). The analytical results for this set of compounds are listed below with their respective sample type and maximum concentration detected.

TABLE 1
COMPARISON OF ORGANIC ANALYTICAL RESULTS
SOIL AND WATER SAMPLES
ELIZABETHTOWN, KENTUCKY

PARAMETERS (ug/kg)	BACKGROUND SOIL	BACKGROUND WATER	SINKHOLE SOIL (MAXIMUM)	SPRING SEDIMENT (MAXIMUM)	SPRING WATER (MAXIMUM)
ACETONE	12UJ	10U	320J	630J	
CARBON DISULFIDE	6U	5U	2J	13J	
1,1-DICHLOROETHANE	6U	5U	8	11J	110
1,2-DICHLOROETHENE	6U	5U	45J		520
1,1,1-TRICHLOROETHANE	6U	5U	6		67
TRICHLOROETHENE	6U	5U	2J		110
BENZENE	6U	5U	42		4J
METHYL ISOBUTYL KETONE	12UJ	10U	220J		3J
TETRACHLOROETHENE	6U	5U	20		26
TOTAL XYLENES	6U	5U	66N		8J
TOLUENE	6U	5U	190		4J

TABLE 2
COMPARISON OF INORGANIC ANALYTICAL RESULTS
SOIL AND WATER SAMPLES
ELIZABETHTOWN, KENTUCKY

PARAMETERS (ug/kg)	BACKGROUND SOIL	BACKGROUND WATER	SINKHOLE SOIL (MAXIMUM)	SPRING SEDIMENT (MAXIMUM)	SPRING WATER (MAXIMUM)
CADMIUM	1.1U		2.6	19,000	
CHROMIUM	21J	10U	34J	33J	18
COPPER	6.7	22U	190	11,000	26
IRON	17,000	160U	63,000	24,000	76,000
LEAD	15J	60UJ	1100J	9700	170J
MANGANESE	410	65	410	340	4500
NICKEL	13J	16U	130J	24J	26
VANADIUM	30U	16U	92		39
ZINC	30U		860	12,000	39

NOTES

	Material analyzed for but not detected above minimum quantitation limit (MQL)
J	Estimated value
U	Material analyzed for but not detected. The number given is the MQL
N	Presumptive evidence for the presence of material

Conclusions

The objectives of this investigation included the source characterization of suspected waste areas, the identification of releases or suspected releases to the environment of hazardous materials, and the establishment of a hydrogeologic interconnection between the sinkholes/landfill and the adjacent springs. The varied analyses did characterize the sources as to the type of material that was placed in the sinkhole/landfill. The spring water and sediment analyses did indicate that hazardous materials were being released into the environment. The matching of 20 compounds between the sinkhole/landfill and the springs does indicate a connection between the two areas. Quinlin's and Ewer's basic premise that the key to monitoring groundwater quality in karst areas is to monitor the springs has been demonstrated.

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Q. How do you know each or any of the springs or surface water sites is hydraulically connected to the landfills?

A. There have been no die traces performed in this exact area near Elizabethtown. Given the hazardous nature of the spring water, the flushing of the karst system would be a questionable procedure. The connection between springs and the landfills is demonstrated by the empirical evidence of the 20 compounds which were detected in both the springs and the landfill soils.

LAND-USE PLANNING AND WATERSHED PROTECTION IN KARST TERRANES

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Abstract

It is critical that the special characteristics of a karst watershed (groundwater basin) be examined when any development within it is to be considered. Local planning boards are most likely not aware of the radically different flow dynamics of karsts, and therefore they may unwittingly allow development that leads to massive contamination. A practical approach for ascertaining the sensitivity of a basin to groundwater contamination, utilizing relatively easily implemented and practical procedures, is described; it will aid land-use planners. Although the approach is kept fairly basic, it is comprehensive. Not only is the hydrology of the karst aquifer characterized, but any present and future sources of contamination are evaluated and compared with the estimated assimilative capacity of the aquifer.

An evaluation of the potential impacts of development should also examine the types and amounts of contaminants which could be introduced into a system, the likely chemical partitioning of these contaminants, and the combined loading to the aquifer or stream receptor. Consideration of the present health of receptor streams should be part of the process. Water chemistry parameters, combined with indices of biotic integrity and ecosystem health, can be used to evaluate the present condition of the ecosystem. This important base-line information should then be incorporated into a comprehensive and practical management strategy.

The Mill Pond karst basin, situated in east-central New York State, is an example of an environmentally sensitive karst aquifer. Recently zoned as Rural Commercial, this groundwater basin will be used to illustrate the practical application of accepted hydrologic techniques in land-use planning. Reference to several other localities will provide further insight into the evaluation process.

Organization

This paper is organized as follows: 1) Introduction, 2) The land-use planners point of view: why does a karst region deserve special consideration, 3) Assimilative capacity of the karst aquifer: how much is too much, 4) A typical development proposal, 5) Characterization necessary for zoning decisions, 6) Practical land-use planning: two examples, 7) Advanced testing and analysis to define a karst system requiring a karst hydrologist, 8) Political realities of the land-use permitting process, and 9) Who to look to for help.

Introduction

Groundwater in soil and most fractured bedrock aquifers moves slowly, enabling contaminants to be partially treated and diluted. Karst aquifers (comprised of dissolutionally enlarged fractures and cave passages), on the other hand, are often characterized by appreciable and sometimes rapid groundwater flow (Quinlan, et al., 1991). They have virtually no ability to treat water-borne contaminants, instead they merely transmit contaminants, much as a sewer pipe would (Ford and Williams, 1989). In some karst aquifers, contaminated groundwater can move from one end of the aquifer to the other in less than one day. Failure to recognize the nature of karst systems, and to plan accordingly, is likely to lead to the degradation of the aquifer, its springs, and its dependent aquatic ecosystem. Chemical loading, and possible resulting deoxygenation of karst groundwaters during periods of base flow, may result in destruction of fauna, including game fish, downstream of aquifer discharge points (i.e., springs). Contaminant loading of a karst system and its receiving stream may also result in cave habitat destruction, impairment of the character or quality of important aesthetic and recreational water resources enjoyed by residents and the community (e.g., streams), and the possible infringement of riparian rights of domestic and agricultural users. It is a difficult, expensive, and sometimes an impossible proposition to remediate a karst aquifer after it and its receiving stream have been degraded. The development of an area must be within the natural constraints of its geology and hydrology.

Urban expansion is causing greater pressure for land development in many karst regions. It is critical that the special concerns of karst watersheds be addressed by local planning boards, environmental conservation departments, and other state agencies. For example, towns in the Helderberg Plateau area of New York continue to receive applications for single residences and planned unit developments to be atop maturely karstified limestone aquifers. Optimal land-use planning would dictate the evaluation of an entire karst system before permitting additional development in any isolated segment of an aquifer. This would include characterization of the extent of the aquifer watershed (its basin boundaries), geology, subsurface flow paths, depth to groundwater, discharge area, and base flow. Only after such characterization is complete, can the net effects of multiple contaminant inputs on the assimilative capacity of an aquifer be judiciously evaluated. It is important that evaluations of this nature not only define the dynamics of a karst system, but that they then utilize the knowledge gained to protect subsurface and surface water resources. Unfortunately, many of the conclusions about what is necessary for protection of the aquifer will, like zoning, be regarded by some to be an illegal taking of their rights.

Successful land-use planning in karst terranes involves two important criteria. First, there needs to be detailed analysis of the area's geology, hydrology, and biology. Of equal or greater importance, however, is the willingness of land-use planners and their counterparts to become knowledgeable and involve themselves with the actual scientific issues and to resist strong pressures to permit land-use beyond the assimilative capacity of the karst aquifer. Without these elements, the analytical process may amount to little more than good intentions. A positive outcome of this process can be obtained when karst hydrologists work with land-use planners.

The Land-Use Planners Point Of View: Why Does A Karst Region Deserve Special Consideration?

The cleanup or remediation of contaminated water supplies or streams can cost a municipality upwards of a million dollars. Prudent land-use planning can avoid this; not only eliminating the unnecessary and avoidable expenditure of taxpayers' dollars, but also saving a great deal of environmental destruction of a precious resource. Excessive development in environmentally sensitive karst groundwater basins may lead to contamination, flooding, and subsidence problems. This paper discusses water quality concerns. Too much of any contaminant introduced into a karst basin is likely to result in degradation of groundwater resources and the water body (i.e., stream, river, or lake) receiving the basin's discharge. Contaminant inputs to karst groundwater basins from sources such as leach fields, landfills, sinkhole dumps, animal feed lots, and industrial disposal operations may combine to exceed the assimilative capacity of the aquifer and receiving springs and streams. Since anything that goes into a karst system comes out virtually untreated, dissolution conduits (e.g., cave passages) can function as large flowing sewer pipes. Worst-case examples of this problem are numerous and include visual and olfactory evidence of septic waste and industrial contaminants (i.e., gasoline, toilet paper), destroying ecosystems so that they support little to no life. This now-polluted water is often critical for water supply, land value, and recreational purposes (e.g., fishing, swimming, aesthetic quality). A further consideration may be adverse human and animal health impacts that may result from ingestion or contact.

Once particular land-uses are permitted and in place (e.g., large housing clusters, factories, dry cleaners, service stations, waste treatment centers, feed lots, liquid waste spreading, etc.) it is difficult, if not impossible to terminate or relocate them and retro-fitting facilities for reduced effluent generation can be costly. Yet, as history has repeatedly shown, failure to properly consider the ramifications of excessive contaminant loading can result in multi-million dollar evaluation and remediation costs. In such settings, where karst springs and related surface streams are used for residential, agricultural, and municipal water supplies, the first action required is the replacement of the primary water supply. This involves a lengthy, and potentially costly, process of alternative water supply source and site selection, full geologic, hydrologic and chemical characterization, design and construction of necessary treatment and delivery systems. In some cases, the contaminated supply can be treated or piping extended to alternative existing sources. This situation is happening with increasing frequency and often costs millions of dollars. The replacement of a water supply can sometimes require years of planning, exploring funding options, evaluation, and set-up. One potential solution, perhaps of equal cost, is the funding, planning, and installation of a sewer district. Another reality which may bear on the land-use planning issue is the legal responsibility of the planners. This consideration may weigh heavily in the future as litigation and pollution-related costs rise. Even now, corporate executives are being held personally responsible for their actions; with some spending time in jail. The best time for planning for responsible land-use is prior to significant developmental pressures.

Assimilative Capacity Of The Karst Aquifer: How Much Is Too Much?

This is one of the more important questions in land-use planning and watershed protection decisions. The answer to this question should logically dictate what are reasonable and prudent land-uses in a karst basin. Ultimately, the relative assimilative (carrying) capacity of individual karst basins must be determined on a case by case basis. In many aquifers, especially those capable of limited contaminant dilution, the assimilative capacity is nearly zero. Many factors must be considered when addressing what types and amounts of contaminants should be permissible in a groundwater basin within a karst terrane. Some of these include: 1) presence or absence of sewers, 2) the type and amount of contaminants (present and planned) incident to the karst system, 3) the amount of waste pre-treatment, if any, 4) the amount of natural cleansing, if any, imparted by overlying soil or bedrock units prior to conduit entry, 5) an assessment of artificial diversion of water into or out of the basin, 6) the maturity and sensitivity of the karst aquifer, including the extent and importance of the upper part of the percolation zone (epikarst), 7) the physical characteristics of conduit flow paths during base flow conditions (e.g., sluggish movement through intermittent pools), 8) residence and travel time of contaminants in subsurface and surface flow paths, 9) water temperature, 10) the base flow and draught flow conditions in the aquifer, 11) the relative sensitivity and discharge of the receiving waterway (does the spring comprise the total flow of a stream in the headwaters of a watershed, or does it enter a stream or water body capable of providing significant dilution?), 12) the rate of deoxygenation and reaeration of the receiving water body, 13) the contaminant loading and biologic health present in the receiving water body, and 14) the current and planned uses of aquifer and receiving stream water.

The dynamics of the contaminant degradation and dilution which occurs, both within and downstream of springs, is critical in the assessment of the assimilative capacity of a karst basin. The quality of streams and rivers used for waste-water dilution depends on natural self-purification to assimilate wastes and restore its own quality. A waterway's capacity to recover from influent waste is largely determined by channel morphology and climatic conditions (Hammer, 1975). Standard environmental calculations inclusive of the rates of biochemical oxygen demand (BOD) and rates of deoxygenation and reaeration can be used as an aid in establishing acceptable aquifer loading criteria for some contaminants; although it must be recognized that both flow and atmospheric conditions in the karst system may vary appreciably from that of surface water. This type of procedure can be particularly useful for back-calculating likely effects of sewage disposal, such as fish kills and eutrophication. Base line chemical and hydrologic data is essential for this type of calculation.

A Typical Development Proposal

In order to address the types of impacts existing or planned development may have on a karst area, a number of facts and interpretations must be considered. Once an isolated area is proposed for a specific land-use, the planner must determine whether it is hydrologically reasonable and prudent to base planning decisions on a site-specific assessment. More often than not the parcel in question is really only a small segment of a much larger geologic and hydrologic framework.

In the rural Town of New Scotland, New York, for example, a 28 acre parcel was proposed for Rural Commercial zoning. The parcel is situated proximal to the Stewart's label on Figure 1, and incorporates a portion of a junk yard property and a segment of the Onesquethaw Creek channel. The proposed zoning allowed for multiple uses of the parcel including: shopping plazas, laundromats, public works garages, solid waste disposal facilities, sewerage systems, car washes, restaurants, etc. The maximum allowable building size proposed was 9 acres. This rural area does not have a sewer district.

The area is maturely karstified as shown by a lack of surface drainage for most of the year. Much of the watershed that includes this parcel consists of exposed limestone, with a thin and sometimes absent soil-mantle. An extensive cave system has been explored and mapped throughout part of the watershed. There are also wells which intercept water flowing in cave passages which are accessible through natural fissures. These wells and others that encounter caves during drilling, are referred to as conduit wells. Cave streams and the related spring (aquifer discharge point) have flashy hydrograph responses to storm and runoff events, thus indicating turbulent flow in conduits typical of karst aquifers.

In an effort to promote a convenience store (Stewart's), the Town decided to advance the Rural Commercial segment of the Comprehensive Land Use Plan still under preparation. The Town did not contract for any detailed studies of the hydrology and geology of the site or of its surroundings. A public hearing was held, with the author presenting the results of his geologic and hydrologic study of the karst basin. The basin was characterized by him as environmentally extremely sensitive. Soon thereafter the Rural Commercial district was approved, Stewart's applied for a building permit, which was granted and the convenience store has now been constructed. Although no comprehensive basin-wide zoning approach was adapted, a written and oral record of known environmental concerns was created.

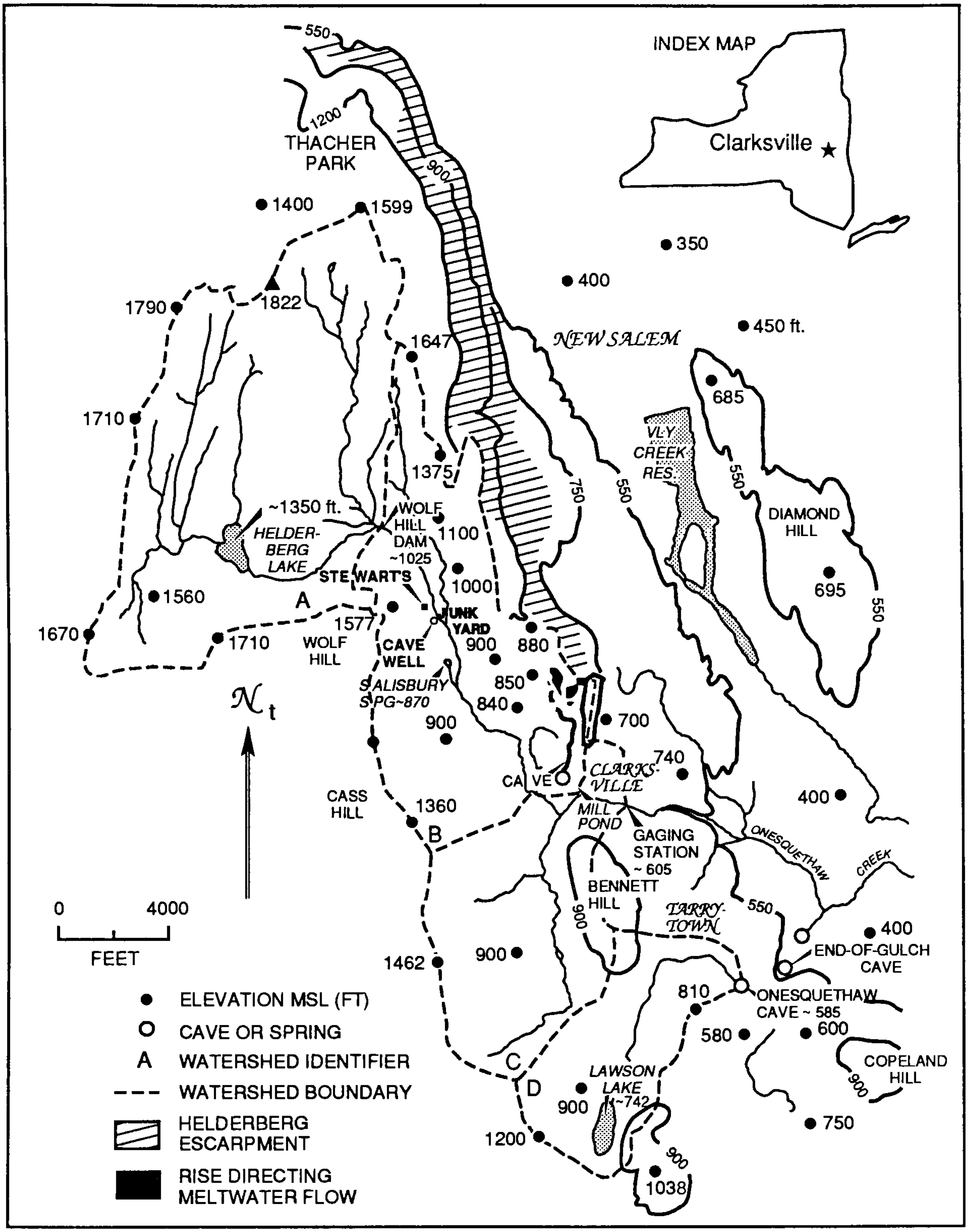

FIGURE 1: TOPOGRAPHY, DRAINAGE BASINS, AND SELECTED FEATURES ALONG THE HELDERBERG ESCARPMENT, ALBANY COUNTY, NEW YORK. WATERSHED BOUNDARIES BASED ON TOPOGRAPHY, GEOLOGY, AND DYE TRACES. CONTOUR INTERVALS VARIABLE FOR CLARITY.

Characterization Necessary For Zoning Decisions

Evidence of the Presence of Subterranean Karst Networks

The first step toward protecting karst aquifers is to determine whether carbonate or other soluble (i.e., halite, gypsum) bedrock is present. When the answer is in the affirmative, an assessment of the nature of the aquifer is in order. Physical evidence of the presence of subterranean karst networks includes: 1) large and small enterable caves; 2) sinkholes; 3) areas of exposed carbonate bedrock with numerous dissolutionally enlarged joints (fractures); 4) limited to non-existent surface drainage in select areas; 5) piracy (sinking) of surface water into sinkholes and streambeds; 6) presence of open conduit and alluviated springs; 7) presence of overflow springs; 8) variable chemical composition of springs and conduit wells; 9) rapid subsurface flow as documented by tracer tests; 10) boreholes having encountered large and/or small conduits; 11) loss of circulation fluids into conduits found during drilling operations; 12) highly variable and abnormally large water table fluctuations in some bedrock wells (especially those with documented cavities).

Karst Inventory

A full inventory of karst features (as indicated above) is needed to monitor the karst systems recharge, discharge, water quality, and environmental sensitivity. Such an inventory is important in focussing future site work, in land-use planning, and in determining appropriate sites for aquifer characterization (e.g., sinkholes, springs, conduit wells). A number of additional sources of information should be explored in assessing the karst features of an area.

Published geologic and hydrologic reports are the most valuable sources of information, but recent and historic topographic maps are also useful. Historic topographic maps often show sinkholes or sinking streams which recent land-use practices may have obliterated. Examples include the filling in of sinkholes with soil or waste materials and physical alteration of sinking streams to create ponds. Although topographic maps are good sources of information, they often have many errors. Errors can include the connecting of stream segments where streams are pirated into the subsurface or the failure to portray numerous shallow sinkholes because they fall within the map's contour interval, have been filled in, or were not recognized by the photo-interpreter making the topographic map.

When available, historic and recent aerial photography can supplement or correct topographic map errors. Low-cost stereo pairs can be obtained from a variety of sources including the United States Department of Agriculture, the United States Geological Survey (USGS), the National Archives, and assorted state and local offices. Often, local aerial photography firms can provide previously flown stereo coverage. Ideally, large scale photography taken at a time of year without leaf or cloud cover is desirable. Interpretation of karst features is best achieved through the use of a mirror stereoscope with attached binoculars, although a simple lens stereoscope will suffice. Aerial photographs can also be used to define the topographic boundaries of a watershed and assist in determining sinkhole growth rates, but topographic boundaries do not necessarily coincide with groundwater basin boundaries. Usually they do not.

One of the more important resources for learning of karst features in an area is people: local homeowners, farmers, hunters, fishermen, well drillers, geologists, and children. Those individuals who roam or work an area for many years can often provide information which might otherwise be overlooked. Examples include small sinkholes filled in by farmers, turbid well water coincident with storm events, caves played in by youths, and wells intersecting conduits. Drillers' and geologists' logs from well and foundation studies may provide valuable insight into a karst system. Occasionally a well may encounter a cavity, a zone of increased permeability, or a zone where drilling fluid circulation was lost. These locations, when properly monitored, may provide important information on the hydraulics and water chemistry in a system. This is particularly true when they are determined to be within a cave conduit. Such conduits may also be used as tracer injection or monitoring points.

Homeowners can often provide important information about a karst aquifer system. In the Clarksville, New York area they identified two "wells" which were actually pipes placed within cave passages (see Cave Well on Fig. 1). Both such wells, and their ultimate discharge point at the Mill Pond, had historic diesel fuel contaminant problems.

Finally, there is no substitute for hydrogeologic assessment and objective field reconnaissance. Some of the important features and items which should be included in a karst inventory are the location of: 1) sinkholes; 2) caves; 3) springs; 4) sinking streams; 5) seasonally active overflow springs; 6) limited to non-existent surface drainage; 7) conduit wells; and 8) evaluation of existing chemical analyses of wells, springs, and streams.

As karst features are added to the inventory, it is necessary to comprehensively evaluate the importance of each and how they combine to define the hydrology of an area. Base maps are valuable for plotting these features. It is also important to place known or suspected contaminant sources on this base map.

Karst Basin Evaluation

Evaluating the hydrology and geology in any karst basin requires a multi-phased approach. Prioritization of these phases is dependent on: (1) the urgency of the particular land-use plan (2) the potential threat to aquifer and surface flow systems (3) on the amount of information already known about the karst terrane (4) time, and last but not least, (5) money. Evaluation of a karst groundwater basin should include all or many of the components discussed in this paper, depending upon the scope of the anticipated project, and its budget. Often, different aspects of the evaluation may be conducted concurrently. It is important that evaluations of this nature not only define the dynamics of a karst system, but that the information then be utilized to protect the subsurface and surface water resources.

The entire groundwater basin needs to be characterized, not only the small segment in question. Land-use planners who do not take this broad view tend to have serious unforeseen problems. Small land-use projects with limited or no waste streams may not require complete definition of all these elements. However, this is generally only the case where a large base flow is maintained year round and/or a significant soil-mantle is present.

Watershed Delineation

In order to assess the potential impacts of waste streams leaving an area of proposed development, it is first necessary to define the surface and subsurface watershed boundaries. This assessment includes at least a partial definition of surface and subsurface flow components. Although the ultimate definition of a groundwater basin will be portrayed as a single irregular watershed boundary, it can be useful in assessing vulnerable aquifer segments and prudent land-uses to determine this boundary via a three phased approach; the topographic basin, the lithologic basin, and the hydrologic or groundwater basin. The term "lithologic basin" is introduced here for its utility in locating particularly vulnerable portions of karst basins, where carbonates are bounded by less soluble and possibly deformed geologic units, and for its function in isolating directions to search for potential aquifer exit pathways. These boundaries may be defined through the use of published geologic maps, low-altitude stereo aerial photography, USGS topographic maps, tracer studies and, where necessary, by detailed geologic mapping.

Topographic Basin

A preliminary assessment of an area's surface water basin may easily be made with topographic maps and supplemented with low-altitude stereo aerial photography. The standard USGS 7.5-minute maps at a scale of 1 inch = 2,000 feet is practical for most problems. Annotations may be made on the topographic map while using a stereoscope to interpret aerial photography. It is always prudent to field check all areas, especially those which are questionable. The surface watershed boundaries determined from this method can be useful as a first step in delineating the actual groundwater basin boundaries. Further refinement can be achieved through the evaluation of other geologic and hydrologic factors; the lithologic and hydrologic basin.

Lithologic Basin

The geology of an area may make groundwater basin boundaries non-coincident with surface water basin boundaries; meaning that subsurface flow routes may extend far beyond, or even in another direction from that of a surface stream. The lower permeability of non-carbonate bedrock units generally constrains groundwater flow, regardless of whether it results from a stratigraphic or structural break (e.g., faulting, folding). It is often possible to expedite the watershed evaluation process by maximizing the use of existing geologic and hydrologic maps of the area. These maps can then be used to predict structural or lithologic (relating to the physical character of a rock unit) controls on watershed boundaries. If no work has been conducted, it may be necessary to do geologic mapping. For example, detailed geologic mapping outside the K-25 area at Oak Ridge, Tennessee has revealed that relatively impermeable non-carbonate units (Rome Formation and Nolichucky Shale) form impermeable boundaries to aquifer waters within the thick carbonate sequence surrounding most of the K-25 complex area (Fig. 2). Upturned Rome and Nolichucky beds force all subsurface water discharging from the drainage basin to move along valley trend which is coincident with strike. Interpretations like this are essential for predicting exit pathways for contaminants and for limiting the area to be searched for spring resurgence points (Palmer, 1986).

The map in figure 2 illustrates the end product of such a geologic assessment for the Oak Ridge, Tennessee area. The original geologic map was subdivided into 33 stratigraphic units. Based on lithology, carbonate bed thickness, and relative bedrock solubility, geologic formations were divided into 2 hydrologic units; carbonate and non-carbonate. Thus, the potential for major karst development within a topographic-based watershed boundary was further subdivided. The cross section in figure 2 shows how bedding or faults may further constrain karst development and contaminant exit pathways. This map delineates several parallel and distinct karst flow systems. Analysis of geologic logs in a portion of one of these carbonate bands revealed that 34 percent of coreholes encountered voids, providing evidence for mature karstification.

In another example, detailed geologic mapping in the Clarksville, New York area revealed that the Esopus Shale had been thrust up against the Onondaga Limestone (Fig. 3). This uplifted wall of impermeable shale (sub-parallel to the east border of the map) forces all subsurface water draining southeast in drainage basin B of Figure 1 to flow south proximal to this fault zone (Rubin, 1991a), resurging at the Mill Pond spring. Without this knowledge, residents east of the fault incorrectly believed that septic contamination of their well water was from liquid farm waste spreading which frequently enters the cave system.

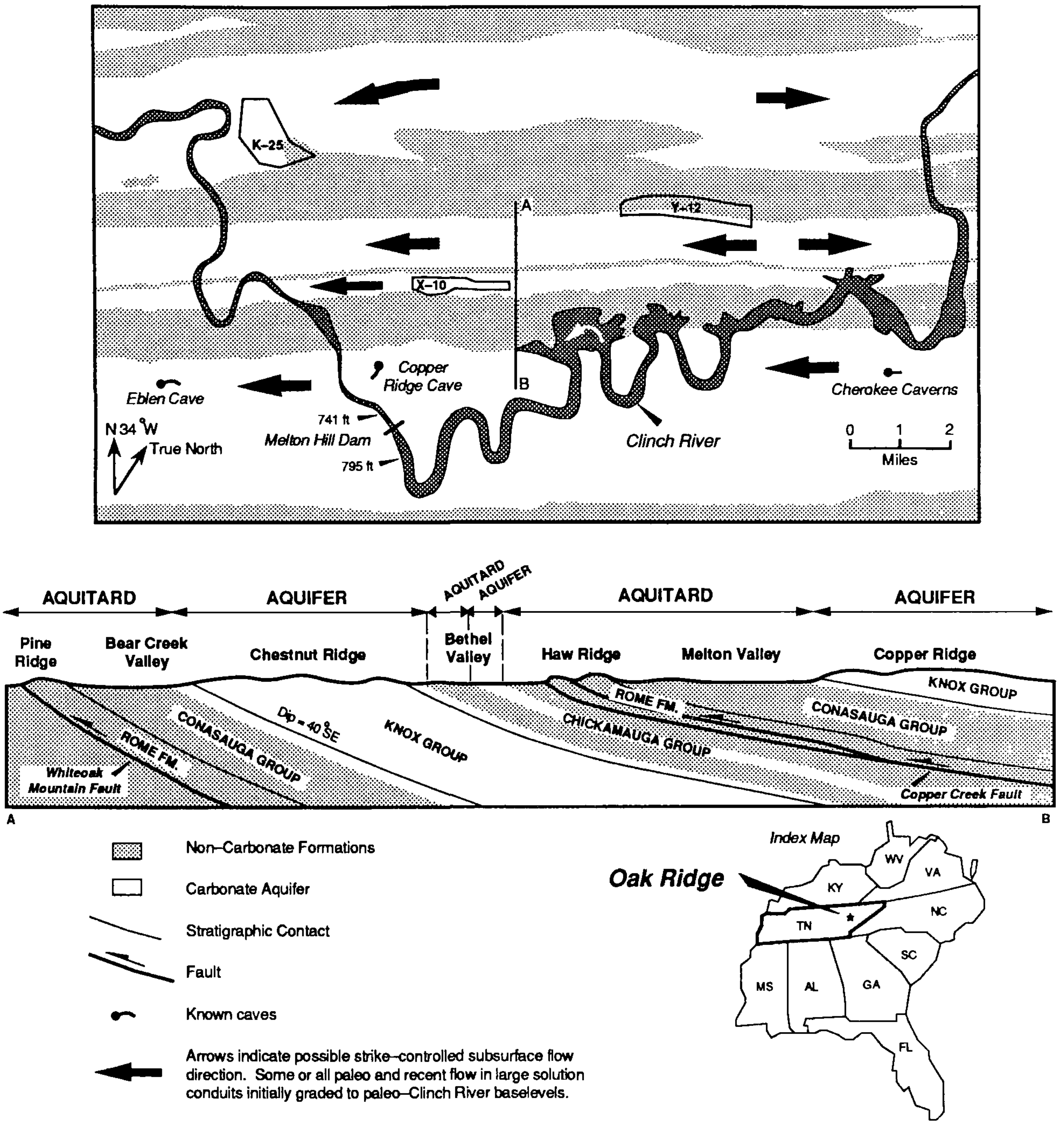

Figure 2: Cave development and possible contaminant exit pathways in structurally deformed regions are often controlled by bedrock lithology and former or recent baselevels. One aspect of characterizing subsurface flow in carbonate aquifers requires locating and monitoring springs. Spring locations may be constrained by strike–controlled lithologic boundaries and river baselevels. In Oak Ridge, TN, discharge points may be located adjacent to the Clinch River towards the southwest and/or northeast. Buried or relict karst, which may still function as dominant subsurface drainage routes, can be defined through geophysical, boring, and tracer studies. (PAR / PJL 10/28/91 ORNL Drawing)

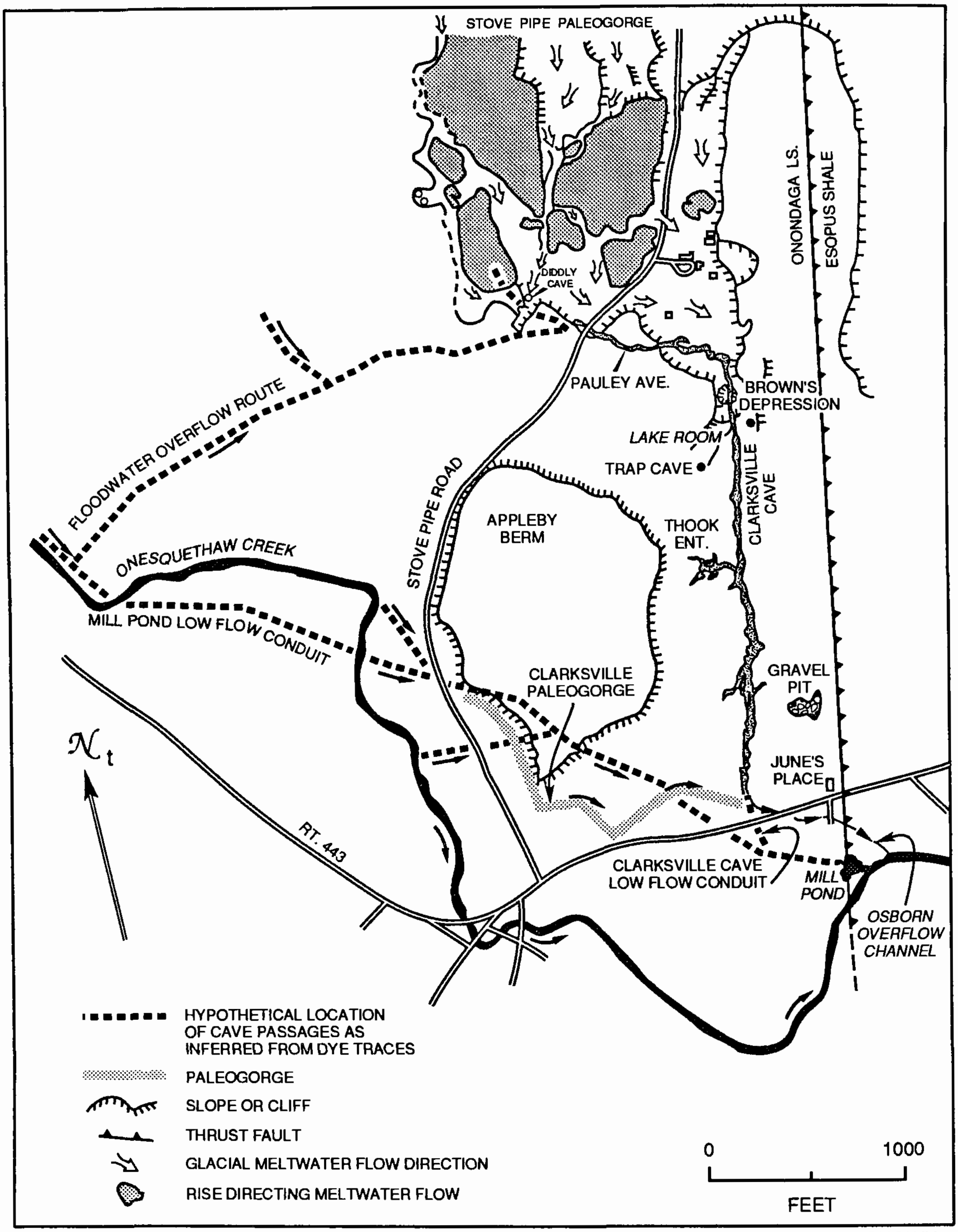

FIGURE 3: CONFIGURATION OF DRAINAGE IN THE VICINITY OF CLARKSVILLE, NEW YORK. SHOWN ARE CLARKSVILLE CAVE AND PRESENT AND PAST ROUTES OF FLOW. DYE TRACES ARE OFTEN IMPORTANT FOR DETERMINING SUBSURFACE FLOW ROUTES ESTABLISHED IN FORMER GEOLOGIC PERIODS.

Hydrologic (Groundwater) Basin

The ultimate definition of a karst groundwater basin requires specific knowledge of the physical boundaries of all water entering and leaving the basin. Definition of topographic and lithologic basins aids in defining the boundaries of groundwater basins, however, topographic divides rarely coincide with subsurface divides in karst terranes (Quinlan, 1989). Sometimes, during and after floods, some groundwater may be diverted from one groundwater basin to one or more others adjacent to it (Quinlan, 1990, Figs. 2 and 3). Regions with complex geomorphic histories, such as Oak Ridge, Tennessee (Fig. 2), may require advanced testing and analysis to define karst systems grading to multiple baselevels. Structural and lithologic controls, however, will still impose certain limits on the extent and boundaries of karst basins, as may hydrologic baselevels.

All karst aquifers have at least one discharge point. Discharge points (resurgences) generally occur as springs at a local or regional baselevel feature such as a stream, river, lake, or ocean. Spring locations may be predicted by evaluating the combined topographic and geologic map. Palmer (1986), documents how bedding orientation may be used to predict likely directions of contaminant transport and groundwater flow. In many settings, it is essential to document the actual subsurface flow direction, destination, and velocity with one or more tracer tests. Where little surface flow is present, it is often possible to force flush a tracer through the karst system. A pre-injection systematic search of potential baselevel outlet areas may serve to locate springs. Local farmers, homeowners, hunters, and fishermen are often excellent sources of information, but there is no substitute for field work to search for them.

The type and thickness of soil cover in an area will influence the amount and extent of infiltration (downward percolating water) reaching an underlying karst aquifer. A detailed examination of all surface streams and their tributaries may reveal that little to no flow is pirated (diverted) to the subsurface. In many karst areas, however, soils perch surface streams for short distances prior to their partial or complete loss to the subsurface. Infiltration areas may be detected by conducting seepage velocity (discharge) measurements at successive points along streams where a losing component is suspected. One common situation found in karst terranes is that of surface flow over impermeable soil or bedrock formations present in the higher reaches of a karst watershed. This water flows downhill until it sinks near an exposed limestone contact, where numerous sinkholes are often aligned.

When considering acceptable land-uses and their resultant contaminant loading, it is important, when possible, to evaluate the ability of the bedrock or soil present to treat infiltrating septic or other waste.

Practical Land Use Planning: Two Examples

An excellent example of a karst area requiring specialized land-use planning in order to avoid both significant aquifer and surface stream degradation is a broad karst aquifer present in the Clarksville area (Fig. 1), specifically the Mill Pond karst basin which forms the headwaters of the Onesquethaw Creek. Since urban expansion was extending into the surface water basin, a study was undertaken to define potential development concerns. No funding was available, so the steps followed had to be capable of reliably characterizing potential contaminant concerns with limited manpower and resources. This type of limited approach may be all that many towns will ever have the resources to have performed. Proper characterization of a karst groundwater basin will require a qualified individual or firm, as discussed in the closing "Who to look to for help" section of this paper.

The key issue for planners in karst areas should be the water quality of the aquifer and its receiving springs and streams (e.g., including water supply, ecosystem, and aesthetic concerns). In order to objectively appraise land-uses and their potential impacts in karst basins, it is necessary to define the hydrologic boundaries (including all discharge points), aquifer and receiving stream vulnerability, and many of the items discussed in the assimilative capacity section of this article. To evaluate potential impacts in the Mill Pond karst basin, a stepwise strategy was followed, with various steps occurring concurrently. These are numbered for ease of reference.

1) Topographic maps were obtained of the area. A first approximation of the surface water basin was defined to determine approximate boundaries of the area of concern.

2) Existing published and unpublished geologic and hydrologic reports, chemical analyses, cave maps, and stream flow data were collected to provide a current basis for evaluating existing conditions.

3) Maps of the bedrock geology and soil thickness were obtained from existing sources. The area of carbonate bedrock was overlain on the surface water basin obtained in step 1), above. It was determined that the geology was such that no subsurface flow in carbonates was likely to enter from outside the topographic basin. This possibility must be evaluated on a case by case basis.

4) All known karst features were inventoried. These included features readily apparent in the basin (e.g., sinking streams, sinkholes, caves, springs) and others learned of from local residents (e.g., conduit wells, caves). Past and present contaminant problems were also noted as part of the inventory. Most state regulatory programs maintain a file of regulated environmental problem sites.

5) Existing geologic and hydrologic reports and maps were evaluated for likely aquifer discharge points. Baselevel discharge areas were searched for springs.

6) The location of significant karst features were field-checked and placed on the working base map.

7) Caves were mapped. (This step may not always be necessary.)

8) A major fault was identified in a cave. This discovery suggested that a faulted shale unit may be forming a groundwater basin boundary to the east. The fault was not present on published geologic maps, but was found and readily mapped in the field (Fig. 3).

9) Existing stereo aerial photographs were obtained and utilized to further refine the geologic contacts in the basin. Several additional karst features recognized on the aerial photographs were accurately plotted on the base map.

10) Known or suspected contaminant sources were plotted on the base map (e.g., junk yard).

11) Areas of dense housing (likely contaminant sources) were also plotted.

12) Several qualitative tracer tests were conducted to confirm suspected flow paths and roughly approximate subsurface travel times. It was determined that the gentle southwesterly dip of the bedrock present in the Mill Pond aquifer fails to direct all subsurface flow in this direction. Instead, the significantly higher surface topography to the southwest (Wolf Hill and Cass Hill) retards dissolution in this direction, in favor of the 1.3° apparent dip between Wolf Hill Dam and a baselevel discharge point at a spring in the Mill Pond. Tracer studies generally verified

predicted flow paths, at least during periods of low discharge when the entire headwaters of the Onesquethaw Creek resurge from the Mill Pond. However, tracer studies also documented the unexpected easterly diversion of moderate to high discharge waters through Pauley Avenue in Clarksville Cave (Fig. 3).

13) A gaging station was established downstream of all karst springs and at a point beyond which no subsurface flow from carbonates was possible. This was monitored daily for 15 months, and periodically thereafter during periods of low and high flow. Monitoring for this duration may be unnecessary, since it is the lowest flows that were of the greatest concern for estimating the likely assimilative capacity for contaminants. Base flow from the basin was gaged at less than 0.1 cubic feet per second (50 gallons per minute).

14) The karst system was evaluated based on the results of the investigations conducted and analysis of stream hydrographs. Additional information could also be obtained on the nature of the system using relatively simple physical properties (e.g., specific conductance, temperature, and turbidity) of spring waters. Quinlan et al. (1991) have put forth a simple, practical, and reliable evaluation technique for classifying the vulnerability of carbonate aquifers based on the coefficient of variation of specific conductance of spring waters. The technique evaluates the net effect of an aquifer's recharge, storage and subsurface flowpath characteristics. A fourth category, which might be evaluated as part of the vulnerability classification, is aquifer base flow. In the nomenclature of Quinlan et al. (1991), the Mill Pond karst aquifer may be classified as hypersensitive.

15) The environmental sensitivity of the karst basin was ascertained, placing particular emphasis on the vulnerability and likely assimilative capacity of the groundwater basin. As part of this process, consideration was given to what adverse effects might result from contaminant types which would be permissible within the broad Rural Commercial District guidelines.

16) Findings of the investigation and concerns regarding contaminant loading were shared with the town planning board (designated lead agency), town engineers, local and state agencies, local environmental groups, and the press via reports, letters, and telephone discussions. The findings were also presented at a public hearing.

17) A final step in the process is to work with the agencies involved to draft a management plan adopting land-uses within the geologic and hydrologic constraints of the karst basin.

The Mill Pond watershed was subdivided into two parts: A) the 3,075 acre portion located upstream of Wolf Hill Dam, and B) the 2,050 acre portion located downstream of Wolf Hill Dam. The boundaries of the aquifer were determined to extend to the north and northwest of the basin's resurgence point at the Mill Pond (Fig. 3). The farthest boundary of the Mill Pond groundwater basin lies some 2.4 miles to the northwest, proximal to the Wolf Hill Dam on the Onesquethaw Creek. The boundaries of the catchment basin are depicted in bold dashed lines. The upstream part is comprised of insoluble rock, with much or all discharge being seasonally diverted to the Vly Creek Reservoir. The downstream part of the Mill Pond watershed has features characteristic of karst terranes. These include sinking streams, limited surface drainage, solutionally enlarged joints, sinkholes, and the Clarksville Cave system. During most of the year, water entering that portion of the watershed that is downstream of the Wolf Hill Dam is pirated into the Onondaga Limestone. Tracer tests and in-cave stream gaging indicate that flow is turbulent and has a maximum straight-line flow velocity of 5.3 km/hr (3.3

mph). During periods of high discharge, contaminants have the ability to travel from one end of the aquifer to the other in less than one hour.

A detailed analysis of the flow dynamics present in the Mill Pond karst basin was conducted. A gaging station was established in Onesquethaw Creek (Fig. 1) in order to examine the relationship between in-cave discharge and surface-watershed discharge. This was monitored twice daily for 15 months, more frequently during flood events, and periodically for 4 years thereafter during major runoff events (Fig. 4). Stream discharge was measured at 13 different stages. Curvilinear regression was then utilized to establish a series of multi-order equations that could be used to correlate stage height with discharge. The greatest discharge recorded for Onesquethaw Creek during the course of this study was approximately 1340 cfs (600,000 gpm). This occurred on March 15, 1986 at 3:00 am following heavy rains (≈2.7 in.) on a 15-inch snow pack. Daily monitoring of stream stage in Clarksville Cave for the same 15-month period revealed that a direct correlation exists between this discharge and that in Onesquethaw Creek. Approximately 8 percent of flood-peak discharge in Onesquethaw Creek flows through Clarksville Cave.

In this example, it is clear that major land-use in the central portion of the basin (e.g., Rural Commercial District), where almost no soil-mantle is present above heavily jointed limestone pavement, may have a significant environmental impact on the basin's receiving stream. Land-uses here which directly and/or continuously discharge contaminants to the conduit system should be carefully evaluated. In this setting, contaminants will quickly enter the fractured limestone and shallow underlying cave passage, where they will be directly piped to the Mill Pond spring and the Onesquethaw Creek. Continuous waste streams are more likely to adversely impact this geologic setting than those which enter the aquifer during times of saturated soil moisture conditions, high runoff, and infiltration. However, it should be noted that one-time or infrequent chemical slugs can, in some karst aquifers, cause greater problems than lower controlled or permitted releases.

The findings of this investigation were not used by the Town to develop a new overall master plan. Instead, the Town ultimately decided to continue issuing building permits on a case by case basis, presumably now taking into account the known environmental sensitivity of the aquifer. All the desired land-uses cited in the Typical Development Proposal section of this paper remain intact. Towns in situations such as this may ultimately have to answer legally and politically to their constituents. The importance and need of a comprehensive karst educational program at local and state levels is apparent and cannot be overemphasized.

Another example of land-use planning on a large raised karst watershed is also situated in upstate New York. A number of private homes were proposed for development upslope of a karst spring used by a village for water supply. The bedrock geology was well documented, with most formations within the watershed including caves. A relatively thin soil-mantle was present, along with several areas of dissolutionally enlarged joints and sinkholes. After many meetings with health officials and politicians, a preliminary decision was made to "protect" the aquifer by imposing a several hundred foot set-back distance from all sinkholes. The set back zone concept incorrectly presupposes that sinkholes are independent from the area-wide karst system (Quinlan et al., 1991). Unfortunately, this decision did not take into account the important fact that any infiltration of septage through the surrounding permeable soil-mantle would probably quickly enter the underlying karst system. Although special considerations may be necessary, development in areas such as this is still possible. The question is really what is reasonable and prudent and within the assimilative capacity of the aquifer.

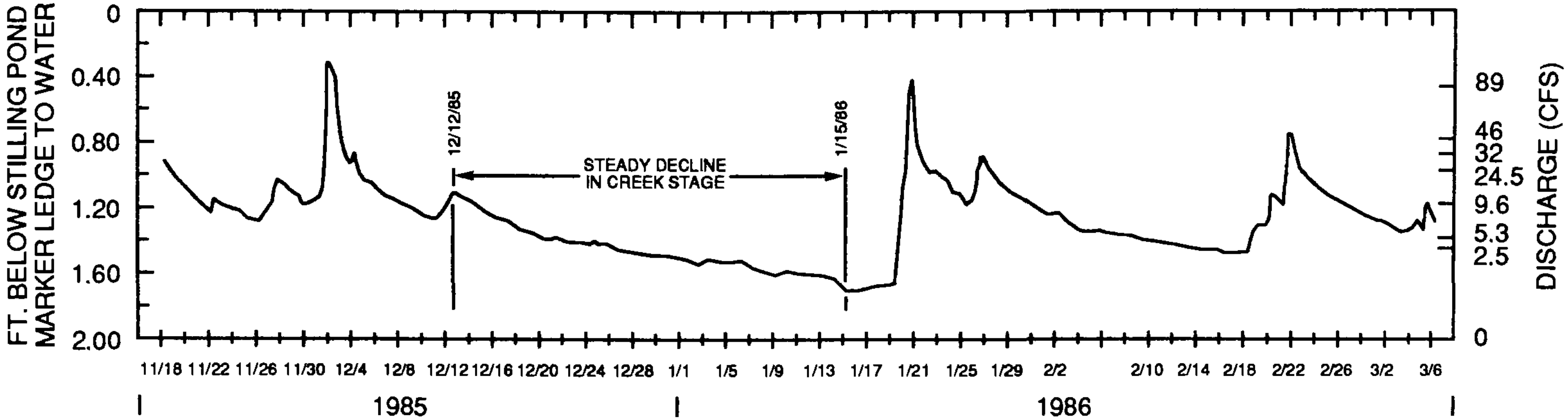

Figure 4: Onesquethaw Creek hydrograph. Stream discharge was measured at 13 different stages. Curvilinear regression was then utilized to establish a series of multi-order equations that were used to correlate stage height with discharge.

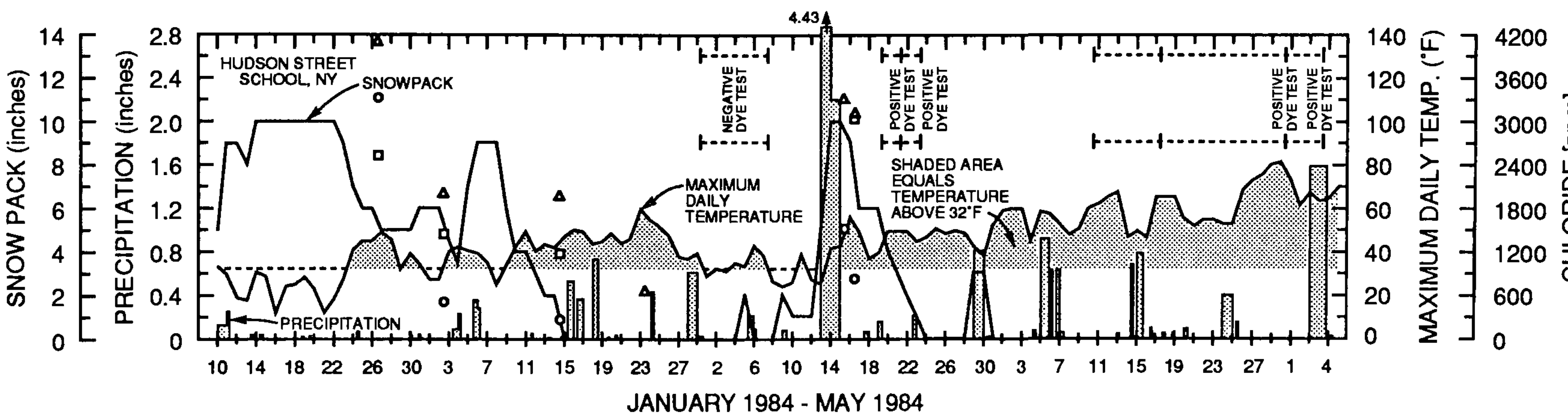

Figure 6: Operable climatic conditions influencing salt contaminant transport from a roadside near Saugerties, New York. Salt infiltrates soil, passes through fractured limestone (against dip), and contaminates wells (Δ, O, and □) and a baselevel spring.

Advanced Testing And Analysis To Define A Karst System

A number of additional steps may be incorporated into a program to define a given karst network. Some may not be necessary if existing or planned land-uses pose little or no threat to an aquifer or its receiving stream. These components may include cave surveying, chemical and hydraulic monitoring, well drilling, geophysics, and tracer tests. Some of these methodologies are discussed below.

Physical Entry Of Karst Networks

The survey and characterization of caves in karst aquifers is valuable because the actual location and extent of segments of the karst system can be examined. There is no substitute for entry and survey of enterable segments of the karst system. Visual reconnaissance of the inner workings of the karst aquifer can provide information of primary transport pathways and can serve to locate monitoring stations. Cave explorers and cave-divers represent one of the best resources when conducting land-use studies in karst terranes. Trained cave surveyors are almost always willing to help towns and federal agencies with this integral aspect of defining karst systems, generally at no charge. Potential access points include cave entrances and springs. Sometimes access can be gained by excavating sinkholes or overflow springs. Large diameter boreholes are good entry points where the conduits are seasonally above the saturated zone. More recently, trained cave divers have penetrated spring resurgences and brought back valuable survey and geologic assessment. Sometimes it may be worthwhile to utilize heavy equipment to dig or drill into blocked portions of a cave system.

Cave Detection With Geophysical Techniques

Geophysical techniques can sometimes be used for identifying conduit locations and connectivity. Micro-gravity and electrical resistivity are two more successful techniques, but no technique is capable of resolving caves of small size at great depth. The work of Kilty and Lange (1991) and Lange and Kilty (1991), however, appears encouraging for locating streams at depth. Geophysical investigation can be followed with coreholes, dye traces, and continuous hydraulic and chemical monitoring of wells and springs.

Water Quality, Hydraulic And Biologic Monitoring In Karst Aquifers

Great emphasis should be placed on learning more about the hydraulics and contaminant distribution, if any, in accessible portions of a karst system. Because a representative chemical characterization of the bedrock flow system can only be obtained from mature conduit segments, storm water should be sampled for contaminant transport after either, 1) karst springs are located, or 2) hydrographs have been obtained from conduit wells, indicating that they are part of the active conduit system.

Water Quality Monitoring Of A Karst System

Select water quality indicators, as well as suspected or planned contaminants should be monitored during periods of low flow and most importantly, during short-term flood events. Monthly or quarterly monitoring is typically conducted in porous and fractured bedrock aquifers; but, the highly transmissive nature and flashy response of some mature karst aquifers requires a more frequent sampling schedule. An infrequent sampling schedule in karst aquifers having rapid flow may miss the short periods of time when chemicals are actively being transported (Quinlan, 1989, 1990). However, different types of contaminant releases often require different sampling schedules. Sampling schedules must be tailored to the geologic and physical setting of contaminant sources, taking into consideration the hydrologic conditions most likely to result in contaminant dispersal, infiltration, and movement. If a contaminant source is an unregulated spill into the soil zone above a karst aquifer, storm flow monitoring may detect it; however, if the contaminant source is a continuous discharge from a point source (e.g., from a package plant) it may well not be detected during flood flow conditions. Traditional monitoring frequencies may thus fail to indicate the true distribution of a contaminant problem in a karstified aquifer. Sampling of water chemistry in monitoring wells placed in the diffuse (non-conduit) flow zone of a karst aquifer will typically yield unreliable data, not representative of the system as a whole (Quinlan, 1989, 1990; Ewers, 1991).

Figure 5 illustrates the water level difference in a physical setting of wells placed only a few meters apart, where substantially different hydraulic conductivity and transmissivity values and chemical concentrations may be obtained. The term "tertiary porosity" has been introduced by Teutsch and Sauter (1991) to refer to dissolutional porosity, rather than secondary fracture porosity. The failure to find contaminants in wells not within the active conduit portions of the karst system may falsely suggest that no contaminant problem is present. It is reasonable to focus contaminant characterization and exit pathway studies on the mature zones of well-karstified aquifers because significant remediation of contaminants found in diffuse-flow zones of karst aquifers is unlikely to impossible (Quinlan and Ray, 1991).

Monitoring must be conducted in locations that intersect the active conduits of a flow system. While wells demonstrated to be in conduits may provide reliable water chemistry data for their sub-watershed areas, the best and most representative sampling locations for a system as a whole are at springs (Quinlan and Ewers, 1985).

An example of the relationship between soluble contaminant movement and assorted climatic conditions in a shallow highly fractured limestone is illustrated in Figure 6. In this example, road salt infiltrates soil, passes through the bedrock (against dip), and contaminates wells and a baselevel spring. Periods of snowmelt, high rainfall, and high temperatures can significantly increase contaminant transport rates. Lack of these conditions can result in low level contamination. Frequent measurements of conductivity would serve as an effective screening tool to select optimal sampling times. Again, the frequency of groundwater sampling in karst terranes must mimic the flow dynamics and the contaminant source type.

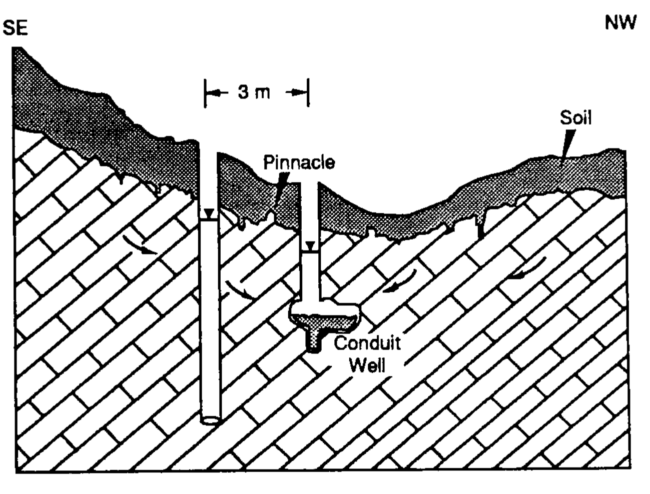

Figure 5: Schematic cross section showing hypothetical low K (hydraulic conductivity) and T (transmissivity) diffuse-flow zone secondary porosity (SE well) and high K and T conduit tertiary porosity (NW well) within carbonate aquifer.

Continuous Chemical Monitoring Of Solution Conduit Waters

Quinlan and Ewers (1985), Quinlan (1989), and Quinlan et al. (1991) have documented practical strategies for monitoring karst aquifers and contaminant transport within them. Basic to these strategies is the understanding that conduit flow may be turbulent and is often quite rapid. An entire contaminant pulse might pass by a monitoring site in a matter of hours, rather than months (see Fig. 7). The rapid response of the conduit system versus that of the diffuse system is a unique feature of some karst aquifers. A sudden influx of water within a conduit system may result in water level rises of tens of feet in a few hours. These sudden rises signal the filling of a groundwater trough, which may bring flood levels up to or above the water table observed in the diffuse flow portion of the karst aquifer. Special efforts are required in karst aquifers in order to detect and characterize episodic or pulse-driven contaminant movement.

Monitoring the relevant water chemistry, water levels, and flow components of a karst system is virtually impossible unless wells have deliberately or accidently intersected active conduits or, as is usually the case, springs have been found by field reconnaissance efforts and shown by tracing to be draining from the source pollutants, and thus to be relevant.

Once relevant springs and conduit wells have been identified, representative sampling of the aquifer can be conducted at times of suspected contaminant transport (i.e., flood pulses). First round sampling might be conducted for major ions, nitrate, ammonia, phosphates, fecal coliform, specific conductance, dissolved oxygen, biochemical oxygen demand, turbidity, pH, temperature, and known or suspected contaminants.

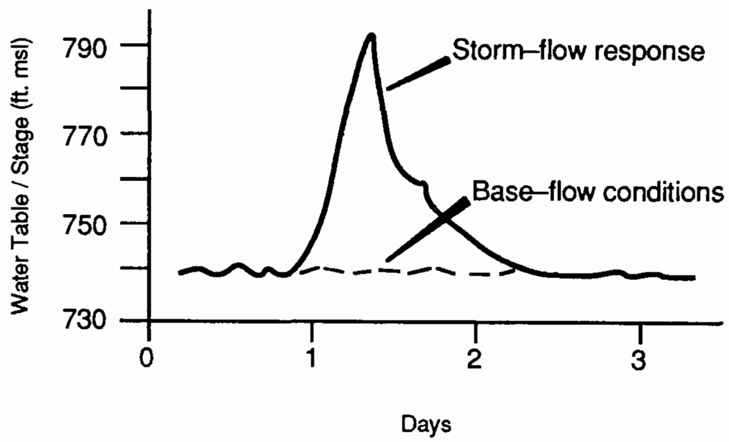

Figure 7: Hypothetical hydrograph of flashy response of a conduit well in a mature karst system.

Hydraulic Monitoring Of Wells Intersecting Solution Cavities

Karst groundwater basins typically contain both diffuse and conduit flow components. Diffuse zone in-feeders, tributary to highly transmissive conduits, would typically exhibit slower water movement until fracture aperture was sufficient to permit groundwater flow to cross from laminar to turbulent flow conditions. Even poorly integrated fractures in the karst system still ultimately drain toward a groundwater trough where transmissivity is relatively higher.

Transducers and data loggers can provide the continuous recording necessary to properly define the hydraulics of wells known to be in the conduit system. Conduit flood pulses may be short-lived (i.e., hours or days), depending on the maturity of the conduit system and the quantity of recharge incident upon the system (Fig. 7). Thus, in order to understand the hydraulics and characterize the water chemistry of the system, it is sometimes necessary to monitor on an hourly rather than monthly or quarterly sampling schedule (Quinlan and Alexander, 1987). In a karst network, long intervals between sample collection are likely to miss storm events and critical contaminant transport entirely. Frequent monitoring and sampling during flood pulses is necessary.

Wells which have large fluctuations of their piezometric surface, especially when followed by a short recession curve, may be interpreted as being part of an active and mature karst flow system (Fig. 7). Water flowing through these wells, and their associated dissolution conduits ultimately must discharge to some baselevel.

Karst Groundwater Basin Flow Components

Continuous monitoring and quantification of discharge entering and leaving a karst basin is important for contaminant characterization and some land-use studies. For larger projects, where contamination is or may be a problem, this may include stream, spring, and conduit well discharge monitoring via weirs and stage recorders (or other devices). Seepage velocity (discharge) measurements on tributaries suspected of being losing streams should also be considered. One or more weather stations are critical for realistic water budget appraisal. For small projects, it may be sufficient to monitor only the base flow discharge leaving the karst system during drought or low-return periods.

Knowledge of expected flood-return intervals and their magnitude can be important in assessing contaminant transport potential. One example of where flood-return information might be applied is in predicting mobilization probabilities of contaminants poorly bound to soils; which may occur as a result of soil-flushing stemming from large precipitation events. In other settings, large point source quantities of hazardous waste may be released from elevated flow system segments only during extreme flood conditions. This technique, which might be applied towards roughly assessing a maximum rate of contaminant movement in a karst system, may be applicable in karst areas where all or most of the drainage exits via springs or where sufficient in-conduit hydrograph data is available. Where time is an element for a particular land-use option being contemplated, and reasonable but limited stream discharge data is available, it is possible to perform a flood probability analysis. Similarly, knowledge of base flow and drought-return intervals, when the assimilative capacity of an aquifer is at its lowest, can be critical to prudent land-use determinations.

An example of such an analysis was conducted for the Mill Pond karst basin in east-central New York (Rubin, 1991a, 1991b) where much of the basin's flow is subsurface for most of the year. The limited data available for statistical comparison among hydrologic years necessitated examination of another roughly comparable basin in order to assess flood-return intervals. The farthest headwater gaging station on Schoharie Creek at Prattsville was selected. Many inherent differences occur between the basins, notably elevation, geology, soil thickness, size, and location. The Prattsville and Onesquethaw Creek gaging stations are approximately 48 kilometers apart. However, the Prattsville and Mill Pond watersheds are comparable under conditions of a saturated soil-moisture bank, high runoff, and similar storm systems. Eighty-two years of data at the Prattsville station were examined.

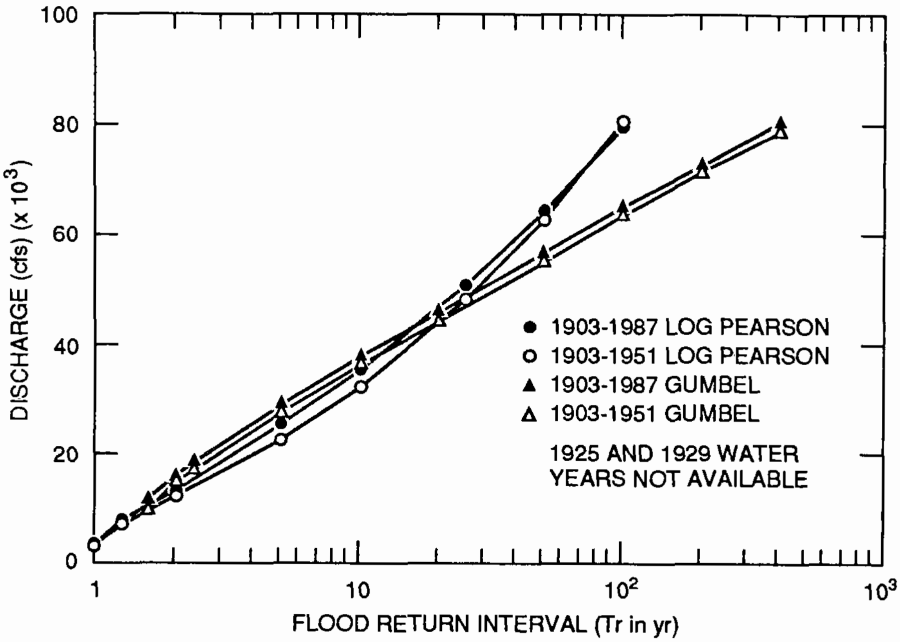

Figure 8: Schoharie Creek at Prattsville, New York. Log-Pearson type III and Gumbel distributions utilizing historic water year peak flow data. Flood return interval for largest Onesquethaw Creek peak discharge of record correlated to Prattsville March 15, 1986 flood of 54,900 cfs. Range of two methods shows a Tr of 30 to 47 years. Similar comparisons may be made for low flows in geologically comparable basins. Analyses of this type can be useful tools in predicting flow velocities, contaminant arrival times, and contaminant dilutions.

A Log-Pearson Type III and Gumbel-distribution statistical comparison (Linsley et al., 1975) of historic peak flow of Schoharie Creek gaging data with this study's hydrograph information for Onesquethaw Creek indicates that the largest Onesquethaw Creek peak of record (March 15, 1986) has a return interval on the order of 30 to 47 years (Fig. 8). This corresponds to a Prattsville hydologic-year peak discharge of 54,900 cfs. Thus, if 40 years was the expected flood return interval, 25 floods of this magnitude could be expected every 1000 years. These infrequent storm or runoff events reasonably represent a near-maximum quantity of water available in the watershed under ideal, thin-soil-mantled, rapid infiltration conditions. Knowledge of peak or base flow return periods can sometimes be correlated with water chemistry results to help assess chemical loading both in the karst system and to stream receptors.

Biologic Monitoring Of The Karst System

Fauna, both within caves and at springs, provide the best indicator of health of the karst system. Various organisms, such as benthic invertebrates, benthic macroinvertebrates, fish, amphipods, isopods, flatworms, and copepods can provide excellent indices of biological integrity. These organisms may be subjected to episodic contaminant pulses and any background contaminant chemistry. Their health, or sometimes absence, as monitored at one or more points in time, is a true measure of the health of the groundwater system (Poulson, 1991; Preddice, T., NYS Dept. of Environmental Conservation, pers. comm.). Acute exposure to various chemicals can influence the ability of some species to reproduce or survive. This can have far-reaching effects on the food chain. The assimilative capacity of karst waters and its biota for contaminants is dependent upon factors such as chemical type and toxicity, chemical injection or leakage rates, discharge in the system (e.g., low flow), and reaeration rates along the flow path. Similarly, plant species can also aid in the health-based assessment of a receptor stream.

Tracer Studies

Tracer studies are the best technique for determining flow direction, destination, and velocity within a karst groundwater basin (Quinlan, et al., 1991). State-of-the-art tracing techniques are capable of simultaneously running multiple tracers to assess different suspected flow routes, or to verify low quantitative results from previous traces. Tracer tests may be successfully used to: 1) determine discharge locations of sinking streams which may flow within shallow or deep karst systems; 2) determine aquifer discharge locations necessary for characterizing contaminant exit pathways; 3) determine groundwater seepage velocity rates and discharge in conduit portions of the karst flow systems. (This is important for water budget and model assessment, as well as preventative planning for episodic contaminant or spill releases); 4) define drainage divides and catchment basin boundaries; 5) define subsurface flow and transport routes within conduit and diffuse zones; 6) assess stream infiltration/piracy to shallow surficial discharge points; and 7) realistically characterize groundwater seepage velocity rates in the diffuse zone between boreholes.

Wells

Wells which are part of the active conduit system, as determined by transducer/hydrograph response, or their locations in known cave passages, may provide the best monitoring locations when springs cannot be located (e.g., as a result of flooding). Wells have value for 1) lithologic characterization, 2) structural characterization, 3) piezometric monitoring, 4) chemical monitoring, and 5) tracer studies.

Areas where an unusually high percentage of boreholes have encountered voids (e.g., Oak Ridge, Tennessee) are indicative of maturely karstified terranes. Such settings are uncommon. Quinlan and Ewers (1985) and Ewers (1991) have estimated the odds of encountering a dissolutional conduit in a karst aquifer by drilling a well at about 1:2600.

Political Realities Of The Land Use Permitting Process

Responsible land-use planning requires a willingness of all parties involved in the planning and permitting process to look beyond the physical boundaries of a particular building proposal which may be in front of them. This may ultimately mean that the scope and scale of development needs to be reduced. This, in turn, may make landowners unhappy that they can no longer subdivide or use their land as they might wish. Unhappy landowners do not reelect their politicians, who are thus hesitant to spend dollars on consultants. Some consultants, who desire continued work with towns, may wish to insure their client's approval. This process may be further complicated by town planning board members who are themselves, or have relatives and friends who are, developers. A further complication in some states may result when lead agency/permitting status is granted to towns which may have vested interests. In instances such as this, a cooperative agreement for technical exchange between knowledgeable state and government people and planning boards might be beneficial to all parties involved. Thus, it should be recognized that prudent land-use planning may, in some instances, be fraught with political conflict and pressures. All to often, the local residents and the environment loses.

Sound land-use planning is best conducted in the absence of specific building proposals. It is important to recognize impending problems before building permits are on the table. Ideally, land-use planners in karst terranes will be able to conceptualize the subsurface flow systems present in their basins and incorporate geologic and hydrologic information into a zoning master plan. Such master plans require 1) planners with some scientific background, 2) sufficient capital and expertise to characterize a karst basin, and 3) an educational process, such as that now being undertaken by the National Speleological Society (NSS) and the American Cave Conservation Association (ACCA) in the United States. Public hearings and board meetings, which provide a forum for the educational process, can be used to good advantage with receptive audiences. Members of these organizations and karst hydrologists are often willing to help in this process when requested. More emphasis also needs to be placed on educating state, federal and local officials.

Whereas development within karst basins has historically occurred on an individual application basis, it may now be prudent to initiate a more broad-based master planning process. Planning should take into account the likely contaminant loading into the karst system, and a reasonable measure of its assimilative capacity. In many respects, the strategy employed should mimic the discharge requirements many states impose on industrial effluents to surface waterways. Instead of granting discharge permits with unlimited contaminant ceilings, permits are only granted when individual contaminant contributions do not exceed the assimilative capacity of collective discharges from multiple facilities. In this manner, a strategy is employed to insure the continued health of the waterway. In karst groundwater basins, a discharge permit of this nature should be based on the assimilative capacity of the receiving aquifer and stream at times of low flow.

Who To Look To For Help

Characterization of the hydrology of a karst basin is beyond the technical expertise of most land-use planners, environmental consultants, or even experts within State and Federal agencies. The flow dynamics in karst terranes are very different from those in porous or fractured media. Significant scientific advances in karst hydrology have occurred in the last two decades. Unfortunately, many of the technical papers have not been published in mainstream journals or received widespread distribution. This poor dissemination of knowledge in karst hydrology has hindered the land-use planning process. More recently, this trend has been broken (Ford and Ewers, 1978; Quinlan and Ewers, 1985; White, 1988; Ford and Williams, 1989; Quinlan, 1989, 1990; Palmer, 1991; and Quinlan et al. 1991).

Plainly stated, extremely few consulting firms, universities, and government agencies with experienced hydrogeologists, geologists and engineers possess the specialized qualifications of karst hydrologists. Finding an experienced consultant knowledgeable about karst hydrology is difficult. Specialists in karst hydrology may be contacted through two organizations:

Karst Waters Institute
c/o John Mylroie; President
Box 2194
Mississippi State Univ., MS 39762
601-325-8774

National Ground Water Association
6375 Riverside Drive
Dublin, Ohio 43017
614-761-1711

The wise consultant will use the unique talents and knowledge of state cave surveys and cave explorers, who are often associated with local chapters of the National Speleological Society (NSS). Such individuals can be a tremendous asset in the karst inventory evaluation process, as well as valuable sources of cave maps depicting segments of karst aquifers. The NSS may be contacted at Cave Ave., Huntsville, Alabama 35810 (205-852-1300). It is important to verify that the firm or individual claiming expertise in karst hydrology actually has it.

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Biographical Sketch

Paul Rubin is a Research Scientist in the Environmental Sciences Division of the Oak Ridge National Laboratory in Tennessee where he is involved in characterizing groundwater flow. Mr. Rubin served as a hydrogeologist for the Environmental Protection Bureau of the NYS Attorney General's Office for 8.5 years prior to his present position. He received his Master's degree in Geology from SUNY New Paltz. He has taken an active role in research and contaminant investigations, as well as land-use planning issues, in karst terranes.

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Land Use Planning And Watershed Protection In Karst Terranes

Paul A. Rubin

1. You mentioned chlorinating septic tank outflow. While this should take care of fecal coliforms, I don't see any impact of chlorination on the problem of affecting nutrient level and nutrient type in cave ecosystems. How would you deal with the elevated organic carbon and phosphate levels that are produced by septic tanks?

   Chlorinating septic waste, as is currently being conducted by Stewart's in the Mill Pond groundwater basin, will reduce viral and bacteriological problems, but not eliminate the problems associated with elevated nitrogen and phosphorus levels. Thus, practical resolution of the problem of excessive nutrient loading and eutrophication lies in 1) sufficient dilution of waste accompanied by adequate stream residence time, temperature, and reaeration; 2) groundwater basin characterization leading to responsible land-use planning; and 3) groundwater basin management (e.g., no sinkhole dumping, removal of existing pollution point-sources); and/or 4) waste collection and treatment prior to discharge. Cave ecosystems are particularly vulnerable to contaminant inputs since they are often in the zone of degradation immediately downstream of pollution sources. Elevated organic carbon and phosphate levels, as well as fecal coliform, will initially lead to a reduction in dissolved oxygen used in satisfying the biochemical oxygen demand (BOD) in streams. Farther downstream of the contaminant source, bacteria and fungi thrive on the decomposition of organics, thus increasing ammonia nitrogen and decreasing the BOD (Hammer, 1975). The resulting increase in inorganic nutrients downstream of the zones of degradation and active decomposition can result in an unnaturally nutrient-rich environment. An increase in inorganic compounds, such as nitrate, phosphates, and dissolved salts is common prior to the clear water zone (Hammer, 1975). In severely contaminated streams, the clear water zone may be far downstream of areas degraded by contaminants.

2. In light of your statements regarding the importance of evaluating carbonate hydrology by using "karst" thinking and techniques, are the methods you discussed being used to evaluate environmental problems on the Oak Ridge Reservation? If not, should they?

   It is recognized that karst dissolution conduits may be a significant factor in groundwater flow in portions of the Oak Ridge Reservation and surrounding area. Figure 2 and the Clarksville example discussed in the text, are illustrative of the type of "karst" thinking and techniques required to evaluate environmental problems on the Oak Ridge Reservation. However, field work and characterization remains to be conducted on the carbonate flow systems at the three Oak Ridge Reservation plant sites and their catchment basins. While some isolated tracer studies have been conducted, a more detailed and systematic approach to defining the nature and extent of the mature karst systems present is necessary. In one such example, the author and groundwater program managers at the Reservation are currently spearheading a groundwater characterization program which emphasizes the karst flow dynamics proximal to the K-25 plant site. The methods which will be employed are consistent with the methods discussed in this paper, and will employ sophisticated characterization techniques, including geophysics, continuous monitoring of hydrographic response in conduit wells, and tracer tests.