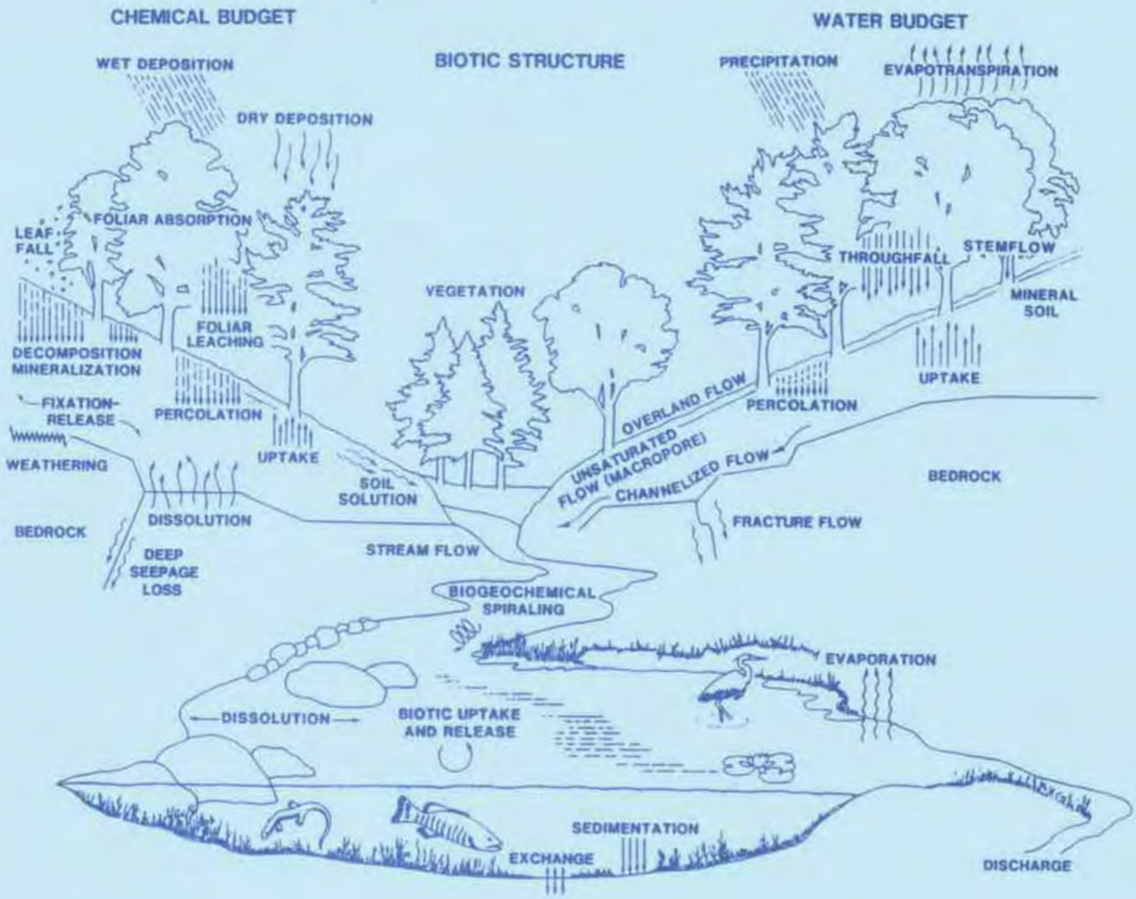
EXTENDED ABSTRACTS FROM

# THIRD TENNESSEE WATER RESOURCES SYMPOSIUM

August 7-9, 1990

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U.S. Geological Survey,
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Oak Ridge National Laboratory
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U.S. Army Corps of Engineers, Nashville District
U.S. Army Corps of Engineers, Memphis District
The University of Tennessee

In cooperation with:

Tennessee Section of the American Water Resources Association

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Edited by
Michael J. Sale
and
Patricia M. Presley

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Nashville, Tennessee
1990

## Contents

## POSTER SESSIONS

## PREFACE

The first and second Tennessee Hydrology Symposiums were hosted by the Water Resources Division of the U.S. Geological Survey under the able guidance of Ferdinand Quinones. They were very successful in bringing many of Tennessee's water people together. At the Second Symposium, the Tennessee Section of the American Water Resources Association (AWRA) was formed. Ferdinand Quinones immediately moved that one of the major functions of this new organization would be to act as the host for future symposia. The motion was readily seconded and passed and the new officers had important work to do.

The first president of the Tennessee Section, AWRA, Michael C. Yurewicz, got us started on the road to this year's symposium and soon we had a large and effective steering committee leading the way. One of this committees first act was to expand the umbrella for the symposium and call it the Third Tennessee Water Resources Symposium. This, we felt, was more in keeping with the interdisciplinary organizational theme of this meeting. We all hope that we have not offended those of you who would rather have maintained the hydrology emphasis.

The purpose of this annual meeting is to provide a forum for communication among diverse groups within the State of Tennessee on water resource management and research topics. Our goal is to promote mutual cooperation among scientists, engineers, and researchers at the federal, state, and private levels.

The importance of the water resources in Tennessee has prompted research activities by universities, county, state, and federal agencies. The Water Resources Research Institute at The University of Tennessee-Knoxville (UT-K) and the Center for the Management, Utilization and Protection of Water Resources at Tennessee Tech in Cookeville conduct significant investigations. Research is also taking place at Austin Peay, Memphis State University, UT-Martin, and UT-Chattanooga. Among the state agencies, the Office of Water Management of the Tennessee Department of Health and Environment, the Tennessee Wildlife Resources Agency, the Tennessee Department of Transportation, and the Tennessee Department of Conservation conduct research activities as part of their regulatory and management programs. In the federal sector, the U.S. Geological Survey, the Tennessee Valley Authority, the U.S. Army Corps of Engineers, and the U.S. Department of Energy's Oak Ridge National Laboratory are involved in complex and extensive water research programs. All of these organizations, as well as industry and private consultants, will be represented at the Annual Tennessee Water Resources Symposium.

The extended abstracts from this year's Symposium are published and distributed with the intent to further water resources communication. Please look at the abstracts and call or write to the presenters. We have included a full mailing address and telephone number for the senior author of each paper as a footnote to each abstract.

While at the Third Annual Tennessee Water Resources Symposium, renew your membership in the Tennessee Section of the American Water Resources Association. Or, if you did not join last year – do it now. You should also consider joining the national AWRA group and attend some of its conferences and symposia. You will find AWRA a most rewarding organization.

We hope that you find this symposia enjoyable and rewarding. Your comments on it are welcome.

John A. Gordon and
Larry M. Richardson
Symposium CoChairman

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# AN OVERVIEW OF THE EPA UNDERGROUND INJECTION CONTROL PROGRAM IN TENNESSEE

William R. Mann[1]
U.S. Environmental Protection Agency
Atlanta, Georgia

The U.S. Environmental Protection Agency (EPA) Region IV is responsible for administering the Underground Injection Control (UIC) Program of the Safe Drinking Water Act in the State of Tennessee. Five classes of injection wells are regulated by the UIC program. Well classification is based on the type of fluid injected and the relationship between the injection zone and the lowermost underground source of drinking water (USDW). Injection classes can be summarized as follows:

Class I: Wells used to inject hazardous wastes or dispose of non-hazardous industrial waste and treated municipal sewage below the deepest underground source of drinking water.

Class II: Wells used to inject fluids associated with the production of oil and gas or fluids and compounds used for enhanced hydrocarbon recovery. These wells normally inject below the deepest underground source of drinking water except in cases where the USDW contains producible quantities of oil or gas.

Class III: Wells that inject fluids used in subsurface mining of minerals.

Class IV: Wells which dispose of hazardous or radioactive wastes into or above an underground source of drinking water. These wells are banned.

Class V: Wells not included in the other classes that normally inject non-hazardous fluid into or above an underground source of drinking water. These wells are commonly referred to as shallow injection wells. Examples of these wells include agricultural drainage wells, improved sinkholes, stormwater drainage wells and automobile service station disposal wells.

Past program emphasis has focused on oil and gas operators and large chemical companies who use deep injection wells to dispose of their waste and coal companies who inject their coal preparation plant waste into abandoned underground mine works. Several permits have been issued for brine disposal wells associated with oil production in Morgan and Fentress Counties and Region IV is currently reviewing several permit applications from chemical companies and coal operators in Humphreys, Maury and Clairborne Counties. Current and future

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[1]United States Environmental Protection Agency, Region IV, 345 Courtland Street, N.E., Atlanta, Georgia 30365 (404/347-3866).

UIC program emphasis has been expanded and will also include shallow injection wells along with the Class I-IV wells.

Class V wells comprise the majority of the injection wells in Tennessee and pose the greatest threat to the groundwater. These wells do not lend themselves to conventional controls and has led the Agency to adapt a Shallow Injection Well Strategy to develop ideas on how to best manage these wells and their possible impact on groundwater. Program personnel from Region IV have met with various Tennessee state agencies, universities and environmental groups. Questionnaires about shallow injection wells have been sent to selected industry groups in several counties and EPA inspectors have conducted several inspection trips looking for possible injection violations. The Agency has funded a Class V research project at Tennessee Technological University from the Shallow Injection Well Initiatives Program and an information mailer has been developed about shallow wells for circulation to automotive repair trade groups. This high level of interest the Agency has in shallow injection wells can be expected to continue as more is learned about the effect shallow injection wells have on the groundwater.

## STRATIGRAPHY/LITHOLOGY

Groundwater systems are influenced by stratigraphy and lithology. Recognition of detailed lithologic variations improves our ability to understand the three dimensional variations in groundwater regimes in different parts of the valley. For instance, the hydrology of the Knox Group is basically different from that of other units of the ORR. Volumes of stored and discharged water and depths of groundwater movement in the Knox are significantly greater. Water levels in places in the Knox are at depths of more than 150 ft, and hydraulic gradients are notably shallower than in other units. Lee and Ketelle (1987) related water movement in the Knox Group, at depths from 200 feet to 400 feet or greater, to stratigraphic control by a thin facies in the Copper Ridge Dolomite. Different units in the Knox influence groundwater flow by being more or less soluble or by variations in susceptibility to fracturing. While the Maynardville Limestone is the uppermost unit of the Conasauga Group, it is hydrologically more like the overlying Knox Group, and very unlike the other much more clastic units of the Conasauga.

Large contrasts in the vertical profile of hydraulic conductivity, particularly in the Conasauga and Chickamauga Groups, distinguish the stormflow zone from other zones (Moore, 1988, 1989), and severely limit the volume of recharge to and thus discharge from the underlying saturated flow system. Further study is required to confirm Moore's estimate of less than 3 in/yr of flow through the saturated-flow zones in the Conasauga and Chickamauga.

Matrix porosity, while insignificant in groundwater movement in bedrock units, plays a major role in groundwater quality. Stratigraphy plays an important role in controlling the relative abundance of fractures and in the complex relationship in dual-porosity systems between fracture permeability and matrix porosity.

## STRUCTURE

Groundwater systems are influenced by geologic structures (Fig. 1). Fracture systems in the ORR formed at different times related to different tectonic or unloading events. Some fracture sets are regional, whereas others occur only near faults or are related to folds. Strike-parallel fractures intersecting bedding influence the dominant movement pattern of groundwater along valleys. Secondary control occurs along lithologic boundaries. More intense fracturing and cleavage development occurs near faults. Groundwater moves through connected, open fractures irrespective of when or how they formed. Recent evidence (Haase and King, 1990) indicates that fractures in Bear Creek Valley may be open to depths of 500 ft. Carbonate solution features recently have been intersected by drill holes in the vicinity of a major fault; the present depth limit and areal variety of fault-related solution feature development remain unknown.

Faults are present at all scales, from microscopic to regional. Fault zones, especially if gouge is present, can act as an impermeable boundary (aquiclude) between two groundwater systems. Surrounding the larger-displacement faults, however, is a zone of fractures overprinted on preexisting fracture sets, which may create a zone of increased permeability. The lateral extent and permeability of such zones may be assessed by combining field and laboratory studies. Both the thickness of a gouge zone and the extent of fracturing are partly controlled by the amount of displacement, the depth where faulting occurs (confining pressure), and rock type. As a consequence, not only is fault orientation an influence on groundwater flow paths, but also the rock types juxtaposed by faulting, minor structures, and the structural style of the fault at the surface and at depth. In the ORR both regional thrust faults, the Copper Creek and Whiteoak Mountain, have nearly the same orientation and large displacement, but differ with respect to the rock types in the hanging wall and footwall, the development of minor faults and folds, and their relative position in a thrust system. In addition, faults of smaller displacement occur within all rock units.

The Copper Creek fault places the clastic sedimentary rocks of the Rome Formation on top of the carbonate-rich rocks of the Chickamauga Group. The fault is a bedding thrust that juxtaposes the same rock types at depth throughout the ORR. In contrast, the Whiteoak Mountain fault is associated with a wider zone of faulting that places the Rome Formation in the hanging wall against rocks as young as Mississippian, depending on location. The variety of footwall rock types is partly related to development of the East Fork Ridge syncline in the footwall.

Minor faults, characterized by small displacement and limited areal extent, may affect local groundwater flow paths. The rock types across minor fault planes are generally of the same group of lithologies, reducing the lithologic contrasts created by the regional faults.

Fractures may have displacement only perpendicular to the face (joints), or some displacement parallel to the fracture plane (faults), or both. Most fractures at the surface tend to be open, rather than mineral-filled. Core analysis indicates that many fractures open at the surface are mineral filled (mostly with carbonate minerals) at depth, suggesting that near-surface weathering removes the filling.

Joint orientations form systematic sets consistent throughout the range of scale. Nonsystematic fractures also occur here and influence groundwater flow. We believe that the systematic joints are a primary control of groundwater flow direction. Throughout the ORR, the dominant systematic joint sets strike N40-50W and N50-60E, the former dipping nearly vertically and the latter dipping moderately northwest. Joint patterns closer to major and minor faults and folds conform to the regional orientation, but also show a wider range of orientations. This is especially true for the Rome Formation adjacent to the major faults and of the Conasauga Group, which contains numerous minor faults and folds. While the aperture of a joint affects fluid flow, so might the linear gap created at the intersection of two or more open joint sets. The intersection of joint sets that are nearly perpendicular to bedding with bedding plane fractures can lead to the development of channelways parallel to bedding strike (northeast striking joints intersecting bedding plane joints), down the dip of bedding (northwest striking joints that intersect bedding plane joints), and perpendicular to bedding (intersection of northeast and northwest striking joints). Therefore, predictable groundwater flow paths may correspond with the more consistent and well-developed intersecting joint sets.

Gouge-filled and slickensided fractures (faults), commonly concentrated near major fault zones, restrict fluid pathways. Mineral fillings in fractures (veins), more common in calcareous lithologies, can effectively seal off fractures. Sediment filling of cavities in carbonates also restricts flow. Open fractures occur in all lithologies to provide a flow path, but fracture density is strongly controlled by rock type and bed thickness.

The depth to which fractures remain open is controlled by fluid pressure, by the present-day rock stress orientation and magnitude, and by the geochemical environment. Knowledge of the stress field can be combined with the different fracture orientations to suggest which fractures are under the greatest compressive stress perpendicular to their walls and those under the least, indicating possible ranges in closing depth, and thus limits on flow depth, for the different fracture sets.

Since most fractures form at depth, confining pressure enhances the relative ductility of different rock types. A sandstone at 3 km depth fails by fracturing, whereas a limestone deforms internally to accommodate the stress. This suggests that deformation of an interbedded sequence of various rock types at approximately the same depth can lead to fracturing of some units and not others. Therefore it is possible that particular lithologic units are more fractured than others, given the same deformational history, and consequently are more permeable.

The direct relationship between fracture spacing and bed thickness is consistently observed in outcrop and drill core: as bed thickness decreases, so does the spacing between fractures in a particular set. Theoretically, a fracture forms to relieve strain and in doing so influences the surrounding stress field to a distance equal to the thickness of the bed, so a new fracture can form at a bed thickness or more away from the old fracture. In general, this result holds qualitatively, but the influence of preexisting fracture sets can affect the spacing.

## CONCLUSIONS

1. Standard approaches to hydrogeologic characterization of the groundwater regime should be revised to include combinations of structure and rock type that influence flow-path configurations, flow rates, and contaminant concentrations.

2. Groundwater systems are stratigraphically and lithologically influenced. Matrix porosity, while insignificant in groundwater movement in bedrock units, plays a major role in groundwater quality. Stratigraphy plays an important role in controlling the relative abundance of fractures and in the complex relationship in dual-porosity systems between fracture permeability and matrix porosity.

3. Strike-parallel fractures intersecting bedding influence the dominant movement pattern of groundwater along valleys. Secondary control occurs along lithologic boundaries. More intense fracturing and cleavage development occurs near faults. Fault orientation is an influence on

groundwater flow paths, but also the rock types juxtaposed by faulting, minor structures, and the structural style of the fault at the surface and at depth.

4. The systematic nature of the joints is a primary control of groundwater flow direction. The aperture formed at the intersection of joint sets that are nearly perpendicular to bedding plane fractures can lead to the development of channelways parallel to bedding strike, down the dip of bedding, and perpendicular to bedding.

## REFERENCES


Haase, C. S. and H. L. King, 1990. Report and Preliminary Assessment of the Occurrence of Dense, Nonaqueous Phase Liquids in the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Oak Ridge Y-12 Plant. Y/TS-629. Oak Ridge Y-12 Plant, Oak Ridge, Tennessee.

Lee, R. R. and R. H. Ketelle, 1987. Stratigraphic Influence on Deep Groundwater Flow in the Knox Group Copper Ridge Dolomite on the West Chestnut Ridge Site. Oak Ridge National Laboratory, ORNL/TM-10479. Oak Ridge, Tennessee.

Moore, G. K., 1988. Concepts of Groundwater Occurrence and Flow near Oak Ridge National Laboratory, Tennessee. Oak Ridge National Laboratory, ORNL/TM-10969. Oak Ridge, Tennessee.

Moore, G. K., 1989. Groundwater and Flow Systems near Oak Ridge National Laboratory. Oak Ridge National Laboratory, ORNL/TM-11368. Oak Ridge, Tennessee.

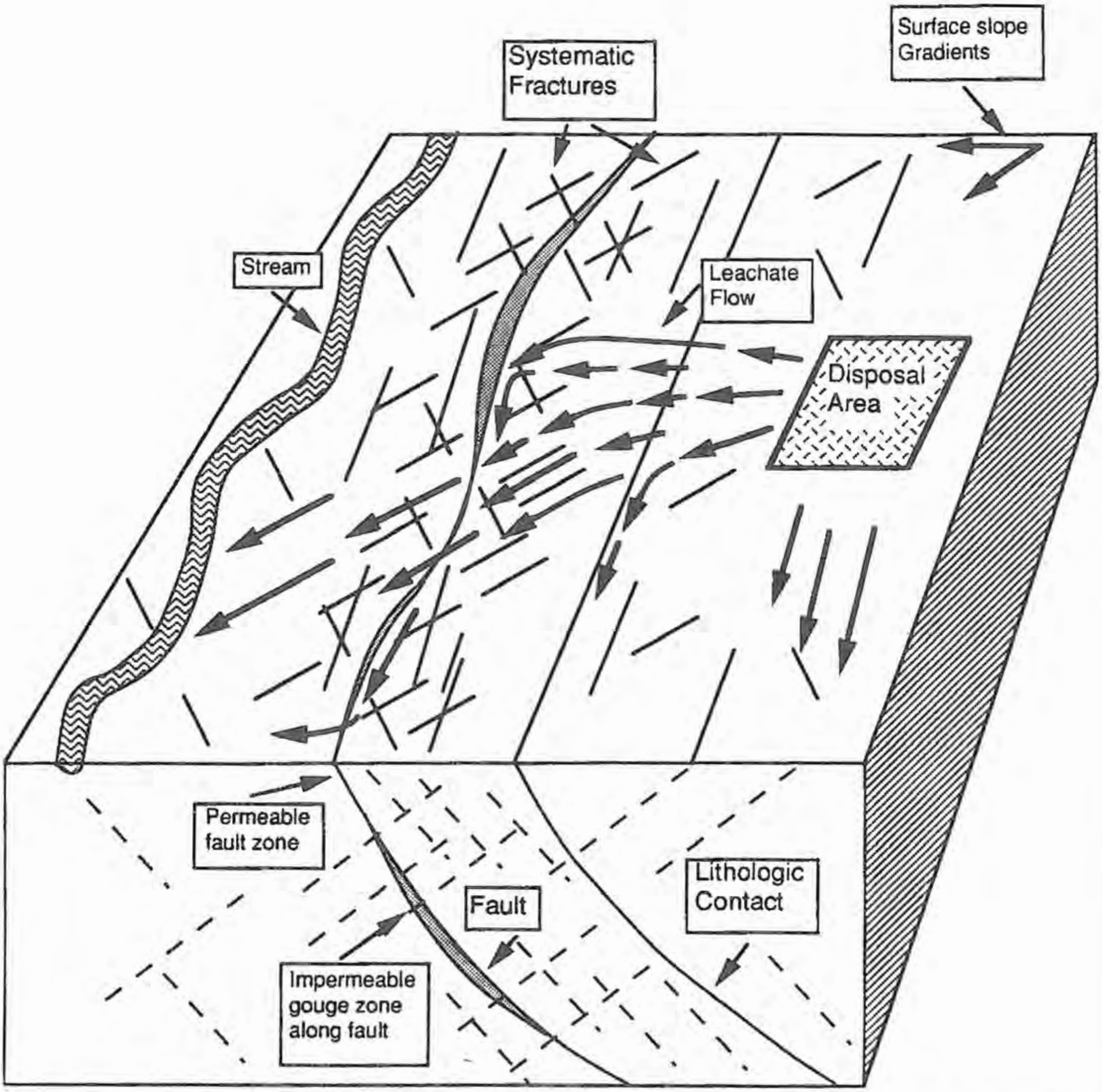


Figure 1. A fluid moving from either a point source - like a waste-disposal facility - or a non-point source will move down existing topographic/hydrologic gradients and enter the groundwater system in the fractured reservoir environment. Groundwater flow will be influenced by systematic and nonsystematic fractures as well as by lithologic and fault contacts. Character of material within fault zones and fracture density near faults controls permeability of faults.

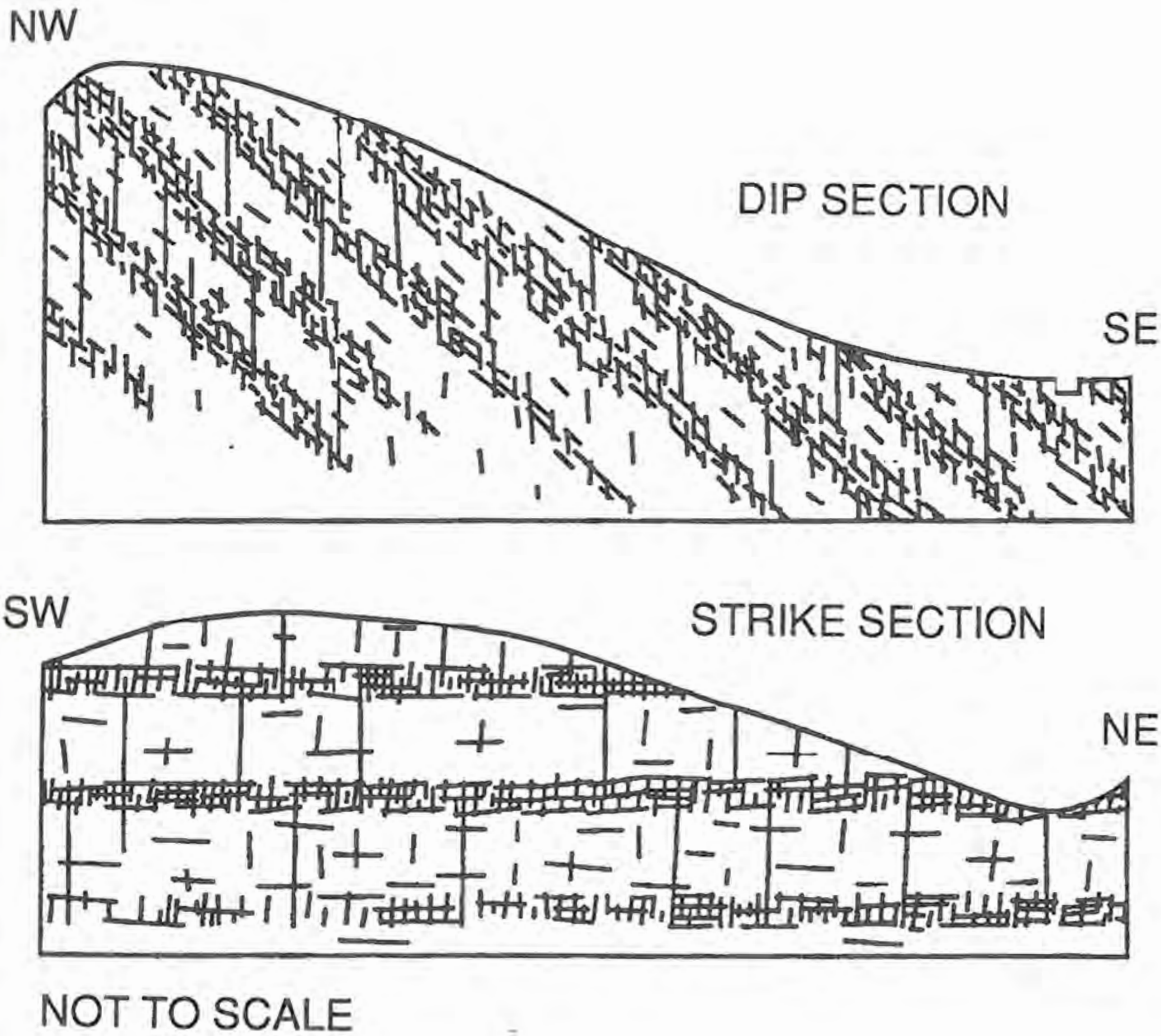


Figure 2. As illustrated in the hypothetical profiles of a set of fracture zones, if bedding-plane fractures control the trends of water-producing zones, along-strike flow paths occur within these zones (upper profile), whereas cross-strike flow paths occur partly within poorly permeable (less fractured) rock (lower profile). From Moore, 1989.

# BIODEGRADATION OF MICROPOLLUTANTS BOUND TO DISSOLVED ORGANIC MATTER

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## INTRODUCTION

Information concerning the movement and fate of groundwater pollutants is necessary to predict future problems at well sites and plan for aquifer remediation. Recently, the influence of naturally occurring dissolved organic matter (DOM) on the transport and fate of organic contaminants has been demonstrated. However, the interaction between DOM and organic pollutants, particularly with regard to microbial availability, has not been fully evaluated. The purpose of this study was to determine the equilibrium distribution and behavior of a model chlorinated-aromatic compound [2,4,6-trichlorophenol (TCP)] sorbed to dissolved humic acid (HA); particularly its availability for microbial degradation.

## METHODS

Batch microcosms, consisting of 250 ml screw top Erlenmeyer flasks, were dosed with various concentrations of humic acid, 2,4,6-TCP, and acclimated bacteria. Humic acid was used as the dissolved organic phase to sorb the target pollutant. This material was extracted from soil using a procedure outline by Schnitzer and Khan[(1)]. The working humic acid solution was diluted with phosphate buffer to obtain the desired final concentration (0-100 mg/l) in each microcosm.

Radiolabelled 2,4,6-TCP was spiked into each microcosm, the flask sealed with a Teflon/silicon septum inside a bakelite cap, and mixed on a shaker table at 100 cycles/min. In addition, the same quantity of labelled 2,4,6-TCP was added directly to scintillation vials for direct counting to determine initial substrate concentrations. Initial 2,4,6-TCP concentrations were 0-1000 $\mu$g/l. Flasks were sampled over time until equilibrium between free and HA-sorbed 2,4,6-TCP was determined. Samples were centrifuged and the liquid fraction decanted. The aqueous 2,4,6-TCP was fractionated by gel chromatography into free and HA-

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sorbed aliquots and analyzed. Controls were also analyzed in the same manner as samples and all samples were analyzed in duplicate.

Biodegradation was determined after addition of acclimated bacteria to equilibrated microcosms. A bacteria culture capable of utilizing 2,4,6-TCP as the sole external carbon and energy source (*Pseudomonas aeruginosa*) was isolated from subsurface soil. Diluted microbial solutions were prepared from the batch microbial culture. Microcosms were incubated at room temperature (20°C) on a shaker table at 100 cycles/min.

Biodegradation rates were determined by measuring $^{14}CO_2$ and $^{14}C$ remaining in solution. $^{14}CO_2$ gas produced during incubation was trapped in a center well suspended in the head space. Each center well contained 100 µl of methylbenzylamine, and organic base, to trap $^{14}CO_2$. Incubation of respiration flasks was terminated by injection of $H_2SO_4$ to a pH of 2, into the microcosm. After acidification, the microcosms were shaken an additional 24 hours to trap all $^{14}CO_2$ as $^{14}CO_3^{2}$. Each center well containing the organic base was detached from the septum and placed directly into 10 ml of scintillation cocktail and assayed for radioactivity to determine mineralization of labelled substrate. A series of controls using $NaH^{14}CO_3$ were processed simultaneously to determine the efficiency of this method for trapping $^{14}CO_2$. Trapping efficiency was determined to be 87% (n = 14).

## RESULTS

### Gel Chromatography

Gel Chromatography (size-exclusion chromatography) was used to separate free 2,4,6-TCP from HA-bound 2,4,6-TCP by molecular size. Small molecules, such as 2,4,6-TCP, follow a more torturous route through the column and, therefore, have long retention times. Large molecules, such as humic acid, are excluded from internal pore spaces in the gel and, therefore, pass more quickly through the column.

Figure 1A is a typical chromatograph for a humic acid solution passed through the gel column. Eluent from the column was collected over time and the presence of humic acid in each fraction was determined spectrophotometrically. Humic acid breakthrough occurred mainly in fractions 3-6 in the Sephadex G-25 column used in this research.

An elution profile for a solution containing only 2,4,6-TCP is shown in Figure 1B. 2,4,6-TCP is much smaller than humic acid and eluted from the column mainly in fractions 16-26. Radiolabelled 2,4,6-TCP was analyzed separately from humic acid by scintillation counting. Activity of $^{14}C$ was converted to mass 2,4,6-TCP using the specific activity of the ($^{14}C$)-2,4,6-TCP.

Figure 1C depicts the chromatograph for a solution containing both 2,4,6-TCP and HA. The 2,4,6-TCP elutes from the column at two distinct intervals. The first peak (fractions 3-6) corresponds to the same retention time of that found for HA. This indicates that a portion of the 2,4,6-TCP is held in association with the HA and travels through the column at the same rate as the HA. The second peak is the same as that found when 2,4,6-TCP alone is passed through the column and represents the unbound (free) 2,4,6-TCP in solution. The method is similar to that used by Hasset and Anderson[2]. Those authors used gel chromatography to separate free cholesterol from DOM-bound cholesterol with good results.

2,4,6-TCP Sorption

The sorption affinity of 2,4,6-TCP for dissolved HA is demonstrated by the equilibrium isotherms in Figure 2. The isotherms are linear and follow the form $q = KC$ where q is the 2,4,6-TCP sorbed to HA (µg/g), C is the equilibrium concentration of the free 2,4,6-TCP (µg/L), and K is the distribution coefficient. The distribution coefficient can be normalized for 100% organic matter by adjusting for the non-organic impurities in the HA. The HA used in this study was determined to be 97% organic matter ($f_{om} = 0.97$)

$$K_{OM} = \frac{K}{f_{om}}$$

The K and $K_{OM}$ values for each isotherm are listed in Table 1. $K_{OM}$ values are in agreement with $K_{oc}$ values found by Schellenberg *et al.*[3] for 2,4,6 TCP. Those investigators reported $K_{OC}$ values ranging from 830 $cm^3/g_{oc}$ to 1310 $cm^3/g_{oc}$ with a mean value of 1070 $cm^3/g_{oc}$.

Values for $K_{OM}$ tend to decrease with increasing HA concentration in the microcosms. Carter and Suffet[14] found partition coefficients for non-polar organic compounds decreased with increased HA concentration and attributed it to possible analytical difficulties. Landrum *et al.*[5] also found partition coefficients for non-polar organics decreased with increased humic acid. The authors suggested that conformational changes in the HA structure with increasing concentration could alter pollutant binding. They also suggested that HA molecules may compete with organic pollutants for binding sites. The decrease in $K_{OM}$ and $K_{OC}$ with increasing organic material found in the three studies, each using a different analytical approach, indicate that changing HA binding characteristics may be responsible for the observed results rather than experimental anomalies.

2,4,6-TCP Mineralization

To determine the effect of HA on biodegradation of 2,4,6-TCP, mineralization by *Pseudomonas aeruginosa* was measured. Radiolabelled $^{14}CO_2$, generated from the complete aerobic oxidation of 2,4,6-TCP, was collected over time. *Pseudomonas aeruginosa* was able to mineralize ≈ 60% of the initial 1000 µg/l of 2,4,6-TCP in the microcosms (Figure 3). Overall mineralization of 2,4,6-TCP in solutions containing HA was reduced between 5-10% over HA-free 2,4,6-TCP solutions. Most of the total 2,4,6-TCP was mineralized rapidly; however a small fraction was mineralized very slowly.

HA-sorbed 2,4,6-TCP Biodegradation: Mineralization rates, determined by $^{14}CO_2$ collection, generate information for total soluble 2,4,6-TCP only. Biodegradation rates of free 2,4,6-TCP and HA-sorbed 2,4,6-TCP in solution cannot be distinguished by $^{14}CO_2$ collection.

To determine if HA sorbed 2,4,6-TCP can be biodegraded, microcosms containing 1000 μg/l 2,4,6-TCP and 100 mg/l HA were first allowed to equilibrate. Samples from these microcosms were then eluted through the gel column and the first peak, containing the HA-sorbed 2,4,6-TCP was collected. Multiple samples were fractionated and the HA-sorbed 2,4,6-TCP portions were combined.

A diluted bacterial inoculum was added to the HA-sorbed 2,4,6-TCP samples and the aqueous phase monitored over time. Figure 4 reveals that HA-sorbed 2,4,6-TCP is biodegraded. The 2,4,6-TCP concentration decreases with the addition of both live and dead (autoclaved) bacteria. This drop is due to sorption of 2,4,6-TCP to the added bacterial mass. The biodegradation rate of HA-sorbed 2,4,6-TCP is slow, which is very different than the rate found for the free 2,4,6-TCP in solution. In addition, the biodegradation rate appears linear, indicating that the availability of 2,4,6-TCP to the microorganisms is rate-controlled.

## REFERENCES

Schnitzer, M. and S. U. Khan, 1972. In: Humic Substances in the Environment. A. Douglas McLaren (ed), Marcel Dekker, Inc., New York, New York. 327pp.

Hassett, J. P. and M. A. Anderson, 1982. Effects of Dissolved Organic Matter on Adsorption of Hydrophobic Organic Compounds by River and Sewage Borne Particles. Water Research, 16:681-686.

Schellenberg, K., C. Leuenberger, and R. P. Schwarzenbach, 1984. Sorption of Chlorinated Phenols by Natural Sediments and Aquifer Materials. Env. Sci. Tech., 18:652-657.

Carter, C. W. and I. H. Suffet, 1982. Binding of DDT to Dissolved Humic Materials. Env. Sci. Tech., 16:735-740.

Landrum, P. F., S. R. Nihart, B. J. Eadle, and W. S. Gardner, 1984. Reverse-Phase Separation Method for Determining Pollutant Binding to Aldrich Humic Acid and Dissolved Organic Carbon of Natural Waters. Env. Sci. Tech., 18:187-192.

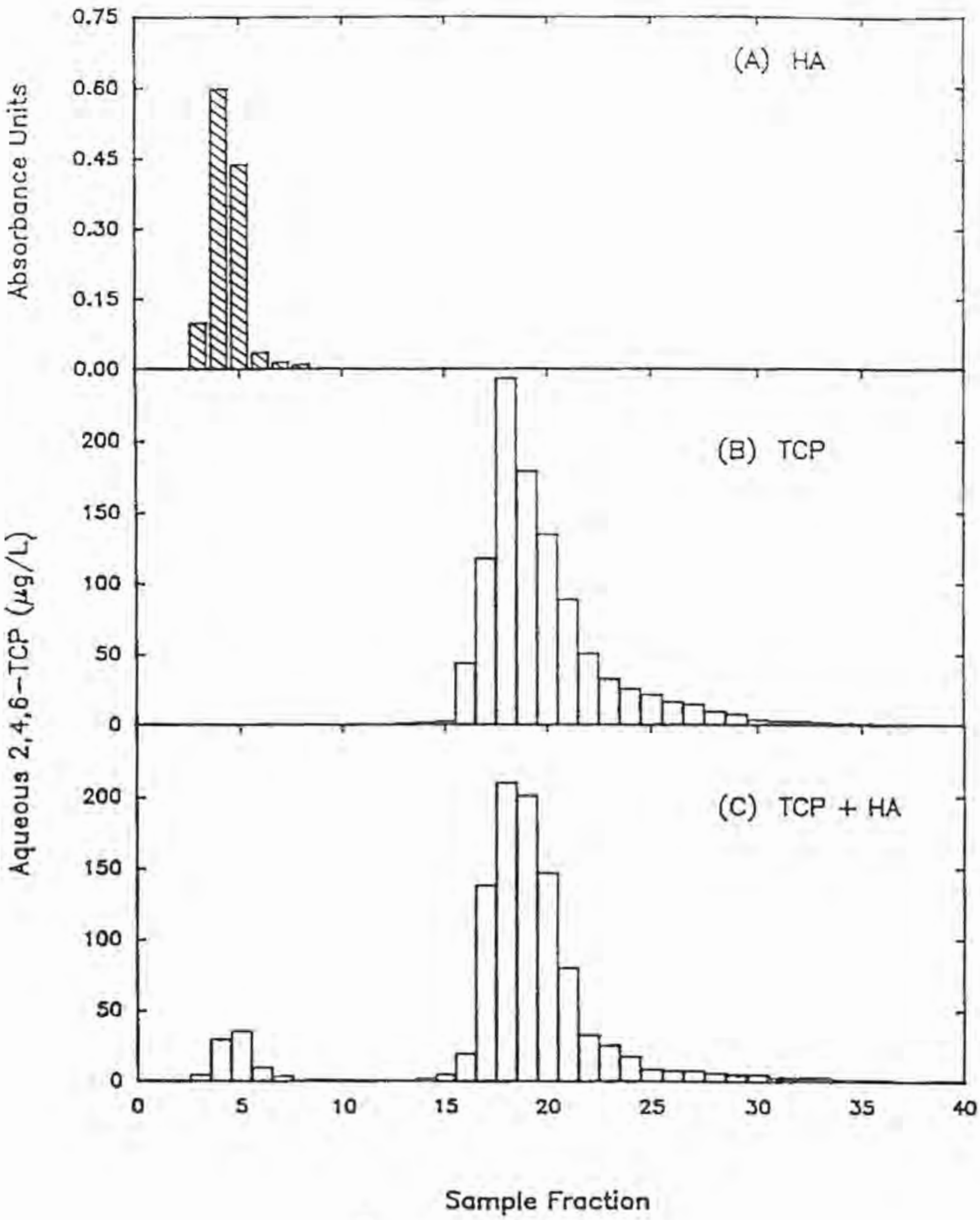


Figure 1. Sephadex G-25 size exclusion chromatograms of solutions containing: a) dissolved humic acid, b) 2,4,6-TCP, c) 2,4,6-TCP and dissolved humic acid.

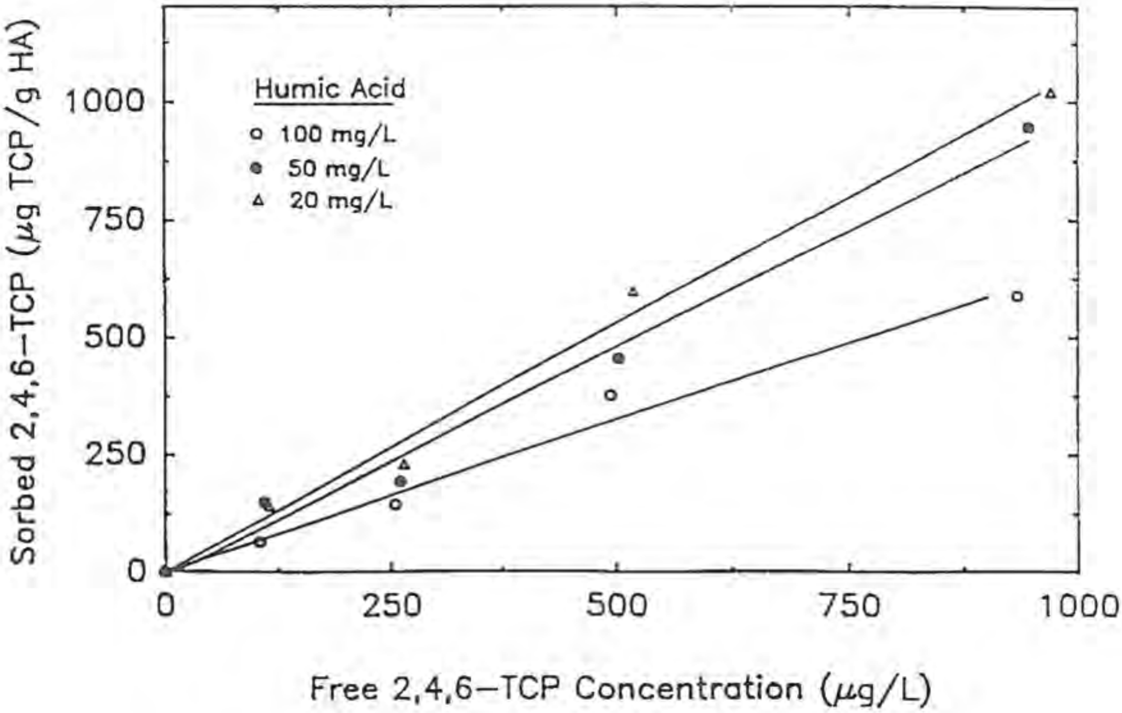


Figure 2. Equilibrium isotherms for 2,4,6-TCP with three concentrations of dissolved humic acid.

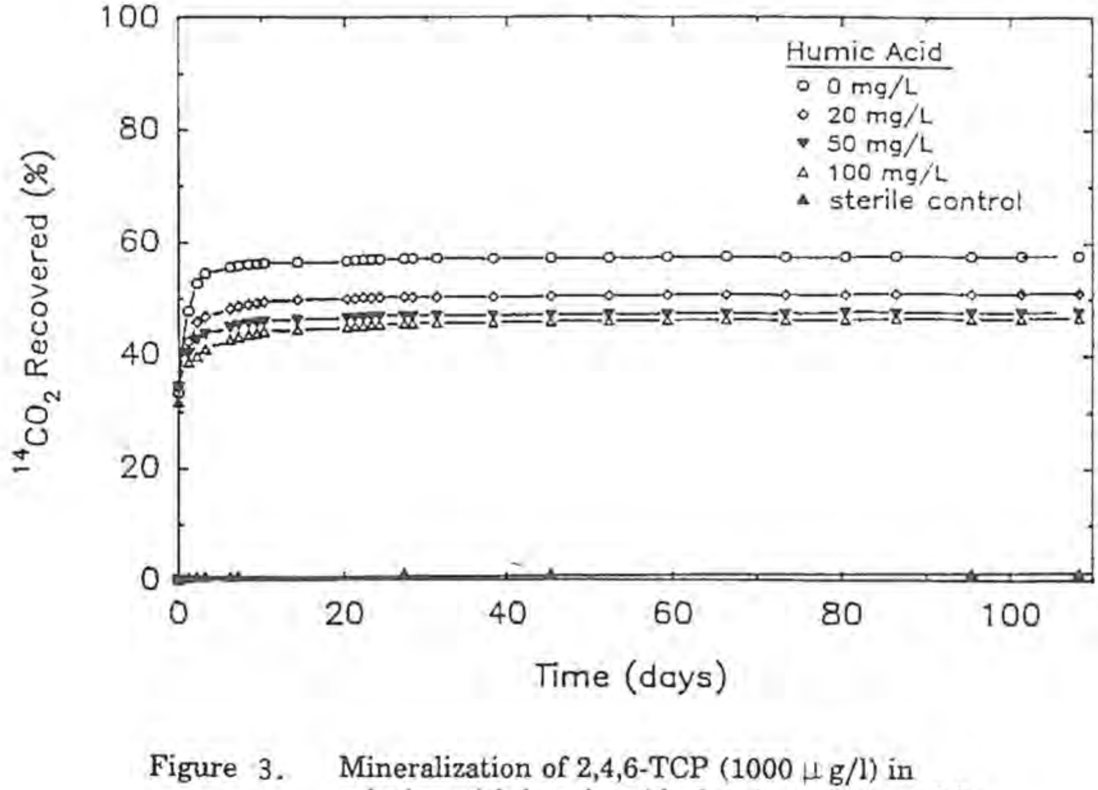


Figure 3. Mineralization of 2,4,6-TCP (1000 $\mu$g/l) in solution with humic acid after inoculation with *Pseudomonas aeruginosa*.

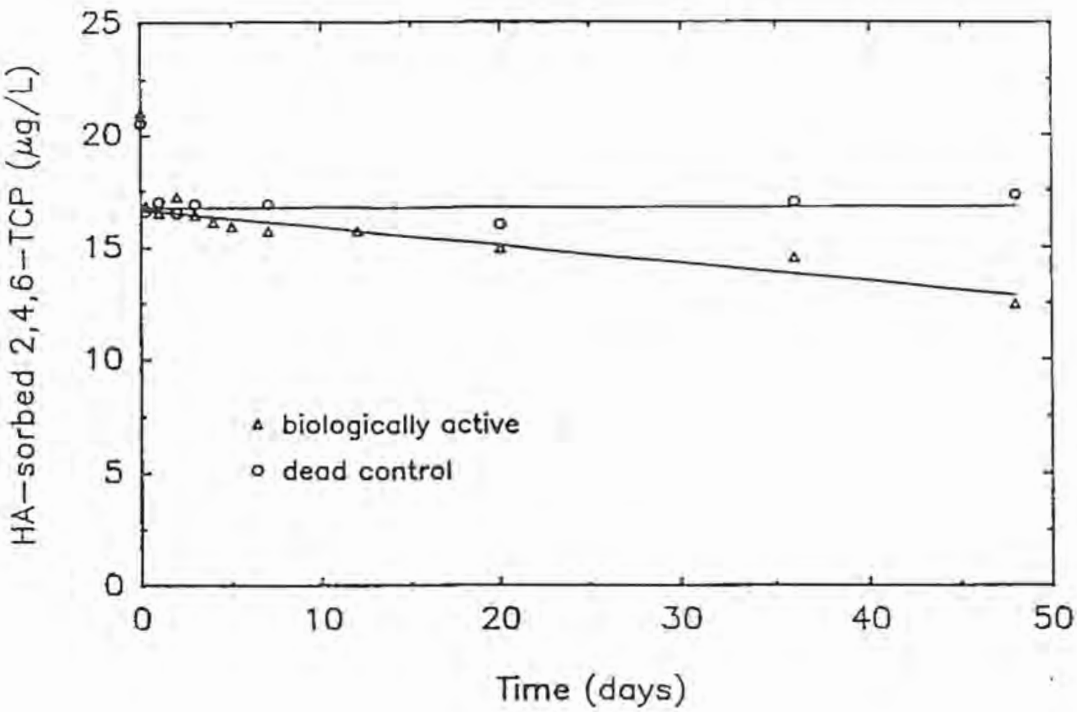


Figure 4. Biodegradation of HA-sorbed 2,4,6-TCP in solution using acclimated bacteria.

Table 1. Distribution coefficients of 2,4,6-TCP for dissolved humic acid (HA).

| HA concentration (mg/l) | K ($cm^3/g_{HA}$) | $K_{OM}$ ($cm^3/g_{om}$) |
|---|---|---|
| 100 | 649 | 669 |
| 50 | 981 | 1011 |
| 20 | 1062 | 1095 |

# EVALUATION OF METHODS FOR ANALYSIS OF SLUG TESTS IN FRACTURED ROCKS[1]

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Since 1985, slug tests have been extensively used to characterize the permeability of fractured rocks on the Oak Ridge Reservation (ORR) of U.S. Department of Energy. There were several reasons for the use of slug tests to obtain this information. First, pumping tests are time consuming for both data collection and data analysis; only nine pumping tests have been made in this area in the past 10 years. Second, hydraulic conductivity and transmissivity values span at least eight orders of magnitude, and it would be impractical to obtain enough data to characterize this range with pumping tests. A hydraulic conductivity or transmissivity value from a slug test is not as accurate as would be obtained from a pumping test, and a slug test represents only a small volume of rock. Nevertheless, the statistical distributions of large numbers of hydraulic conductivity and transmissivity values from slug tests should be more representative of the rocks than is a small number of values from pumping tests. Third, slug tests produce minimal disturbances and little or no waste in contaminated areas.

Presently, 650 hydraulic conductivity values and 475 transmissivity values have been calculated from slug tests and a few pumping tests on the ORR. In general, the statistical distributions of these data adequately characterize both water-producing zones and matrix zones in the regolith and the principal rock units. Remaining deficiencies include too few data for the Knox Group and the Rome Formation, minimal data for wells deeper that 30 m, few transmissivity values near both ends of the range, and large areas that have few or no tests.

The Bouwer and Rice (1976) method for analysis of slug test data has been used to calculate a large majority of the hydraulic conductivity values. Other tests were analyzed with the Hvorslev (1951) method. The hydraulic conductivity values calculated with these two methods are nearly the same. Also, both methods are flexible and do not require perfect early data. For these reasons, both methods are widely used even though the assumption of an isotropic and homogeneous material within the screened or open interval requires careful consideration in fractured rock.

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[1]Research sponsored by Nuclear and Chemical Waste Programs, Office of Energy Research, U.S. Department of Energy under contract DE-AC05-84OR21400 with Martin Marietta Energy Systems, Inc.

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Imperfect early data are observed in a majority of slug tests and have a variety of causes, including (1) a non-instantaneous slug, (2) a wellbore-storage or sandpack-storage effect, (3) a sandpack-permeability effect or inadequate well development, (4) head losses caused by turbulent flows, (5) erroneous signals from the pressure transducer, (6) an inertial effect, which produces cyclic fluctuations of water level, and (7) inhomogeneities near the well. The effects of these causes of imperfect early data decrease with time, and both the Bouwer and Rice method and the Hvorslev method of data analysis depend upon fitting a straight line to a semilogarithmic graph of valid data points. A series of valid data points almost always occurs in the data and is easily identified by a good fit to a line during the period of time that includes most of the water-level response to the slug. The fitted line can be projected back to zero time, and hydraulic conductivity can be calculated from the slope and intercept of this line.

An assumption of an isotropic and homogeneous aquifer is made because the total permeability of the screened or open interval is divided by the length of this interval to calculate hydraulic conductivity in both methods of data analysis (Bouwer and Rice 1976, Eq. 5; Freeze and Cherry 1979, Eq. 8.34). The calculation of hydraulic conductivity thus assumes that the length (L) of the screened or open interval in the well is the same as the thickness (D) of the fracture zone that responds to the slug or that $L < D$ and $K_h >> K_v$, where $K_h$ and $K_v$ are the horizontal and vertical hydraulic conductivities. A calculated hydraulic conductivity value is too small if $L > D$ because the screened or open interval includes matrix zones. In comparison, a transmissivity value calculated from slug test data is too small if $L < D$, but the value is correct if $L = D$ or $L > D$.

If the lengths of screened or open intervals in wells on the ORR are the same as the thicknesses of the fracture intervals that respond to slug tests ($L = D$), average transmissivity will increase with screen length. Instead, as shown by the data below, the geometric mean of transmissivity is nearly the same for screen lengths of 1-6 m; some longer lengths of screen or openhole might include a second responsive fracture zone.

| | | Transmissivity ($m^2/d$) | | |
|---|---|---|---|---|
| Screen length (m) | Number of values | Geo-metric mean | Mean minus one standard deviation | Mean plus one standard deviation |
| 1.0 - 1.5 | 75 | 0.17 | 0.087 | 0.31 |
| 1.6 - 3.0 | 245 | 0.17 | 0.070 | 0.43 |
| 3.1 - 4.5 | 30 | 0.14 | 0.087 | 0.22 |
| 4.6 - 6.0 | 20 | 0.18 | 0.050 | 0.65 |
| 6.1 - 9.0 | 30 | 0.28 | 0.14 | 0.56 |

In wells on the ORR, $L > D$, and the average borehole interval that responds to a slug test is apparently less than 1.0 m.

The results obtained above indicate that transmissivity values are approximately correct in this area but that hydraulic conductivity values are too small and should be multiplied by L/D. A number of tests with a sensitive borehole flowmeter or closely-spaced packers will be needed to accurately determine the lengths of the borehole intervals that respond to slug tests. The results also suggest that a water-producing zone consists of a single fracture that has a nearly horizontal orientation. It is more likely, however, that a large range of fracture apertures occurs within any rock volume but that only the one or two fractures with the largest apertures produce nearly all of the response to a slug.

All transmissivity values obtained from slug tests on the ORR have been calculated with the method of Cooper et al. (1967). This method is mathematically exact and includes the effects of storativity. It is also interesting that slug test data from a well in fractured rock were used by the authors (pp. 265-268) to demonstrate the method. This method is sensitive to imperfect early data, however, and some data cannot be matched to a type curve. Commonly, the data plot is too steep or the first series of points is below the type curves. These problems are the main reason for a larger number of hydraulic conductivity than transmissivity values on the ORR. If a reasonable value of storativity can be assumed in these cases, a match of the test data to a type curve is unnecessary.

The alternative method of calculating transmissivity from slug test data depends on the fact that adjacent type curves represent a difference in transmissivity much less than the order of magnitude difference in storativity (Cooper et al. 1967, pp. 267-268). Nearly all slug test data from the ORR can be matched to curves that represent storativities of $10^{-3}$ to $10^{-7}$ ($a = 10^{-3}$ to $10^{-6}$; Cooper et al. 1967, pp. 265-266). The midpoint of the $H_t/H_o$ range at the match point for these curves is 0.65 ($a = 10^{-4}$), where $H_t$ is the water level stage at time t, and $H_o$ is the stage produced by the slug at time zero. If it is assumed that $H_t = 0.65H_o$, values of $H_o$, $H_t$, and t can be obtained from the fitted line on a Bouwer and Rice plot of the data. If actual storativity is in the assumed range, the calculated transmissivity will have a maximum error of plus or minus 50%. For comparison, an erroneous match of perfect early data with an adjacent type curve produces an error in transmissivity of plus or minus 10-35% for the same range of storativity. The maximum error for the alternative method should be acceptable for most purposes.

The simpler method of determining transmissivity can be demonstrated with slug test data (Fig.1) from a well at Oak Ridge National Laboratory. The early data from this well are imperfect because of a non-instantaneous slug and possibly because of inadequate well development. Late data also deviate from a straight line on a Bouwer and Rice plot because the effect of the slug on pressure heads in the fractures is a significant percentage of the residual water level stage in the well (Bouwer 1989, p. 306). A projection of the fitted line on Figure 1 shows that $H_o = 5.0$ ft. Then,

$$H_t = 0.65(5.0) = 3.25 \text{ ft}$$

at time, t = 47 sec. The calculated transmissivity is 4.5 $m^2/d$. If $H_o$ is assumed to be 3.75 ft, the maximum stage during the test, the data plot is too steep for any of the type curves. If the slug test data are corrected to determine $H_o$ by either the regression or the translation methods of Pandit and Miner (1986, p. 746), the calculated transmissivity value is approximately the same as that of the alternative method.

The alternative method of calculating transmissivity is similar to the regression method of Pandit and Miner (1986), but bad data points can be ignored, and a series of valid data points is easily identified with a graphical procedure. If early data are perfect, the alternative method is not as accurate as the original method of Cooper et al. (1967). Also, the storativity range must be known or assumed for the alternative method, and the maximum error should be calculatedto determine whether or not it is acceptable. Nevertheless, the alternative method can be used to calculate reasonable values of transmissivity where early data are imperfect and where the reason for the problem is not easily determined; cases of this type occur in about 25% of the slug tests on the ORR.

## REFERENCES

Bouwer, Herman, 1989. The Bouwer and Rice Slug Test--An Update. Ground Water, 27 (3): 304-309.

Bouwer, Herman and R. C. Rice, 1976. A Slug Test for Determining Hydraulic Conductivity of Unconfined Aquifers with Completely or Partially Penetrating Wells. Water Resour. Res., 12 (2): 423-428.

Cooper, H. H., Jr., J. D. Bredehoeft, and I. S. Popadopulos, 1967. Response of a Finite-Diameter Well to an Instantaneous Charge of Water. Water Resour. Res., 3(1): 263-269.

Freeze, R. A. and J. A. Cherry, 1979. Groundwater. Prentice-Hall, Englewood Cliffs, New Jersey.

Hvorslev, M. J, 1951. Time Lag and Soil Permeability in Groundwater Observations. U.S. Army Corps of Engineers, Waterways Experiment Station Bull. 36, Vicksburg, Miss.

Pandit, N. S. and R. F. Miner, 1986. Interpretation of Slug Test Data. Ground Water, 24 (6): 743-749.

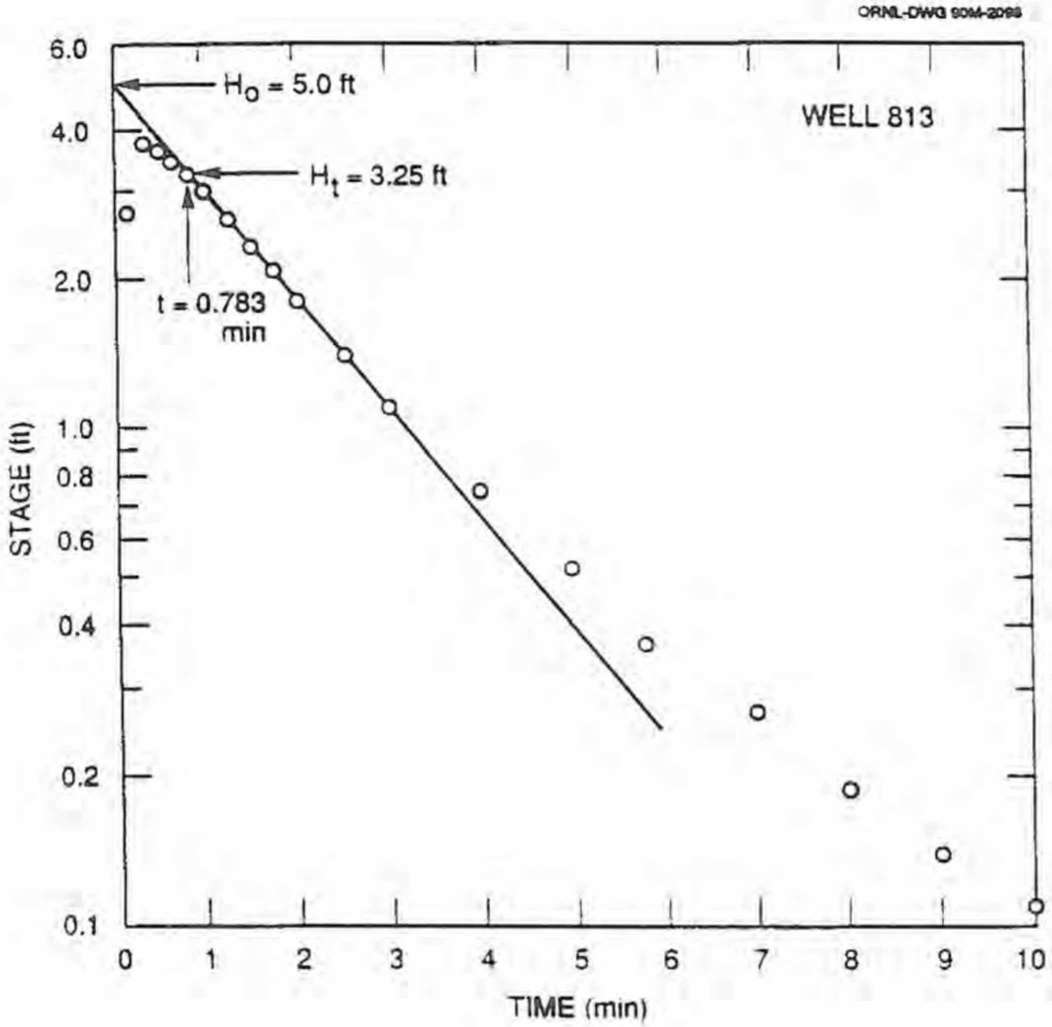


Figure 1: Bouwer and Rice plot of slug test data from a well in fractured rock at Oak Ridge National Laboratory.

# GROUNDWATER QUALITY OF FARMSTEAD WELLS, TENNESSEE

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A statewide sampling program of 150 farmstead wells is being conducted throughout Tennessee. The sampling program consists of a one-time sampling of ground water from wells that are used primarily for drinking purposes. Sampling sites were selected to provide data that are representative of the various hydrogeological characteristics and major agricultural activities within the State, including crop, dairy, cattle, swine, and poultry production. Samples are being analyzed for selected nutrients, bacteria, common and trace inorganic ions, and total organic carbon. They are also being scanned using gas chromatograph/flame-ionization detection techniques to semi-quantitatively determine organic-compound concentrations. Sampling is scheduled to be completed in June 1990.

Analysis of samples from nearly 100 wells have been partly or fully completed. Results of the study to date indicate generally good but variable ground-water quality. Specifically:

- About 75 percent of the wells sampled in central and eastern Tennessee tested positive for bacteria, compared to only about 25 percent of the wells sampled in the western part of the State.

- Although samples from only three wells contained concentrations of nitrate nitrogen that equaled or exceeded the 10 milligram per liter primary maximum contaminant level established for drinking water, samples from 18 wells had concentrations greater than 3 milligrams per liter, a figure suggested as a threshold indicative of human influence.

- Widespread contamination by organic compounds is not evident. Concentrations of total organic carbon generally were less than 1 milligram per liter and concentrations of gas chromatograph/flame-ionization detectable organic compounds were less than 5 micrograms per liter in nearly all samples.

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## APPLICATION OF CHEMISTRY TO IDENTIFICATION OF GROUNDWATER FLOWPATHS IN BEAR CREEK VALLEY NEAR OAK RIDGE, TENNESSEE[1]

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### BACKGROUND

The groundwater system in the vicinity of the Bear Creek Burial Grounds study area (Fig. 1) can be conceptualized as a single interconnected aquifer with markedly different hydraulic properties which are attributable to contrasting lithologies and structural features (Geraghty and Miller, Inc. 1985). The aquifer generally is composed of an upper zone of weathered unconsolidated material overlying a lower bedrock zone. Although the unconsolidated zone is sometimes more permeable than the bedrock zone, there is no sharp discontinuity of permeability between them and both respond similarly in terms of water-level fluctuations and ground-water flow directions. Two bedrock components of the aquifer system underlying the study area are of particular importance to groundwater flow. These components are; (1) the primary shale formations of the Conasauga Group (Maryville Limestone and the Nolichucky Shale), and (2) the Maynardville Limestone.

The northern portion of the study area is underlain by the Maryville Limestone, and the middle portion of the study area is underlain by the Nolichucky Shale. Within Bear Creek Valley, both of these formations consist of interbedded shales and limestone; the Maryville Limestone consists of approximately 60% limestone and 40% shale, while the Nolichucky Shale consists of approximately 30 percent limestone and 70 percent shale (King and Haase 1987). Aquifer pumping tests conducted to determine the hydraulic properties of these formations have been typified by very low yields (usually less than five gallons per minute) and ellipsoidal water-level cones of depression, elongated parallel to geologic strike (Geraghty and Miller, Inc. 1985). These observations have been interpreted by several investigators to reflect the low permeability and strong anisotropy of the formations where bedding planes provide preferred ground-water flow paths along strike and down dip. The southern portion of the study area is underlain by the Maynardville Limestone, which is the principal water-bearing formation within the Conasauga Group (King and Haase 1987). The water-bearing capacity of the Maynardville Limestone has been greatly enhanced by solutional

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[1] Research sponsored by the Environmental Surveillance Section of the Environmental Monitoring Department at the Oak Ridge Y-12 Plant, operated by Martin Marietta Energy Systems, Inc. under contract DE-AC05-84OR21400 with the U.S. Department of Energy.

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enlargement of structural and stratigraphic features such as fractures, joints, and bedding planes. Evidence of the solutional cavity system in the Maynardville Limestone can be observed in outcrops in Bear Creek and inferred from drilling logs. This cavity system is believed to be the major discharge area for shallow and intermediate-depth groundwater within the primary shale formations of the Conasauga Group.

The direction of shallow groundwater flow within the Bear Creek Burial Grounds study area is generally towards Bear Creek. Studies have shown that the solution cavities in the Maynardville Limestone are the major discharge areas for shallow and intermediate depth groundwater moving through the primary shale formations of the Conasauga Group. Groundwater discharge from the Maynardville Limestone sustains the flow of Bear Creek, or at times of low flow, moves through the solution cavities underlying Bear Creek (Geraghty & Miller, Inc. 1985).

In the low-lying parts of the Bear Creek Burial Grounds study area, upward components of groundwater flow are commonly observed in wells screened at depths below 50 ft in the Maryville Limestone and Nolichucky Shale. Such gradients are expected, based on a two-dimensional hydrologic cross section of Bear Creek Valley that was determined for a locality approximately 1.75 miles east of the study area (King and Haase 1989). Downward components of flow have been noted locally within the Maynardville Limestone between depths of 40 and 200 ft near the headwaters of Bear Creek. This finding supports the hypothesis that the Maynardville Limestone has a comparatively high permeability and drains groundwater from adjacent shale formations in Bear Creek Valley (Geraghty and Miller 1988, 1989; Bailey 1988).

## DATA SOURCES

Results of quarterly sampling of groundwater monitoring wells in the vicinity of Burial Ground A-South are presented in annual Groundwater Quality Assessment Reports (GWQARs) for 1986 through 1989 (Geraghty and Miller 1987, 1988, 1989, 1990). Data presented in these GWQARs for inorganic and organic constituents in wells GW-45, GW-46, GW-47, GW-68, GW-71, GW-72, GW-94, GW-95, GW-117, GW-118, GW-119, GW-126, GW-237, GW-374, and GW-375 (see Fig. 1) were examined to determine groundwater chemistry trends.

## GROUNDWATER CHEMISTRY TRENDS

All groundwater analyzed from within the study area was obtained from wells completed in either soils and residuum, weathered bedrock, or unweathered bedrock of the Nolichucky Shale. Three chemically distinctive groundwaters occur within the study area . A fourth groundwater type, calcium-magnesium-bicarbonate-chloride, is noted within the area, but that groundwater type is associated with a contaminant plume associated with Burial Ground A-South and is interpreted to be a derivative of one of the three basic groundwater types. The three types of groundwaters occur at differing depths and represent a chemical evolution of groundwater from a calcium-magnesium-bicarbonate type at shallow depths (≤80 ft below ground surface), through a sodium-bicarbonate type at intermediate depths (80 to 300? ft), to a sodium-chloride-bicarbonate type at depths of 500 ft and greater. The depths at which the transitions occur between the sodium bicarbonate and the sodium-chloride-bicarbonate groundwaters are poorly constrained because of the lack of groundwater monitoring wells at appropriate depths within the study area.

A calcium-magnesium-bicarbonate groundwater and a derivative groundwater type, calcium-magnesium-bicarbonate-chloride, are sampled by wells GW-45, GW-46, GW-47, GW-68, and GW-237. The interval of investigation in these wells ranges from 3-13 ft below ground surface in well GW-45 to 71-82 ft below ground surface in well GW-68. Wells GW-45, GW-47, and GW-237 sample groundwater that is not affected by the contaminant plume associated with Burial Ground A-South. The groundwater from these three wells is the calcium-magnesium-bicarbonate type, and plots within the "Shallow-U" field in Fig. 2. The groundwater typically exhibits pH values of 6.5 to 7.5, and has Ca/Ca+Mg ratios of ≥0.70 and Na+K/Na+K+Ca+Mg ratios ≥0.60. Mineral saturation calculations for groundwaters from wells GW-45 and GW-237 using the computer code WATEQ indicate that shallow calcium-magnesium-bicarbonate groundwaters are typically saturated with calcite and slightly undersaturated with dolomite.

Wells GW-46 and GW-68 sample a calcium-magnesium-bicarbonate-chloride groundwater that is contaminated with various volatile organic compounds (VOCs) that range in concentration from several hundred to approximately 2800 µg/L (Haase and King 1990). This groundwater type plots within the "Shallow-C" field in Fig. 2. It exhibits a similar range of pH and Na+K/Na+K+Ca+Mg values, a slightly lower Ca/Ca+Mg ratio, and a higher $Cl/Cl+HCO_3+CO_3$ ratio than the uncontaminated shallow groundwater. Mineral saturation calculations for well GW-68 indicate that contaminated shallow groundwaters are also typically saturated with calcite and slightly undersaturated with dolomite. Because of the similar cation compositions of the two shallow groundwaters and nature of the compositional differences between them, the contaminated shallow groundwater (calcium-magnesium- bicarbonate-chloride type) is interpreted to be an anthropogenically-produced derivative of the uncontaminated calcium-magnesium-bicarbonate type.

A sodium-bicarbonate groundwater is sampled by wells GW-71, GW-72, GW-94, GW-126, GW-374, and GW-375. The interval of investigation in these wells ranges from 87-98 ft below ground surface in well GW-72 to 198-219 ft below ground surface in well GW-71. The groundwater from these wells plots within the "Intermediate" field in Fig. 2. The sodium-bicarbonate groundwater typically exhibits pH values of 6.0 to 8.0, and has Ca/Ca+Mg ratios of ≥0.50 and Na+K/Na+K+Ca+Mg ratios ≥0.95. Mineral saturation calculations for groundwaters from wells GW-126, GW-374, and GW-375 indicate that intermediate- depth sodium-bicarbonate groundwaters are typically saturated with both calcite and dolomite. Additionally, for those analyses where barium concentrations were above detection limits, the calculations indicate that the groundwaters are saturated with barite. Only one of the wells in this intermediate depth group, well GW-71, exhibits significant VOC contamination. Summed levels of VOCs in that well typically range from 1500 to 4800 µg/L. The composition of the VOCs within the groundwater sampled by well GW-71 is significantly different from that in the groundwater sampled by shallow, contaminated wells GW-46 and GW-68 (Haase and King 1990). Additionally, unlike groundwater from the shallow, contaminated wells (GW-46 and GW-68), the groundwater from well GW-71 does not exhibit significant differences in the concentration of anions when compared to uncontaminated groundwater from intermediate depths. Based on available sampling points, the transition between shallow calcium-magnesium-bicarbonate groundwater and intermediate-depth sodium-bicarbonate groundwater occurs at depths of approximately 80 to 100 ft below ground surface. The data also suggest that this transition is quite abrupt.

A sodium-chloride-bicarbonate groundwater is sampled by wells GW-117, GW-118, and GW-119. The interval of investigation in these wells ranges from 480-530 ft below ground surface in well GW-117 to 525-575 ft below ground surface in well GW-118. The groundwater from these wells plots within the "Deep" field in Fig. 2. The sodium-chloride-bicarbonate groundwater typically exhibits pH values of 8.5 to 10.0, and has widely variable Ca/Ca+Mg ratios and Na+K/Na+K+Ca+Mg ratios ≥0.95. Mineral saturation calculations for groundwaters from wells GW-117, GW-118, and GW-119 indicate that deep sodium-bicarbonate groundwaters are typically saturated with both calcite and dolomite. Additionally, for those analyses where barium concentrations were above detection limits, the calculations indicate that the groundwaters range from saturation to slight undersaturation with barite. Based on available sampling points, the transition between intermediate-depth sodium-bicarbonate and deep sodium-chloride-bicarbonate groundwaters occurs between depths of 200 and 500 ft below ground surface. The precise location of the transition and its abruptness cannot be determined from available data.

## SUMMARY

The three types of groundwaters represent a chemical evolution of groundwater from a calcium-magnesium-bicarbonate type at shallow depths (≤ 80 ft below ground surface), through a sodium-bicarbonate type at intermediate depths (80 to 300? ft), to a sodium-chloride-bicarbonate type at depths of 500 ft and greater. Available hydrogeological data for the site suggest that this chemical evolution with depth represents a sampling of groundwaters with progressively longer flowpaths through the Nolichucky Shale and that the chemical evolution exhibited by groundwaters at the site is due to progressively greater water/rock interaction as flowpath length increases.

## REFERENCES

Bailey, Z. C., 1988. Preliminary Evaluation of Groundwater Flow in Bear Creek Valley, the Oak Ridge Reservation, Tennessee. U. S. Geological Survey Water-Resources Investigations Report 88-4010.

Geraghty and Miller, Inc., 1985. Remedial Alternatives for Bear Creek Valley Waste Disposal Area. Y/SUB/85-00206C/3.

Geraghty and Miller, Inc., 1987. Groundwater Quality Assessment Report for the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Y-12 Plant, 1986. Y/SUB/87-00206C/3.

Geraghty and Miller, Inc., 1988. Groundwater Quality Assessment Report for the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Y-12 Plant, 1987. Y/SUB/88-00206C/3.

Geraghty and Miller, Inc., 1989. Groundwater Quality Assessment Report for the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Y-12 Plant, 1988. Y/SUB/89-00206C/9.

Geraghty and Miller, Inc., 1990. Groundwater Quality Assessment Report for the Bear Creek Burial Hydrologic Regime, 1989: I - Groundwater Quality Data and Calculated Rate of Contaminant Migration. Y/SUB/90-00206C/1 Part 1.

Haase, C. S. and H. L. King, 1990. Report and Preliminary Assessment of the Occurrence of Dense, Nonaqueous Phase Liquids in the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Oak Ridge Y-12 Plant. Y/TS-629.

King, H. L. and C. S. Haase, 1987. Subsurface-Controlled Geological Maps for the Oak Ridge Y-12 Plant and Portions of Bear Creek Valley. ORNL/TM-10112.

King, H. L. and C. S. Haase, 1989. Results and Preliminary Interpretation of Hydrogeologic Packer Testing in Core Holes CH-157, GW-131, and GW-132 through GW-135 at the Oak Ridge Y-12 Plant. Y/TS-495.

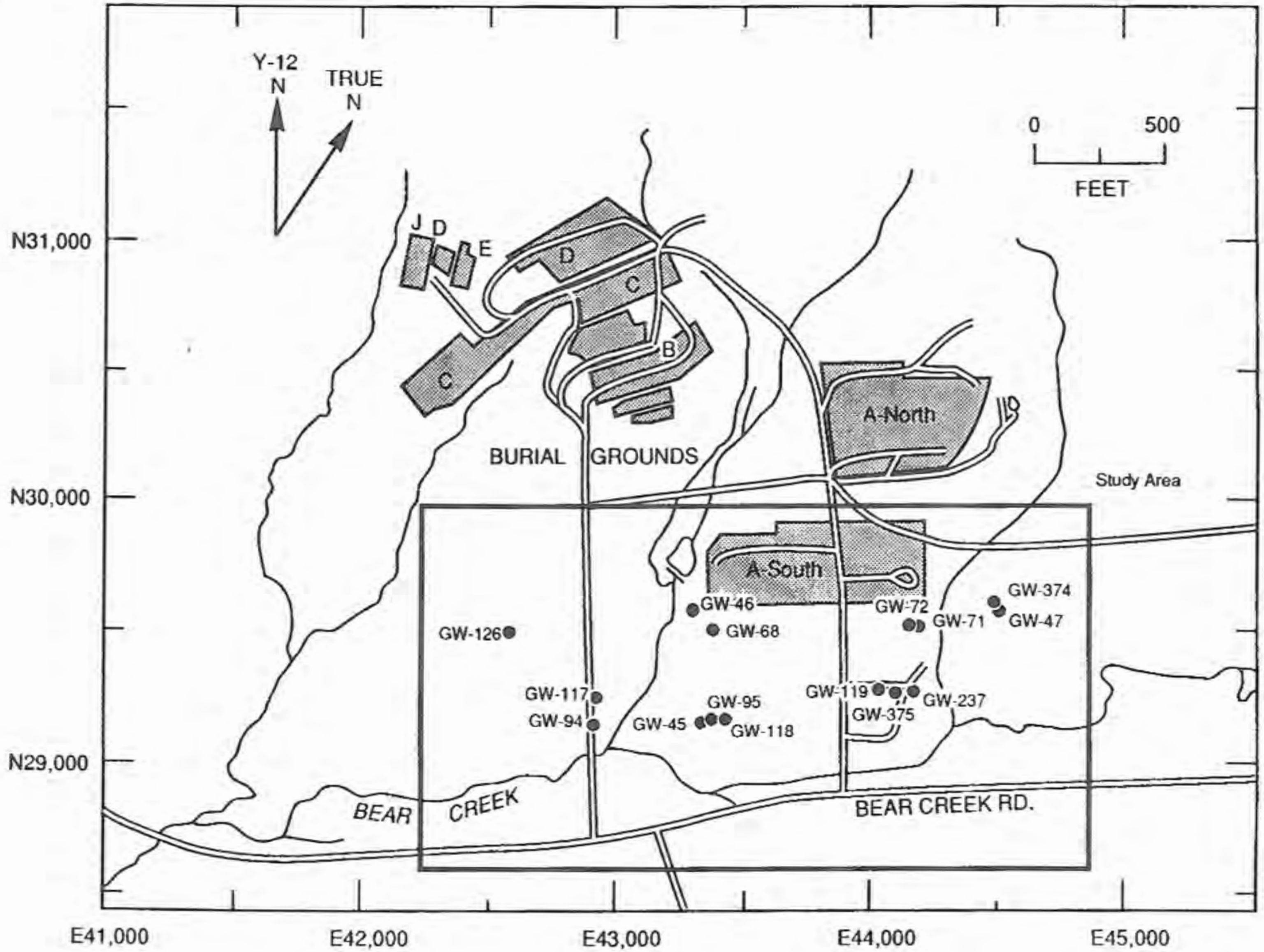


Figure 1. Map of study area showing location of selected groundwater monitoring wells from which groundwater samples were obtained.

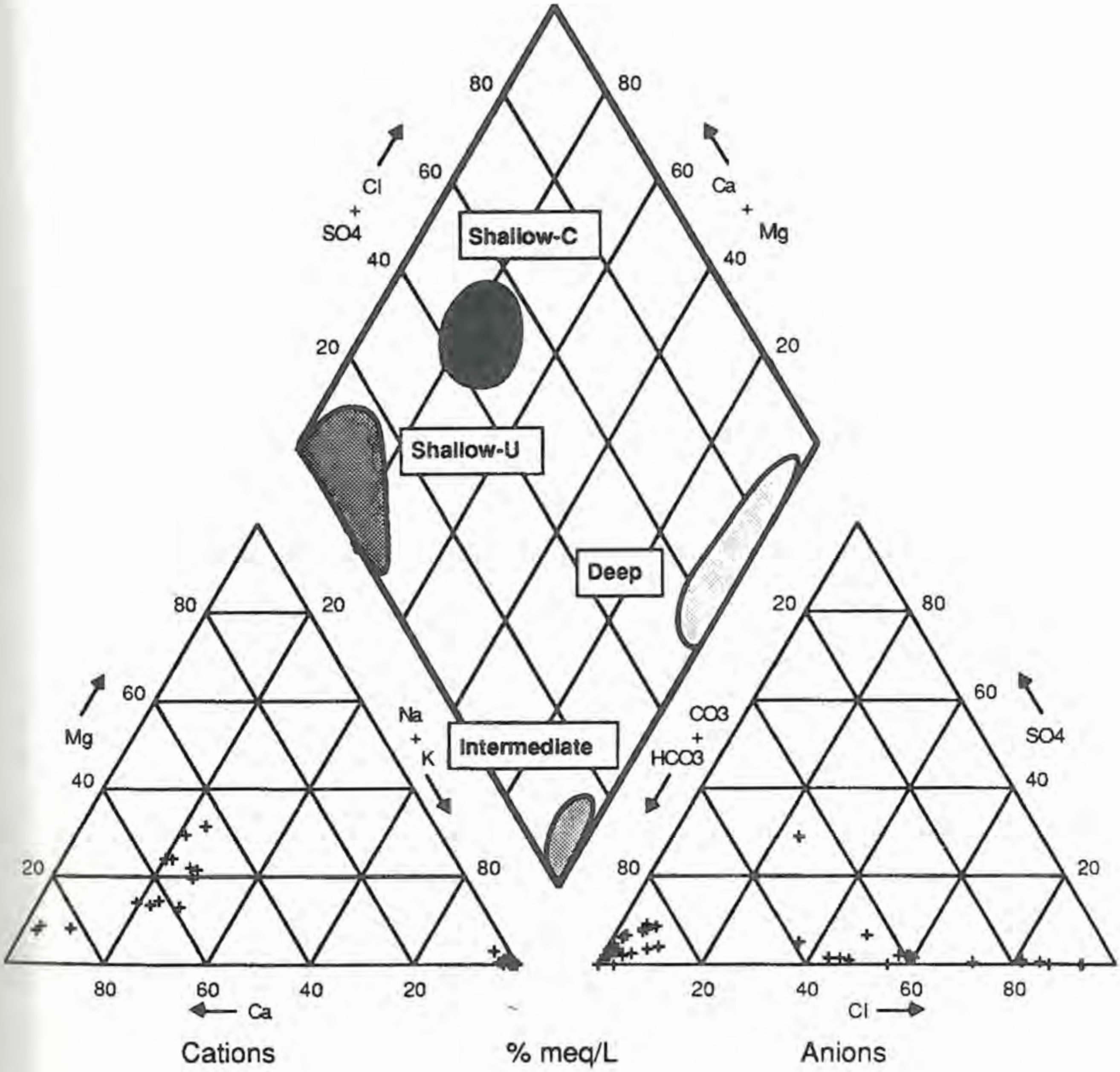


Figure 2. Piper diagram illustrating groundwater types within the Bear Creek Burial Ground study area.

# A HYDROCHEMICAL SURVEY OF CARBONATE GROUNDWATERS, PUTNAM AND JACKSON COUNTIES, TENNESSEE

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## INTRODUCTION

Water-quality analyses of well and spring water samples collected in aquifers in the Mississippian Fort Payne and the Ordovician Catheys and Leipers Formations during 1988-1989 (Collar, 1989) reveal diagnostic differences between the two carbonate-aquifer waters. An examination of water-quality trends among total dissolved solids (TDS), calcium, magnesium, sodium, and the saturation indices of calcite and gypsum reveal that Ordovician ground waters have generally higher ionic strengths than Fort Payne waters and are closer to chemical equilibrium with the aquifer wall rock. Fort Payne waters were bimodal, with a low-TDS group of waters indicating ground-water flow through highly open, solutionally enlarged fracture systems, and a higher-TDS group of waters indicating more diffuse ground-water flow. The distribution of Ca and Mg in the two aquifer waters is controlled by the presence of dolomite in the Fort Payne aquifer rock and the absence of dolomite in the Ordovician aquifer rock.

## STUDY AREA

The field area encompasses parts of the Eastern Highland Rim and the Outer Central Basin physiographic provinces in north central Tennessee (figure 1). The resistant nature of the siliceous Fort Payne Formation (Miller, 1979) has promoted the development of the low-gradient Highland Rim landsurface. The Ordovician limestones below and the Mississippian limestones above are more susceptible to dissolution, and greater rates of chemical weathering are probably responsible for the development of both the Highland Rim and Cumberland Escarpments (Reesman and Godfrey, 1972). The organic-and uranium-rich Devonian-Mississippian Chattanooga Shale (Glover, 1959; Conant and Swanson, 1961) separates the Fort Payne and Ordovician rocks by an average of 40 feet. The Chattanooga Shale behaves geomorphically as a slope-forming unit and typically crops out on the shoulder of the Highland Rim Escarpment. The Chattanooga Shale forms a regional aquiclude and prohibits the descent of recharge waters originating as surface runoff along the Cumberland Escarpment and Highland Rim Plateau. Locally, intense fracturing of the Chattanooga Shale may allow leakage of Fort Payne waters into the Ordovician rock.

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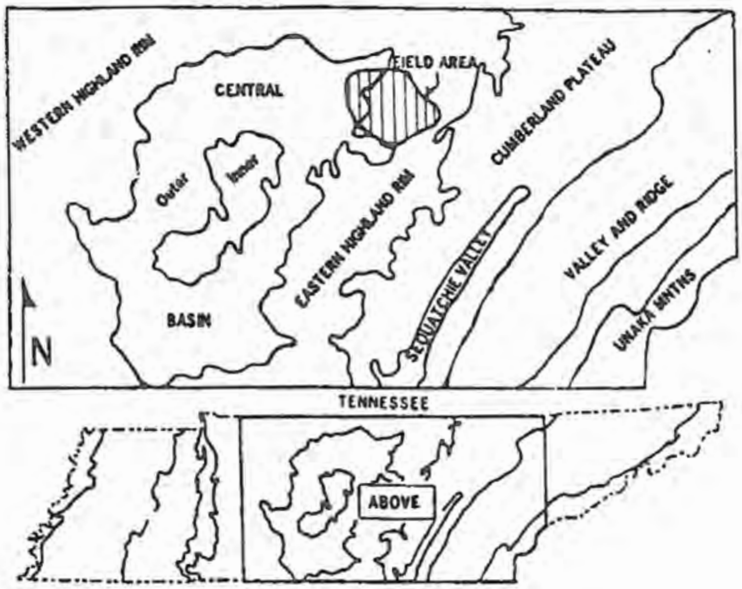


Figure 1: Location of study area.

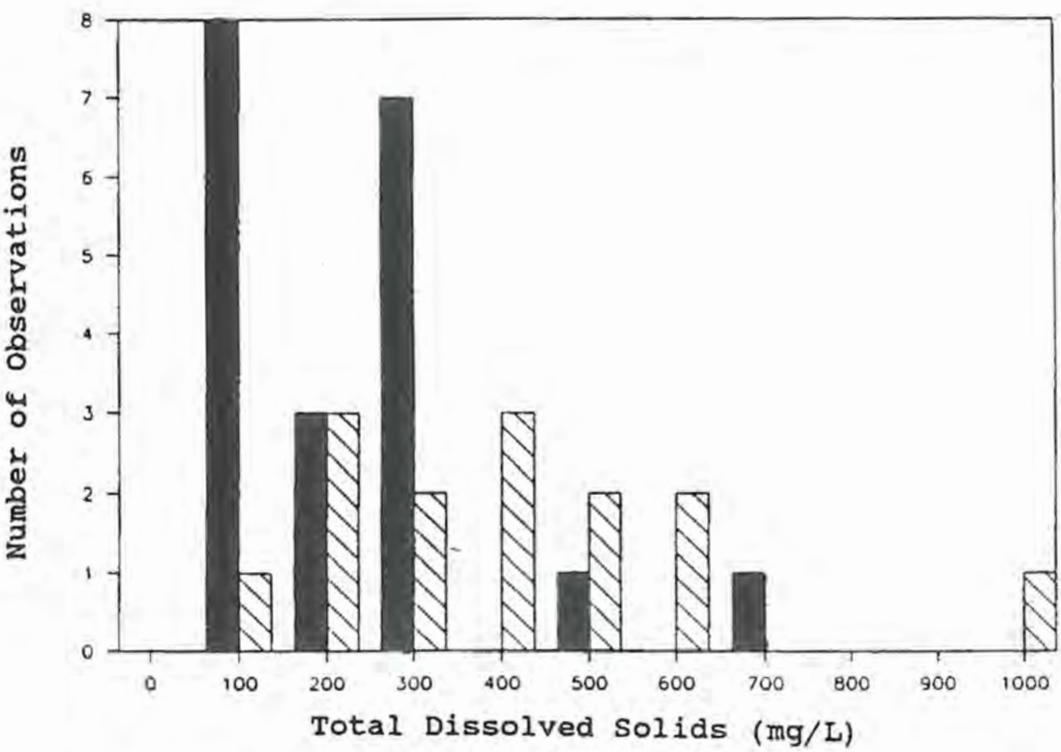


Figure 2: Distribution of total dissolved solids in Fort Payne well water (bold) and Ordovician well water (hatched).

## SUMMARY STATISTICS

The distribution of total dissolved solids (TDS) (figure 2) is grossly representative of the distributions of constituent ions and shows that both Fort Payne and Ordovician samples have a bimodal tendency. Overall, Fort Payne waters are lognormally distributed, and Ordovician waters are normally distributed. The ranges and means of well depths, ground water depths, ionic concentrations, and saturation indices are summarized in Table 1. The Mann-Whitney U-test indicated that the differences in the means of all chemical parameters except manganese and nitrate plus nitrite in Fort Payne and Ordovician well waters were significantly ($\alpha = 0.01$) different.

Table 1: Ranges and means of measured parameters. Fort Payne well-water chemical means are geometric means; all other means are arithmetic means.

| PARAMETER | WELLS | | | | SPRINGS | | | |
|---|---|---|---|---|---|---|---|---|
| | FORT PAYNE | | ORDOVICIAN | | FORT PAYNE | | ORDOVICIAN | |
| | RANGE | MEAN | RANGE | MEAN | RANGE | MEAN | RANGE | MEAN |
| Well Depth | 60 -- 175 | 106 | 94 -- 300 | 161 | | | | |
| Ground Water Depth | 15 -- 140 | 75 | 28 -- 170 | 103 | | | | |
| Calcium | 3.7 -- 83.3 | 17.5 | 15.1 -- 97.7 | 58.6 | 4.5 -- 28.5 | 18 | 6.8 -- 50 | 28.9 |
| Magnesium | 0.9 -- 39 | 6.5 | 3.1 -- 31.1 | 11.4 | 2.7 -- 9.2 | 6.4 | 2.8 -- 9.3 | 5.6 |
| Sodium | 0.6 -- 16 | 2.3 | 1.2 -- 214 | 21.4 | 0.7 -- 5.4 | 1.6 | 0.9 -- 2.4 | 1.7 |
| Potassium | <0.1 -- 1.9 | 0.4 | 0.2 -- 11.6 | 2.1 | <0.1 -- 1.9 | 0.7 | <0.1 -- 1.3 | 0.5 |
| Iron | <0.005 -- 4.54 | 0.053 | <0.003 -- 0.958 | 0.147 | <0.003 -- 0.197 | 0.076 | <0.003 -- 0.094 | 0.037 |
| Manganese | <0.005 -- 0.586 | 0.017 | <0.005 -- 0.178 | 0.027 | <0.005 -- 0.054 | 0.016 | <0.005 -- 0.014 | 0.002 |
| Alkalinity (as $HCO_3^-$) | 2.5 -- 451.9 | 68.6 | 15.1 -- 381.2 | 184.7 | 33.9 -- 123.6 | 83.9 | 28.4 -- 155 | 103.1 |
| Chloride | 0.5 -- 30 | 3.2 | 2 -- 27.2 | 27.9 | 1 -- 7 | 2.7 | 1 -- 10.9 | 4 |
| Sulfate | <0.1 -- 12 | 1.4 | 5.3 -- 284 | 46.4 | <0.1 -- 9 | 3 | 1 -- 16.3 | 7 |
| Nitrate-Nitrite | <0.1 -- 31 | 1.2 | <0.1 -- 8.1 | 2.8 | <0.1 -- 2.9 | 1.4 | <0.1 -- 3.3 | 1 |
| Total Dissolved Solids | 27.3 -- 600.8 | 120.2 | 92.2 -- 1008. | 357.3 | 50.2 -- 170.1 | 117.8 | 53.2 -- 239.1 | 151.9 |
| log $pCO_2$ | -3 -- -0.1 | -1.9 | -2.4 -- -1.2 | -2 | | | | |
| SI calcite | -5.2 -- 0.5 | -1.9 | -2.8 -- 0.3 | -0.3 | | | | |
| SI dolomite | -10.2 -- 0.9 | -4 | -6 -- 0.1 | -1.1 | | | | |
| SI gypsum | -4.5 -- -2.6 | -3.6 | -2.8 -- -1.7 | -2.1 | | | | |

## DISCUSSION

A plot of the molar concentrations of calcium and magnesium (figure 3) shows that the Ca/Mg ratio varies distinctively between Fort Payne and Ordovician waters. Excepting Fort Payne water samples 54, 58, and 64, all Fort Payne water compositions plot along a highly significant ($R^2 = 0.96$) trend with an average Ca/Mg of 1.4. The confidence of the empirical relation between Ca and Mg in Ordovician strata is lower but still significant ($R^2 = 0.66$), and the mean Ca/Mg ratio is 3.5. The different Ca/Mg ratios imply that dolomite is an important facies of the Fort Payne rock in most of the study area. Milici (1979) reports that dolomite is characteristic of the Fort Payne Formation. Samples 54, 58, and 64, which were collected in the central part of the Highland Rim as opposed to the western edge and farther south than the other samples plotted, suggest that dolomite is not a ubiquitous constituent of the Fort Payne. Correspondingly, the higher Ca/Mg ratios of Ordovician waters imply that dolomite is not a significant component of the upper Ordovician carbonates of the study area. Although not shown, spring water concentrations revealed Ca/Mg trends identical in orientation to the well water compositions, though spring waters had lower ionic strengths.

Despite the fact that Fort Payne Ca/Mg ratios indicate that dolomite is important in the Mississippian section, Ordovician waters are on the average 800 times closer to saturation with respect to dolomite than are Fort Payne waters. This is evidence that the Fort Payne ground waters sampled move in a system sufficiently open that little water/wall rock interaction occurs during its subsurface residence. Conversely, Ordovician water chemistry implies close contact between ground water and wall rock, resulting in the closer approach to equilibrium. The relative locations of individual water compositions along Fort Payne and Ordovician trends are indicators of the relative degrees of aqueous/wall rock interaction. Low ionic strength waters plotting near the origin reflect less aqueous/wall rock interaction than the more evolved water compositions plotting farther from the origin. The Fort Payne water samples 23, 27, 28, 29, 33 and 54 form a group of very low ionic strength waters. These

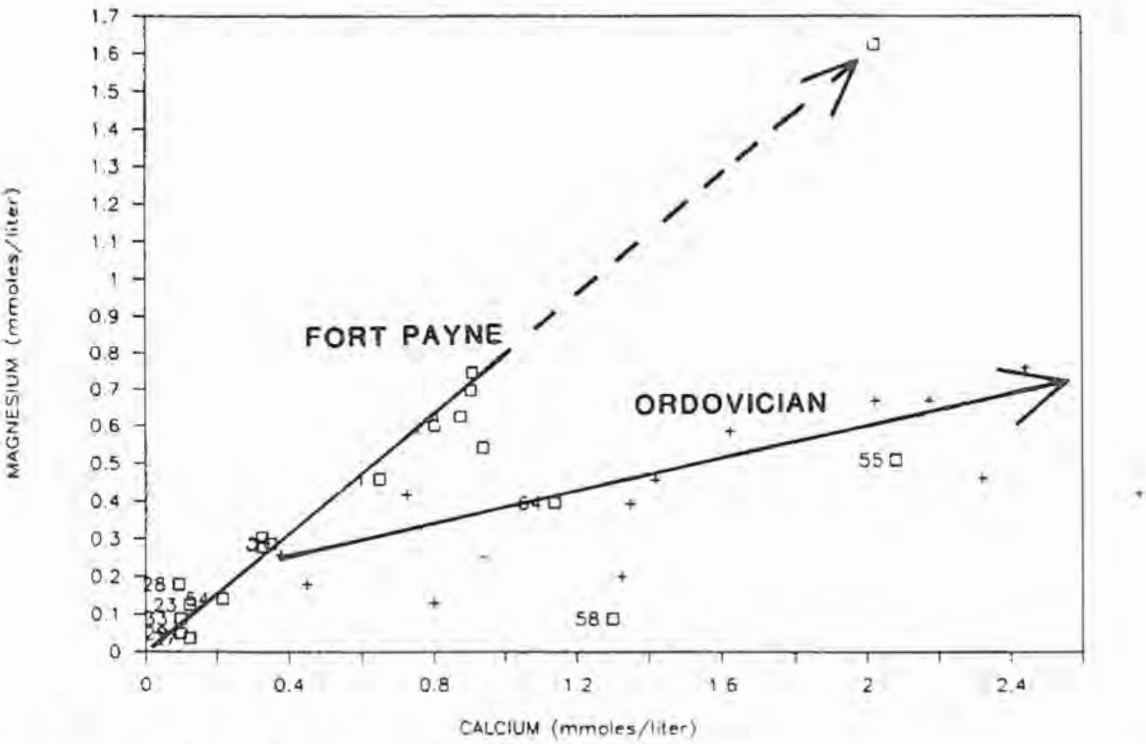


Figure 3: Plot of molar calcium and magnesium concentrations. Arrows indicate direction of increasing total dissolved solids concentration.

samples constitute the low-TDS mode shown in figure 2. The chemical behavior of these waters indicates that they were recharged through nearby, open fracture systems. All of these samples were collected from a water-table aquifer perched upon the Chattanooga Shale along the western margin of the Eastern Highland Rim.

The relation between calcite and gypsum saturation indices is shown in figure 4. Both Fort Payne and Ordovician trends have similar slopes, which may indicate either: 1) sulfate sources are more abundant within the Ordovician section, 2) ground-water flow is through a more open fracture system within the Fort Payne section, or both. Because evaporite sequences have not been reported from either lithology within central Tennessee, it is probable that the occurrence of sulfate minerals in either unit is limited to the disseminated grains. The locations of the low-ionic strength Fort Payne well waters are shown in figure 4 to indicate that these waters are highly undersaturated with respect to both gypsum and calcite. This is thought to result from nearby recharge through relatively open fracture systems that promote a low degree of aqueous/wall rock interaction.

Fort Payne and Ordovician well-water compositions were plotted on a Piper diagram (figure 5), which revealed further chemical nuances between the waters, particularly with respect to the activity of sodium. Trends 1 and 2 in figure 5 are formed by Ordovician well-water compositions, and trend 3 is formed by Fort Payne water compositions. Trend 1 consists of three sodium-chloride waters and is widely separated from the Ordovician calcium-magnesium bicarbonate waters of trend 2. Piper (1932) concluded that brines in the Central Basin subsurface

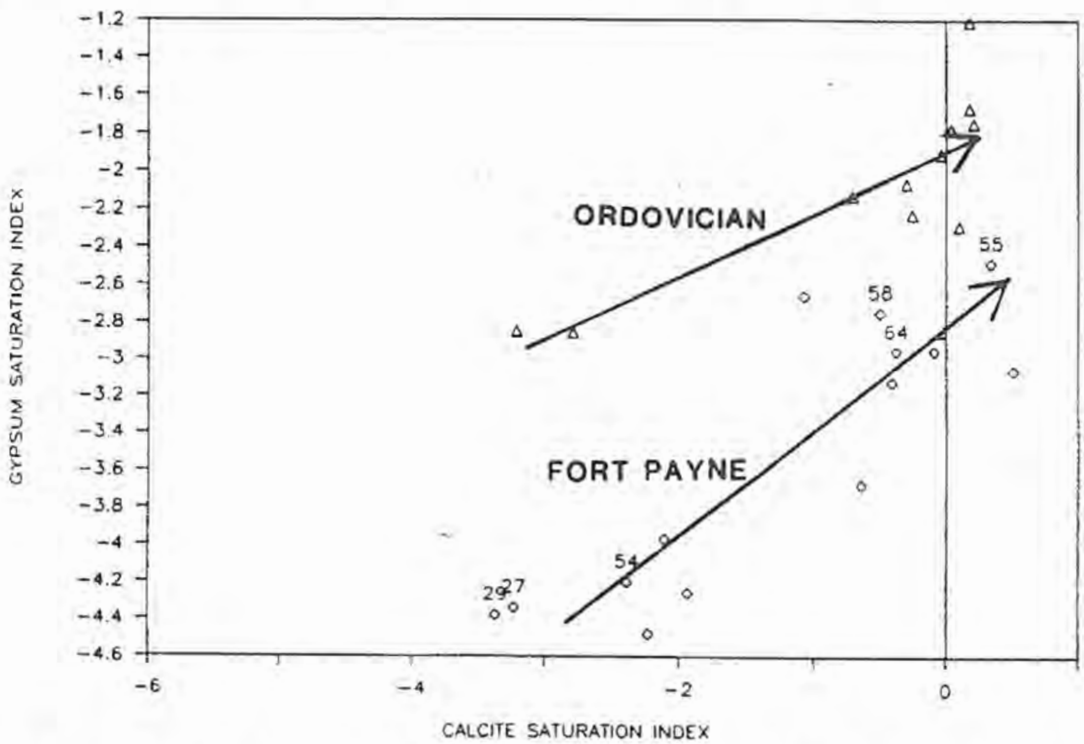


Figure 4: Plot of calcite and gypsum saturation indices. Arrows increase direction of increasing saturation.

represent formation waters trapped since deposition by confining clay layers. Trend 1 may reflect that mixing of deep connate brines with near surface ground waters and reaction of the resulting mixture waters with wall rock may be the source of sodium-chloride type Ordovician ground water. Other than the waters associated with trend 1, all other Ordovician waters (trend 2) had negligible sodium activity and a chemical matrix predominated by calcium, magnesium, and bicarbonate. Fort Payne waters (trend 3) plot between Ordovician trends 1 and 2 and reflect a sodium activity intermediate between the two Ordovician trends. The location of low ionic strength Fort Payne waters relative to other Fort Payne well waters indicates that the proportional importance of sodium is restricted to the higher ionic strength waters. The behavior of sodium in Fort Payne waters is possibly the result of the interaction of ground water with interbedded shales

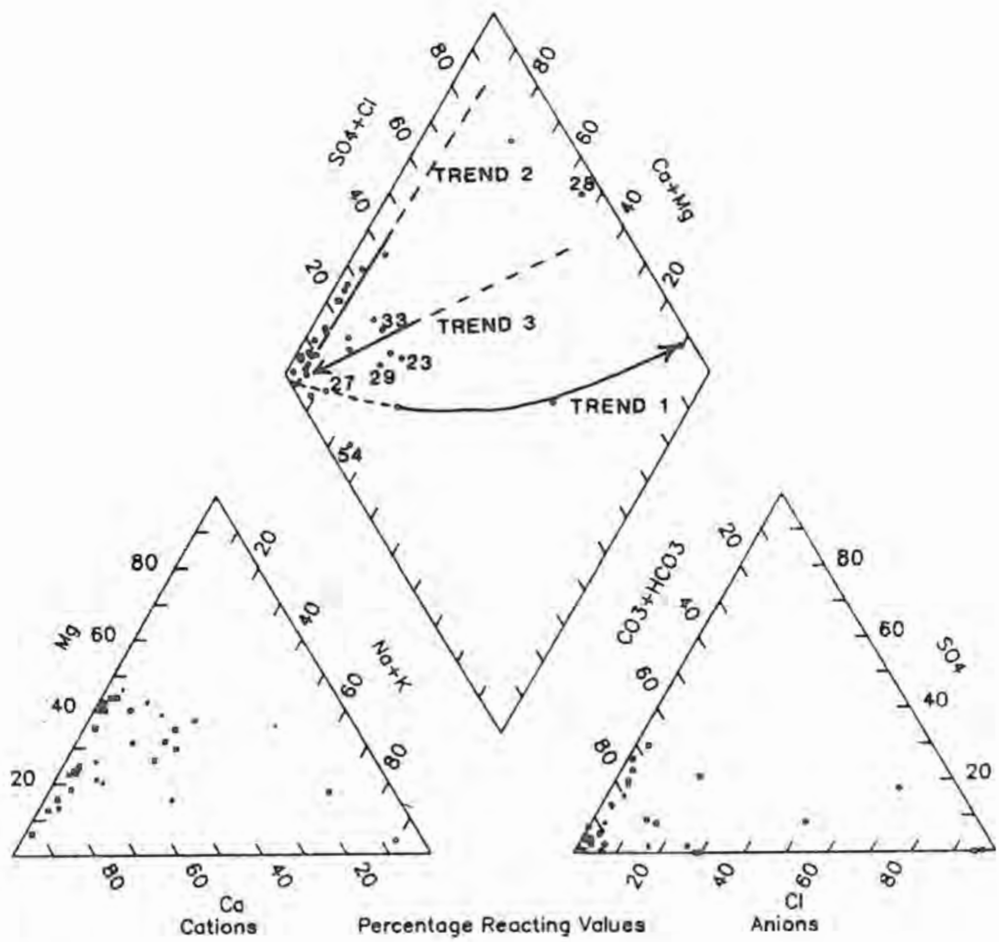


Figure 5: Piper diagram showing location of ordovician well-water trends 1 and 2 and Fort Payne well-water trend 3. Arrows indicate direction of increasing ionic strength.

and disseminated clay minerals. The relatively higher adsorptive strength of Ca++ and Mg++ over Na+ may promote ion exchange in clay-rich horizons of the Fort Payne aquifer. In this manner, Na+ activity in the water may have increased at the expense of Ca++ and Mg++ activity. However, shales are important components of the Ordovician section as well. The fact that similar ion-exchange reactions have not caused an increased sodium activity in ground water in Ordovician rocks may be evidence of dissimilar clay mineralogy. Alternately, this observation could reflect that clays in the Ordovician section have already been reverted to a calcium form as a result of a greater degree of aqueous/wall rock interactions.

## REFERENCES

Collar, P. D., 1989. Radon in Homes, Soils, and Caves and Water Quality of Wells and Springs of North Central Tennessee. M.S. Thesis, Tennessee Technological University, 246 pages.

Conant, L. C. and V. E. Swanson, 1961. Chattanooga Shale and Related Rocks of Central Tennessee. U.S. Geological Survey Professional Paper 357.

Glover, L., 1959. Stratigraphy and Uranium Content of the Chattanooga Shale in Northeastern Alabama, Northwestern Georgia, and Eastern Tennessee. U.S. Geological Survey Bulletin 1087-E.

Miller, R. A., 1979. The Geologic History of Tennessee; Department of Conservation, Division of Geology, Bulletin 74.

Milici, R. C., G. Briggs, L. M. Knox, P. D. Sitterly and A. T. Statler, 1979. The Mississippian and Pennsylvanian Systems in the United States--Tennessee. U.S. Geological Survey Professional Paper 110-G.

Piper, A. M., 1932. Ground Water in North-Central Tennessee. U.S. Geological Survey Water-Supply Paper 640, 238 pages.

Piper, A.M., 1944. A Graphic Procedure in the Geochemical Interpretation of Water Analyses. American Geophysical Union Transactions, 25: 914-923.

Reesman, A.L. and A. E. Godfrey, 1972. Chemical Erosion and Denudation Rates in Central Tennessee. Tennessee Department of Conservation, Division of Water Resources, Series 4, 35 pages.

## UTILITY OF A GIS AS A GROUNDWATER MANAGEMENT TOOL

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Groundwater management in West Tennessee is a growing concern to urban and rural citizens alike. Over 90 % of the water used in West Tennessee is groundwater. The vast majority of the 90% comes from one aquifer. To effectively manage the quality and the impacts of man's activities on the aquifer, access to large volumes of inter-related data is a necessity. The utility of a GIS to perform this data archival/retrieval/analysis function was explored for the Memphis/ Shelby County region of West Tennessee. Utilizing data available from published sources and the water utilities in Shelby County, a GIS was established which contains the location of all known wells (municipal, private, commercial), landfills and dumps, industries with high potential for releasing contaminants, suspected areas of shallow/deep aquifer interchange, and known areas of aquifer contamination. Utilizing the overlaying capability of the GIS, the potential of aquifer contamination and possible water treatment alternatives were developed for the MLGW service system. The utility of this system to serve the regional groundwater quality control board will be discussed.

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# GROUNDWATER FLOW SYSTEM STUDIES AT THE OAK RIDGE Y-12 PLANT[1] USING MULTI-PORT MONITORING SYSTEMS INSTALLED IN COREHOLES

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As part of the Oak Ridge Y-12 Plant Environmental Surveillance Activities, five existing deep coreholes were instrumented with multi-port (MP) systems during March, 1990. Data from the MP systems will include detailed pressure profiles, groundwater chemistry from samples obtained under in situ conditions, and hydraulic conductivity values for each monitored zone. The data will be used to better understand the regional groundwater flow system so that potential flowpaths of contaminants from numerous waste sites at the plant can be identified. The Y-12 plant is located in the Alleghanian foreland fold and thrust belt, and the stratigraphy of the area is characterized by shale and limestone, which are interlayered at all scales, and by massive dolomite. Because of the complex geologic setting, the groundwater hydrology of the area is correspondingly complex.

The coreholes range in depth from 600-1275 feet. Four of the coreholes are along a line traversing Bear Creek Valley and the northern flank of Chestnut Ridge near the western end of the Y-12 plant; the fifth corehole is located along strike on Chestnut Ridge at the eastern end of the plant. Subsequent to well construction and prior to installation of the MP system (a period of four and a half years) the coreholes were geophysically logged, and straddle packer tests (King and Haase, 1988) were conducted to determine local hydraulic conductivities and head values. These data, as well as the core samples, were used to help select the MP system sampling zones.

Geophysical logs were used primarily to identify fractures and fracture zones. In general, the caliper and temperature log deflections were most useful for identifying fractures, and supporting evidence was collected from spontaneous potential (SP), long-short normal (LSN), single-point resistance (SPR), variable density (VDL), and neutron log anomalies. In addition, the electric logs (LSN and SPR) together with results from the straddle packer testing were used to estimate vertical transitions in flow systems.

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On the basis of geophysical logs and packer testing results, 7 to 10 monitoring intervals, 20 to 25 feet long, were selected in each well. Each monitoring interval is isolated by double packers that form a guard zone. Both hydraulic head and hydrochemical data will be obtained from the monitoring intervals, but only hydraulic head data will be obtained from the guard zones. The monitoring intervals were chosen to sample several features; 1) fractures/ fracture zones; 2) apparent vertical transitions between flow systems; 3) stratigraphic overlap between coreholes, primarily in a strike direction; and 4) representative zones that show none of the above.

Two wells (GW-131 and GW-135) are located along geologic strike; each intersects roughly the same stratigraphy of the Copper Ridge Dolomite and the Maynardville Limestone. Therefore, an effort was made to sample similar stratigraphic horizons so that along-strike variations in flow systems could be investigated. In addition, the SPR log for GW-131 shows several baseline changes that may represent changes in the groundwater electrical properties, and also may represent a vertical stratification of groundwater flow systems. The upper zones in GW-135 and GW-131 were picked to characterize features of this possible vertical stratification.

Currently, two sets of hydraulic head measurements have been collected from the MP systems; the first was collected within a week of instrumenting the wells, and the second was collected approximately one month later (Figure 1). Most local head anomalies appear to be diminishing in magnitude with time and probably are associated with gradual dissipation of packer squeeze effects. However, anomalous head levels in GW-135 at an approximate elevation of 0 ft (Figure 1a) and a broad elevated head profile in GW-134 (not shown) are persistent in both sampling periods. Other pressure trends suggest that lithologies, in part, control hydraulic head distributions in the wells. For example, the low head values observed in GW-135 (Figure 1a) between elevations of 25 - 475' correspond to the Maynardville Limestone, parts of which are an important flowpath for groundwater (Geraghty & Miller, Inc. 1990).

The pressure profile for GW-132 shows an abrupt change at approximately 575' elevation (Figure 1b), below which the head values rise rapidly in a stepped fashion. The origin of the stepped head behvior is not apparent; however the change at 575' corresponds to the transition between the upper and lower Pumpkin Valley Shale, as identified in core. This change also occurs at a relative resistivity change observed in the LSN log; below 575', the long-normal log is less resistant than the short-normal log, possibly reflecting a change to more saline formation waters. These pressure, core, and geophysical log characteristics are preliminarily interpreted to reflect very low hydraulic conductivities, flowpaths primarily down dip along the stratigraphy, long groundwater residence times, and possible recharge from the southern flank of Pine Ridge.

Currently, interpretations of pressure profiles across Bear Creek Valley may indicate a floor to the hydrologic system, which shows no flow from Bear Creek Valley to the south underneath Chestnut Ridge. This interpretation has also been presented in King and Haase, 1988. Within this system, the Maynardville Limestone shows a strong influence on head values. Hydraulic head values for the Maynardville Limestone are relatively low and appear to remain fairly constant in both vertical and dip directions, suggesting a dominant lateral flow in this formation.

No water samples have been collected to date. At present, the zones are being purged to remove stagnant water from monitoried intervals. However, purging has proved to be difficult because of the extreme low hydraulic conductivity of the units. In addition, it has not been determined if the effects of four years of borehole circulation can be completely removed from these zones.

## REFERENCES


Geraghty and Miller, Inc., 1990. Groundwater Quality Assessment for the Bear Creek Hydrogeologic Regime at the Y-12 Plant, 1989. Y/SUB/90-00206C/1, Oak Ridge, Tennessee.

King, H. L. and C. S. Haase, 1988. Summary of Results and Preliminary Interpretation of Hydrogeologic Packer Testing in Core Holes GW-131 Through GW-135 and CH-157, Oak Ridge Y-12 Plant. Y/TS-495. 49pp.

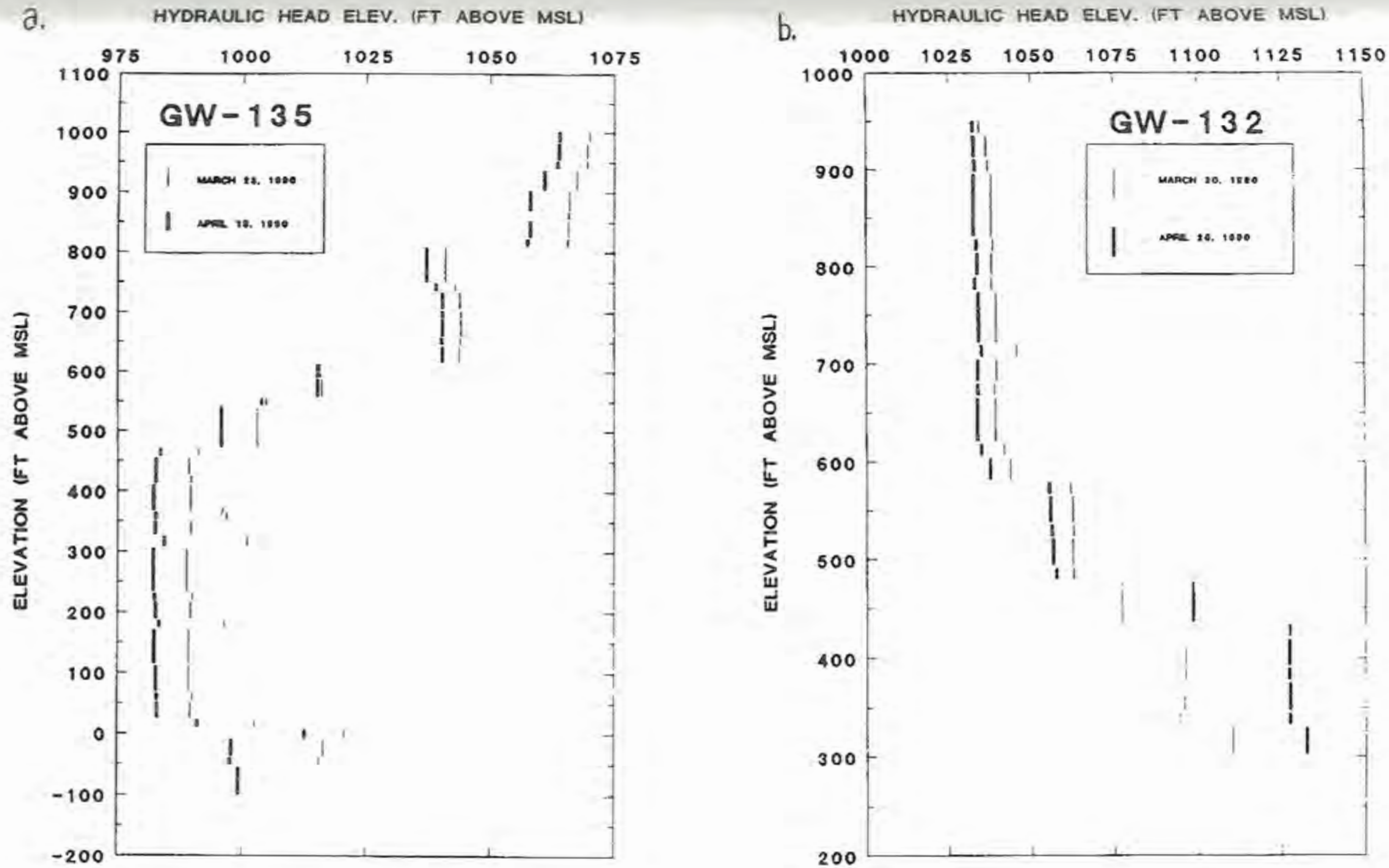


Figure 1. Hydraulic Head - Elevation Profiles for GW-135 (a) and GW-132 (b). Both boreholes are part of a corehole transect across Bear Creek Valley in the vicinity of the S-3 Ponds. GW-135 is at the southern end of the transect and intersects the Copper Ridge Dolomite and the Maynardville Limestone; GW-132 is at the northern end of the transect and intersects (from top to bottom) the Rogersville Shale, Rutledge Limestone, Pumpkin Valley Shale and the Rome Formation.

# AN OVERVIEW OF SUBSURFACE TRANSPORT STUDIES IN THE VADOSE ZONE

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In no other part of the United States is the erosion potential worse than in the western parts of Tennessee. Because of the severe erosion hazard, no-tillage and other methods of conservation tillage have been adopted by many farmers. The no-tillage system has proven its effectiveness in reducing overland flow and sedimentation. However, the added water entering the soil and the lower evaporation rate from the soil surface lead to a higher leaching potential under no-tillage. Because soil is left practically undisturbed under no-till, there is a tendency for macropores (cracks between aggregates and biological pores) to form and preferential flow of water and nutrients to occur through continuous macropores towards the groundwater (Thomas et al., 1973; Edwards et al., 1988). Conventional tillage reduces the macroporosity and shifts the pore size distribution to greater microporosity. Ankeny et al. (1990) showed that as soil desaturates, the infiltration rate under no-tillage decreases faster than under conventional tillage due to this shift in pore size distribution.

During storm events, it would be reasonable to expect greater leaching of solutes through no-tillage soils, both because of greater macroporosity and because of lower evaporation promoting wetter antecedent conditions. This combination greatly enhances the potential for rapid solute fluxes through preferential paths, bypassing or short circuiting the retentive and biologically active soil zone. There may be a tradeoff between the contamination of surface and contamination of groundwater supplies unless methods of manipulating the soil hydrologic conditions to prevent short-circuiting can be developed for no-tillage. However, practically no information is available comparing subsurface transport of chemicals from no tillage and conventional tillage under field-scale conditions except for continuous corn (Tyler and Thomas, 1977). Other cropping systems require research on transport of nitrate and pesticides in order to better manage them for minimum groundwater contamination.

## PROCEDURES

### In-Situ Transport Experiments

The effects of cropping systems and tillage practices on nitrate movement is being evaluated under four experimental designs. Tension-free pan lysimeters are used to collect soil water draining from the soil profile of each

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tillage/cropping system treatment. Three size lysimeter pans were constructed; 60 x 76 cm, 45 x 35 cm, and 13 x 30 cm pans. Six plots of no-tilled versus conventionally tilled tomatoes with winter wheat cover is being studied for surface and subsurface transport of chemicals. Each plots was instrumented with a medium sized lysimeter, and two of these plots also contained a large and four replicates of the small lysimeters. Four plots of no-tillage versus conventional tillage cotton with winter wheat cover have been instrumented with a large lysimeter. Eight plots of soybean-wheat-corn rotations under no-tillage and various conventional tillage practices have been instrumented with large lysimeters. Six plots of no-tillage versus conventional tillage corn with various cover crops and nitrogen fertilization rates have been instrumented with large lysimeters. In addition, a medium and four replicates of the small lysimeters were installed in two of these plots. A total of forty-four lysimeters have been installed.

Each lysimeter is filled with sterilized sand and crushed granite to establish continuity with the soil profile. At 91 cm depth, the soil was excavated laterally from a trench adjacent to each plot. A lysimeter was inserted into the excavated area and high pressure tubing was connected to route water collected into a buried 60 L polypropylene carboy. Following storm events, all water in each carboy is siphoned out for analysis of nitrate concentration and the quantity of subsurface flow recorded. Samples are filtered with 0.45 μm Nuclepore polycarbonate filters immediately after sampling. Nitrate is determined by Ion Chromatograph (IC) analysis.

## Surface Hydraulic Characterization

To better understand the effects of tillage and cropping systems on the transport of nitrate, infiltration experiments are being conducted to quantify the degree of preferential flow. The effects of long-term soil physical changes as a result of no-tillage is being compared to that of conventional tillage practices. Double ring and tension infiltrometer (Wilson and Luxmoore, 1988) measurements are being conducted on the soil surface. The steady-state infiltration rate is determined as a function of soil water potential from saturation to -3.0 kPa of matric potential. Estimates of the hydrologically active macroporosity and the effective pore surface area will be made from these measurements.

## Undisturbed Soil Column Experiments

The effect of the incorporation of straw from wheat cover crop on transport of Imazaquin is being studied. Six large (0.3 m diameter x .61 m long) undisturbed soil columns were extracted and wheat straw mulch was incorporated at 0 and 4480 kg $ha^{-1}$. Steady state unsaturated flow was established and each soil system was made monoionic by infiltrating several pore volumes of 0.1M KCl. The ionic strength was lowered by following this application with several pore volumes of 0.01M KCl. A pulse of 0.25 pore volume of 0.14 kg ai $ha^{-1}$ and 0.1M KBr was applied and effluent collected. Bromide was determined by ion chromatography and ion specific electrodes and imazaquin determined by HPLC.

## PRELIMINARY RESULTS

### In-Situ Transport Experiments

Tension-free lysimeters collected saturated flow through preferential flow paths even when the soil matrix was unsaturated. Such nonequilibrium conditions are prevalent during storm events and may be the predominant mechanism of subsurface transport towards groundwater. The collection area of the large lysimeters were considered to have effectively integrated the heterogeneity in flow paths, thus yielding a representative sample of the nitrate transport within each plot. However, significant differences were observed in the subsurface flows collected by the four replicates of small lysimeters. These differences are believed to represent spatial heterogeneity in preferential flow paths.

### Surface Hydraulic Characterization

Steady-state infiltration rates were determined on 3 no-tillage plots at 0, 30 150 and 300 mm matric potential and 3 conventional tillage plots at 0, 30, and 300 mm matric potential. The no-tillage plots consistently exhibited approximately one order of magnitude greater infiltration rates than the conventional tillage plots (Fig. 1). A shift in pore size distribution to lower hydrologically active macroporosity with conventional tillage was observed in these plots (Table 1). Destruction of soil structure by tillage dramatically decreases the number of macropores and also interrupts the pore continuity. This not only limits the ability of pores to transport water but also enhances the entrapment of air further limiting infiltration. These processes lead to greater surface runoff during high intensity rainfall events from conventionally tilled soils. However, the shift to smaller pores has been reported (Ankeny, et al., 1990) to potentially result in greater infiltration capacity in conventional tillage at low matric potential such as might occur with low rainfall or irrigation events during the growing season. This transition from greater infiltration rates near saturation under no-tillage to greater infiltration rates at low matric potentials under conventional tillage was not observed in these data.

### Undisturbed Soil Column Experiments

The one dimensional convection-dispersion equation was used to model with CXTFIT (Parker and van Genuchten, 1984) the solute breakthrough curves (BTC) for the unsaturated soil columns. The retardation factor for $Br^-$ was assumed equal to one and the $Br^-$ BTC (Fig. 2) was modeled by fitting the dispersion coefficient. Low dispersion coefficients and considerable tailing of the BTC was observed, indicating a large degree of preferential flow. Simulations based on Kd values from the literature (Wolt et al., 1989) ranging from 0.30 to 0.001 were made to determine the necessary input conditions to observe imazaquin breakthrough. However, imazaquin was not detected above 20 $\mu g\ L^{-1}$ indicating a retardation factor considerably greater than reported elsewhere. The mechanism of the greater reactivity observed in this study is not certain but is believed to be an artifact of the experiment. The effect of continuous flow of high salt concentrations prior to the tracer test on pH, and redox conditions is suspected to have altered the soils reactivity with imazaquin.

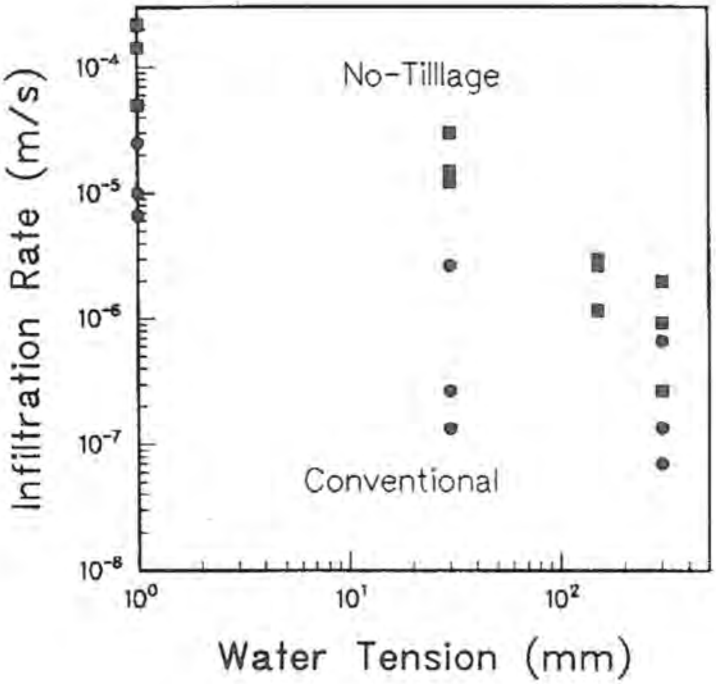


Figure 1: Ponded and tension infiltration rates for conventional (circles) and No-tillage (squares) soils.

Table 1: Average values for effective porosity, pores per unit area, and pore surface area for the hydrologically active pores.

| Infiltration tension range | Effective porosity ($m^3/m^3$) | Number of pores ($\#/m^2$) | Pore Surface area ($m^2/m^3$) |
|---|---|---|---|
| | **Conventional Tillage** | | |
| 0 - 30 | 0.00006 | 69 | 22 |
| 30 - 300 | 0.00030 | 39500 | 1240 |
| | **No-Tillage** | | |
| 0 - 30 | 0.00052 | 637 | 200 |
| 30 - 300 | 0.00460 | 587131 | 18400 |

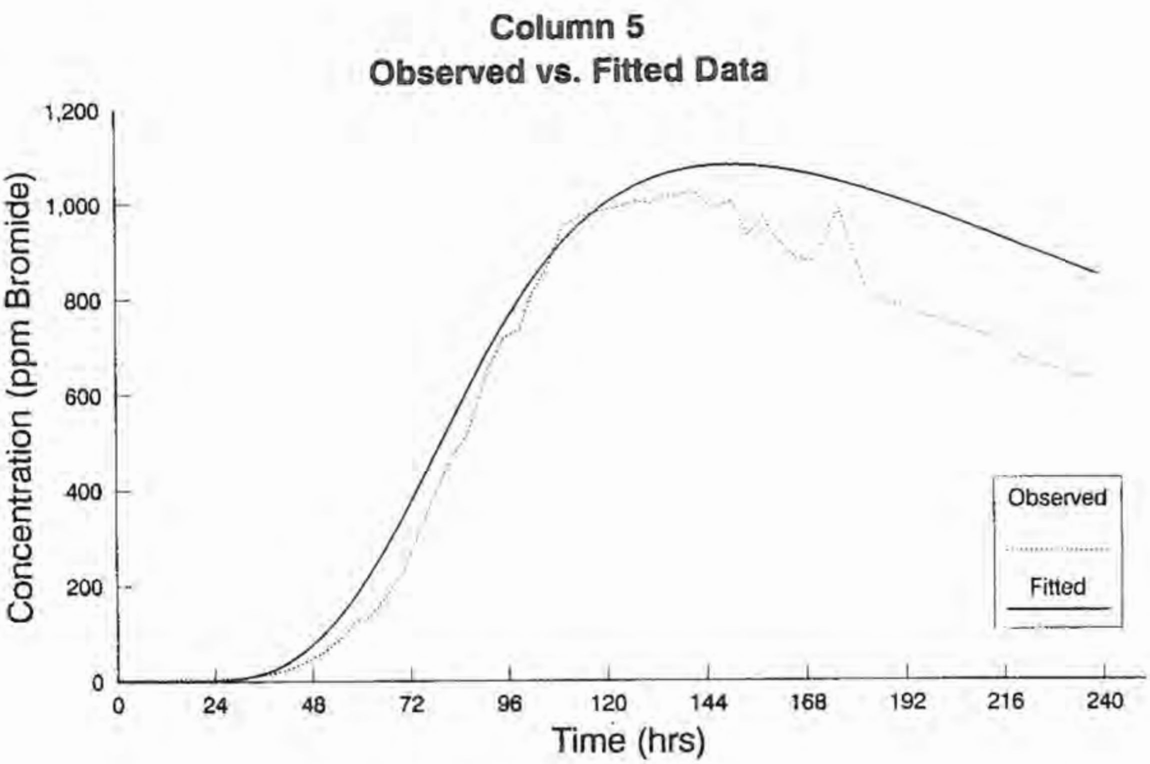


Figure 2: Observed (dashed) and fitted (solid) bromide BTC for an unsaturated soil column from a no-tillage plot.

REFERENCES

Ankeny, M. D., T. C. Kaspar, and R. Horton, 1990. Characterization of Tillage and Traffic Effects on Unconfined Infiltration Measurements. Soil Sci. Soc. Am. J. (in press).

Edwards, W. M., M. J. Shiptalo, and L. D. Norton, 1988. Contribution of Macroporosity to Infiltration into Continuous Corn No-tilled Watershed: Implications for Contaminant Movement. J. Contam. Hydrol., 3:193-205.

Luxmoore, R. J., P. M. Jardine, G. V. Wilson, J. R. Jones, and L. W. Zelazny, 1990. Physical and Chemical Controls of Preferred Path Flow Through a Forested Hillslope. Geoderma (in press).

Parker, J. C., and M. Th. van Genuchten, 1984. Determining Transport Parameters from Laboratory and Field Tracer Experiments. Virginia Agricultural Experiment Station, Bulletin 84-3, Blacksburg, Virginia.

Thomas, G. W., R. L. Blevins, R. E. Phillips, and M. A. McMahon, 1973. Effect of a Killed Sod Mulch on Nitrate Movement and Corn Yield. Agron. J., 65:736-739.

Tyler, D. D., and G. W. Thomas, 1977. Lysimeter Measurements of Nitrate and Chloride Losses from Soil Under Conventional and No-tillage Corn. J. Environ. Qual., 6:63-66.

Wilson, G. V., and R. J. Luxmoore, 1988. Infiltration, Macroporosity, and Mesoporosity Distributions on Two Forested Watersheds. Soil Sci. Soc. Am. J., 52:329-335.

Wolt, J. D., G. N. Rhodes, J. G. Graveel, E. M. Glosauer, M. K. Amin, and P. L. Church, 1989. Activity of Imazaquin in Soil Solution as Affected by Incorporated Wheat Straw. Weed Science, 37:254-258.

# EVALUATION OF BMP'S EFFECTS ON WATER QUALITY IMPROVEMENT USING HSPF

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## INTRODUCTION

The Hydrological Simulation Program in Fortran (HSPF) was applied to North Reelfoot Creek watershed above Highway 22, a 56.3 square-mile basin in northwest Tennessee characterized by severe cropland erosion problems. Soil erosion and sedimentation has adversely impacted water quality, most notably in the downstream Reelfoot lake.

Best management practices (BMP'S) have been widely implemented in the watershed, most since 1981. Although many BMP'S have been implemented, two (BMP # 1 and BMP # 9 ) have had the greatest effect in reducing cropland erosion. BMP # 1 is the establishment of permanent vegetative cover on farmland, while BMP # 9 involves the use of conservation tillage systems such as no-till, minimum till, contour farming, and other approved conservation tillage practices.

## MODELING APPROACH

Water quality and stream flow data obtained from the U.S. Geological Survey over a 54-month study period (April 1984 to September 1988) were used to calibrate and verify the model for the hydrologic/hydraulic and sediment washoff/transport processes. The dynamics of watershed land use throughout the entire simulation period presented a challenge to modeling strategy development. To solve this problem, the BMP data were evaluated on an annual basis so that appropriate adjustments in land use categories and/or calibration parameters could be made.

Simulation of the hydrologic/hydraulic and sediment washoff/transport processes was achieved by using two HSPF application modules, PERLND and RCHRES. Module PERLND simulates water quantity and quality processes occurring on pervious land segments, while module RCHRES simulates processes in a single open channel reach or a completely mixed lake. Calibration of the model was accomplished using data from April, 1984, through December, 1986, while data from January, 1987, through September, 1988, were used in model verification.

Following calibration and verification, the model was used to evaluate alternative scenarios involving hypothetical BMP implementation. Primary emphasis was placed on evaluating the effects of BMP's on water quality improvement or potential sediment loading reductions to Reelfoot Lake. Four

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alternative scenarios were examined under two annual sets of weather conditions, a wet year and a dry year. Scenario 1 assumed that all cropland in Area I (steep uplands) was taken out of production due to BMP # 1 conversion. Scenario 2 assumed that conservation tillage (BMP # 9 ) was employed on all cropland in Area II (fairly level uplands), in addition to cropland conversion throughout Area I. Scenario 3 reflected the construction of three sedimentation reservoirs in Area I, and Scenario 4 was a combination of Scenarios 2 and 3. Each scenario was compared to the base conditions, characterized by 1988 land use.

## RESULTS

Annual sediment reduction was substantial for all BMP scenarios. Figure 1 shows the annual sediment comparison of base conditions with BMP scenarios for both wet and dry weather cases. Percent sediment load reduction under wet year conditions, in increasing order, was as follows: 24% for scenario 3, 53% for scenario 1, 57% for scenario 2, and 67% for scenario 4. The corresponding percentages for a dry year were 33%, 61%, 66% and 74%. Based on these values, the effects of BMP's in reducing sediment loads were greater in a dry year than in a wet year. The largest decline in sediment load was observed in scenario 4, as expected because it included the combined effects of BMP #1, BMP # 9 and BMP # 12 (sediment retention basins). Of all the BMP's analyzed, cropland to grassland conversion alone appeared to have the highest sediment reduction impact. Dividing the sediment load reduction by 3470 acres of land taken out of production due to BMP # 1 gives approximately 10 tons/acre of sediment yield reduction. Since sediment yield was about one-fifth to one-third of the Universal Soil Loss Equation (USLE) erosion rates, gross erosion reduction then fell between 30 to 50 tons per acre of land converted to grassland.

The difference in percent sediment reduction for scenarios 1 and 2 was rather small; therefore, use of conservation tillage systems throughout Area II would not result in major water quality improvement in North Reelfoot Creek at Highway 22. This is not surprising for two reasons. First, much of the sediment coming form the fairly level uplands ( Area 11 ) is removed by the existing reservoir # 10 on the main channel. Second, about 0.5 tons of sediment yield reduction occurred at Highway 22 for each acre of Area II cropland receiving conservation tillage; accounting for the effects of reservoir # 10 and the difference between sediment yield and gross erosion, the estimated gross erosion reduction for BMP # 9 was 5.5 tons/acre, which is very realistic.

The impact of these alternative scenarios on runoff volumes was insignificant regardless of rainfall pattern even though annual runoff resulting form a dry year was reduced slightly more than that for a wet year. Figure 2 compares the yearly runoff volumes simulated for base conditions and different BMP scenarios. Runoff was essentially the same for wet year conditions, and was reduced by 3% in scenarios 2 and 3, 4% in scenario 1, and 6% in scenario 4 for a low rainfall year.

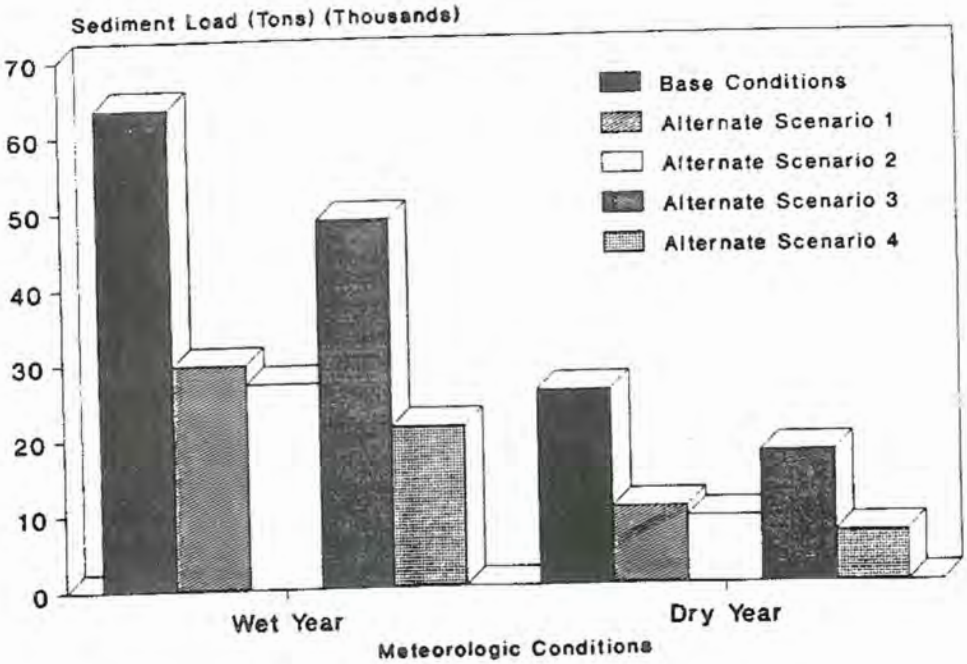


Figure 1: Annual Sediment Loads for BMP Scenarios.

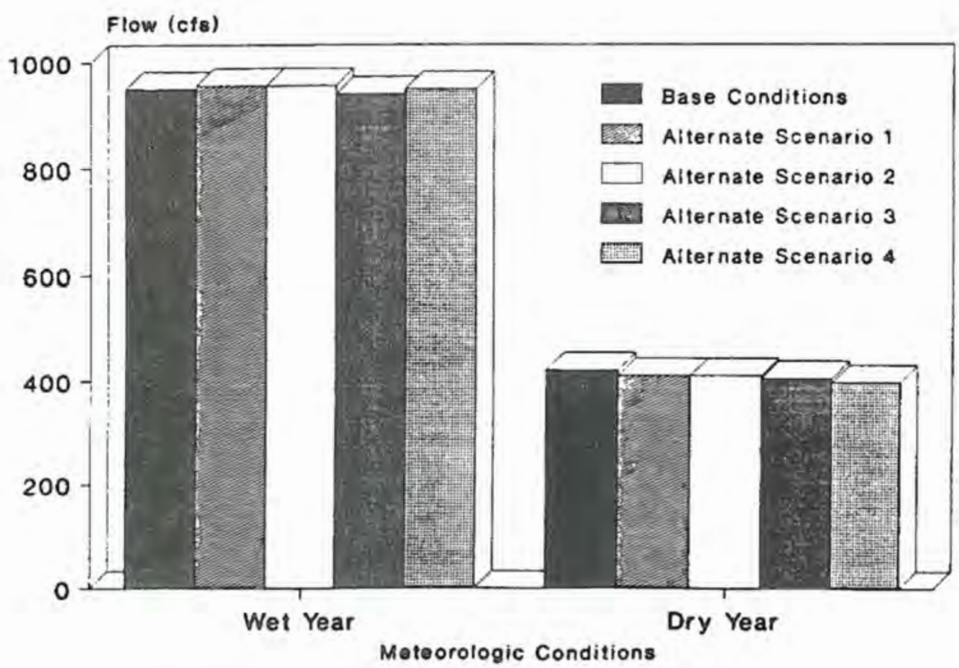


Figure 2: Annual Flows for BMP Scenarios

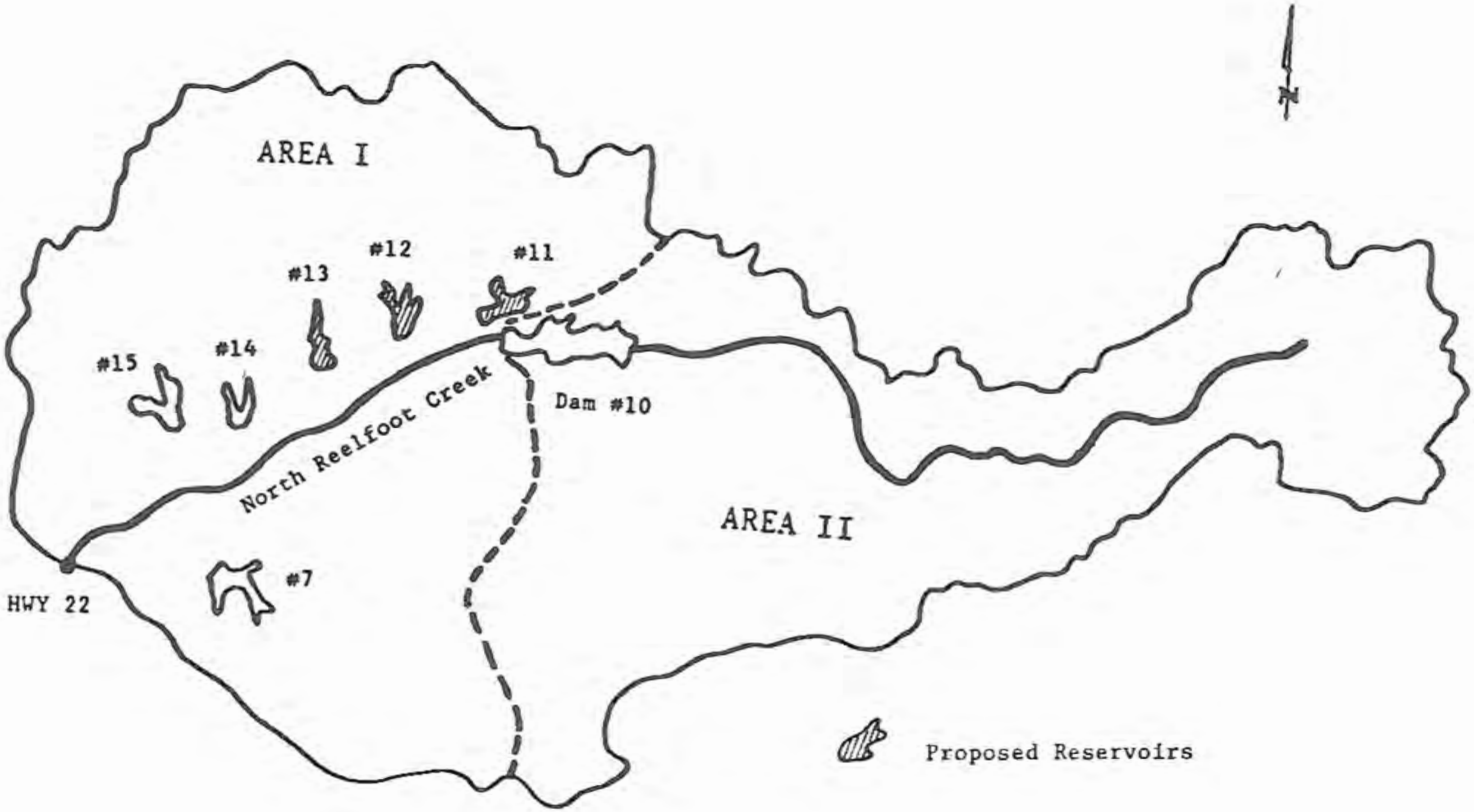


Figure 3. Location map of reservoirs and two distinct areas in watershed

# DETERMINATION OF MINIMUM OPERATING GUIDES FOR MULTIPURPOSE RESERVOIRS

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## INTRODUCTION

Competing demands for water resources require extensive analyses in planning and operating single reservoirs or systems of reservoirs which can be used to store and release water to maximize benefits. These demands, such as flood control, hydropower, minimum flow maintenance, and recreation often conflict with each other. Variations in seasonal and annual rainfall often makes it difficult to fully accomodate all objectives. A hydrological analysis technique is described in this paper which is useful in developing annual operating guides which ensure that various purposes can be served with a selected reliability.

Priorities are first placed on achieving four objectives, as listed above. Flood control is represented by a flood guide curve which denotes a seasonal operating level for a given reservoir. This determines the maximum planned water allowed in storage (except when regulating a flood event). A minimum flow requirement is assumed, which can vary seasonally but which must be met 100% of the time. The recreation objective is represented by requiring at least certain minimum reservoir elevations during the prime recreation season which must be achieved 90% of the time. Hydropower concerns are addressed by providing the maximum operating flexibility for hydropower operations, while still achieving the aforementioned objectives.

Historic flow records are used with a backward simulation model to determine the minimum seasonal storage which must be provided based on each year of the hydrologic record. These storage traces are then analyzed and the most binding 10% of the years are discarded. An envelope curve is then constructed based on the remaining years, which becomes the seasonal minimum operating guide.

The examples used in this paper reflect assumed priorities and reliabilities for various purposes. The methodology is general and can be used for different operating objectives as desired.

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## PURPOSE

The purpose of this paper is to describe a method which can be used to determine a minimum operating guide (MOG) for multipurpose reservoirs which ensures that operating objectives can be met with a prescribed reliability, while providing the maximum operating flexibility for the use of "discretionary" storage. Some basic assumptions are:

(1) Flood control storage requirements are seasonal in nature, with greater needs in winter, and less in summer and fall. This requirement is shown as the upper guide curve on Figure 1. Operation above the flood guide is not permitted except during flood events for regulation.

(2) There are one or more downstream flow requirements which vary seasonally. These requirements must be guaranteed with 100% reliability.

(3) Recreation/lake use functions require relatively high pool levels during the summer. To satisfy public pressure and encourage development, these levels must be provided with 90% reliability. Recreation pool level requirements are shown in Figure 1 for June and July.

Figure 1 also shows an idealized MOG. This curve provides operating guidance as to when releases from the reservoir must be curtailed to only meet minimum flow requirements to ensure achieving the objectives listed above. In Figure 1 there are three operating regions shown. In the area below the MOG, operation of the project must be such that releases can only be made to meet minimum flow requirements downstream, and no "extra" water can be released. In the area between the MOG and flood guide curve, (FGC), discretionary releases (i.e for hydropower or other purposes) can be made, subject to not violating flood control requirements or downstream minimum flow requirements. Above the FGC releases are scheduled to draw the reservoir back to the FGC subject to downstream flooding constraints. The following discussions describe the computation of the MOG.

## METHOD

The MOG is computed by performing a backward simulation of reservoir operation. The basic data requirements are few: reservoir inflows and the storage vs. elevation relationship for the reservoir. For our case studies, a computation time step of one week was used. Other time steps such as daily or monthly would also work, the basic tradeoff being the time required for the derivation of the inflows and computation time vs. the degree of detail provided in the results.

As with any type of hydrologic simulation model used in planning or operating studies, a long period of historical flows will provide a more accurate representation of the stochastic nature of the water supply, whether these flows are used directly in the simulation, or as a basis for determining statistical parameters for synthetic flow generation. For this application, an 86-year record of historical weekly inflows was used. The analysis method would be the same whether historical or synthetic flows were used.

The first step is to perform a backward simulation for the reservoir. In Figure 1, the desired summer recreation level (and thus storage) at the end July provides the starting point. Working backward one week from this point, the required storage (and thus pool level) one week prior to this date can be computed by a simple water balance:

$$STOR(I-1) = STOR(I) - INFL(I) + DISC(I) \qquad (1)$$

where:
STOR(I-1) = reservoir storage at the beginning of week I;
STOR(I) = reservoir storage at the end of week I;
INFL(I) = reservoir inflow for week I;
DISC(I) = required reservoir discharge for week I;

The beginning-of-week storage thus computed is the minimum storage which ensures that the flow requirement can be met and still attain the desired end-of-week storage. For the months of June and July, an additional requirement is present, that being that the beginning-of-week storage thus computed cannot be less than the storage corresponding to the minimum recreation level for that week. The backward simulation is performed each week of each year for which flow information is available. This produces N individual storage (elevation) traces, where N is the number of years of flow information available. Figure 2 shows a sampling of these traces for three different general hydrologic conditions. Curve D, the uppermost curve, depicts the results for a dry year. Notice that beginning in July and working backward, the curve extends upward back through the early spring. This is typical of what happens during weeks when the reservoir inflow is not enough to meet the downstream flow requirement. Water must be withdrawn from storage to augment the inflow, thus the storage requirement increases. The storage at any point on curve D shows the storage requirement which ensures meeting the flow requirement and being at the minimum desired pool levels in June and July. Curve N represents the elevation trace during a normal (or average) flow year. Less storage is required throughout the year, because the inflows for most weeks are sufficient to meet the downstream requirement. Curve W shows results for a wet year, when inflows are always enough to meet the flow requirement. This curve shows that the reservoir could be operated at low pool elevations throughout the fall, winter, and early spring and still be able to meet both the flow and summer pool level criteria.

To ensure meeting the flow and summer pool levels for the very dry years would require high pool levels be carried over during the entire year, because the operator would not know beforehand which sequence of flows would occur in a given year. However, flood control considerations require low pool levels in the winter, as shown by the FGC. Thus flood control and summer pool level criteria cannot both be guaranteed 100% of the time. The original criteria for summer pool levels was that these levels would be achieved 90% of the time. For an 86-year record, failures could be allowed in about 8 years. Thus, 8 individual flow traces could be discarded. It is obvious that to ensure flood control, any year whose trace is above the flood guide curve must be eliminated (the driest years). This is tantamount to saying that in those years, the summer pool levels will not be reached. If only three years, for example, are above the FGC, five additional years can be discarded. These years would be subjectively selected to produce the widest space between remaining years not selected and the FGC, to produce the largest "discretionary" operating space.

After the eight years are discarded, an envelope curve is drawn through the uppermost point of all remaining years. This envelope curve may be comprised of points from only one year, but more likely will contain points from many years. This curve defines the MOG. The operator can then use the MOG and FGC as described earlier, with the assurance that flood control, minimum flows, and summer pool levels will be achieved with at least the desired reliability, and that the discretionary operating zone is as wide as possible to provide the maximum flexibility for operating for other purposes.

## RESERVOIR SYSTEM ANALYSIS

The method described above used a single reservoir as an example. The same procedures can be used to analyze reservoir subsystems and systems. Consider a two-reservoir subsystem as shown in Figure 3. Reservoirs A and B both have individual FGC's, minimum flow requirements, and summer pool elevation criteria. There is also an additional downstream flow requirement which either, or both, reservoirs can help supply. The problem is to develop MOG's which ensure that all requirements are met. If the common flow constraint is less than the sum of the two individual requirements, there is no problem, since meeting the individual requirements ensures meeting the common constraint. The more interesting case is one where the common constraint is larger, therefore additional storage in reservoir A and/or B is required. In this case, each individual reservoir is analyzed as before, to construct a MOG for each individual reservoir's requirement. An additional MOG for the combined A and B subsystem is also constructed. For this analysis, a combined FGC is constructed, using the sum of the storages. Also, the required summer pool levels are converted to storage and summed, resulting in a combined storage requirement. The analysis then proceeds as before, with the combined reservoirs being treated as one reservoir. This produces a subsystem MOG, which the reservoir operator must also consider along with the individual MOG's when scheduling operations.

Various combinations of reservoirs within a system can be analyzed in a similar manner, with appropriate subsystems first being disaggregated for analysis to produce the MOG's and then the results being "layered" for use in operational scheduling of the system.

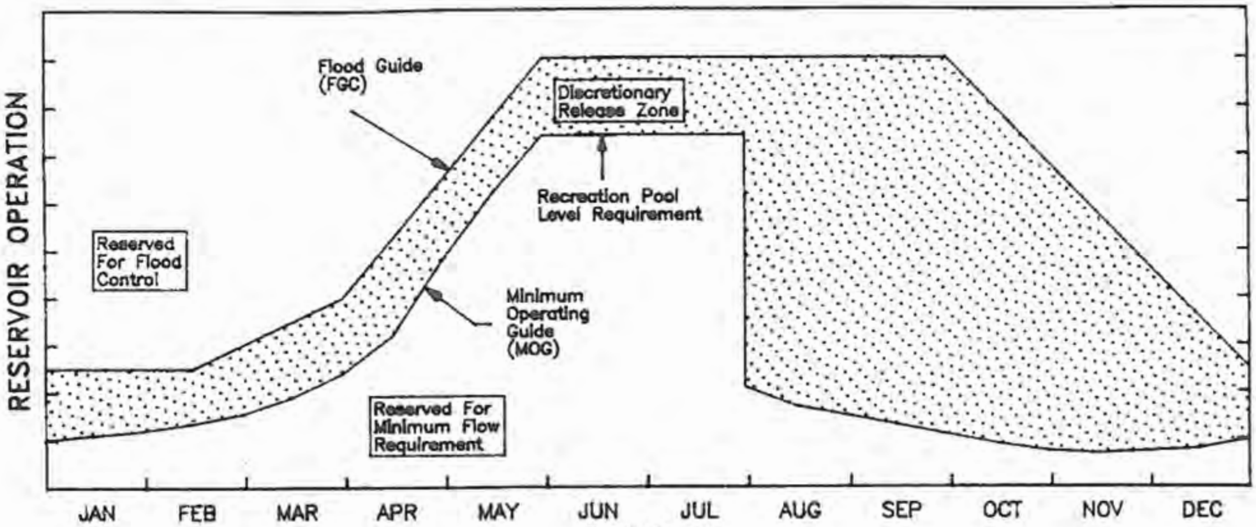


FIGURE 1

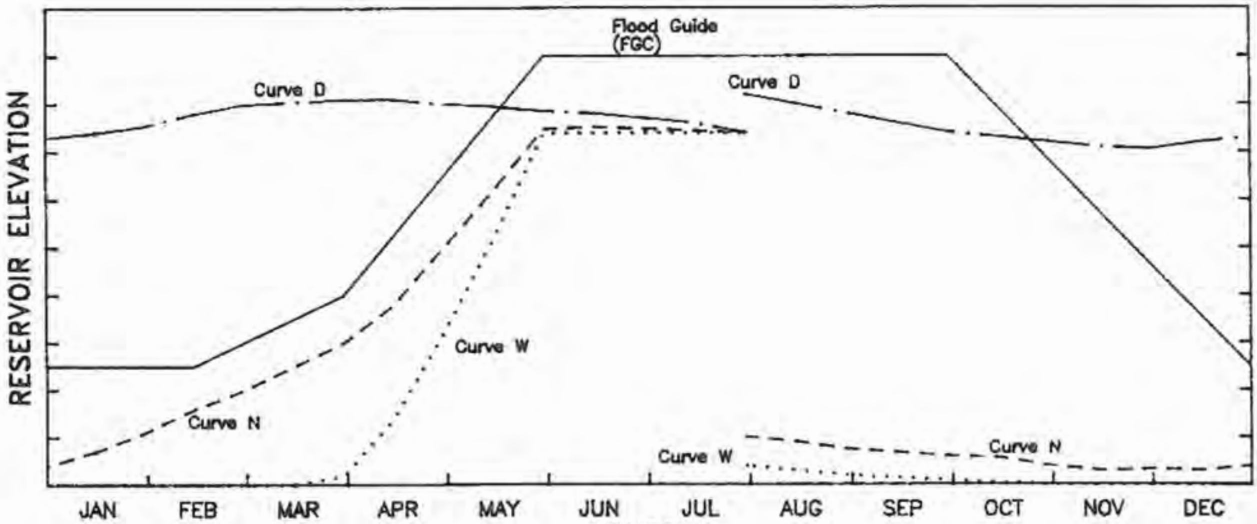


FIGURE 2

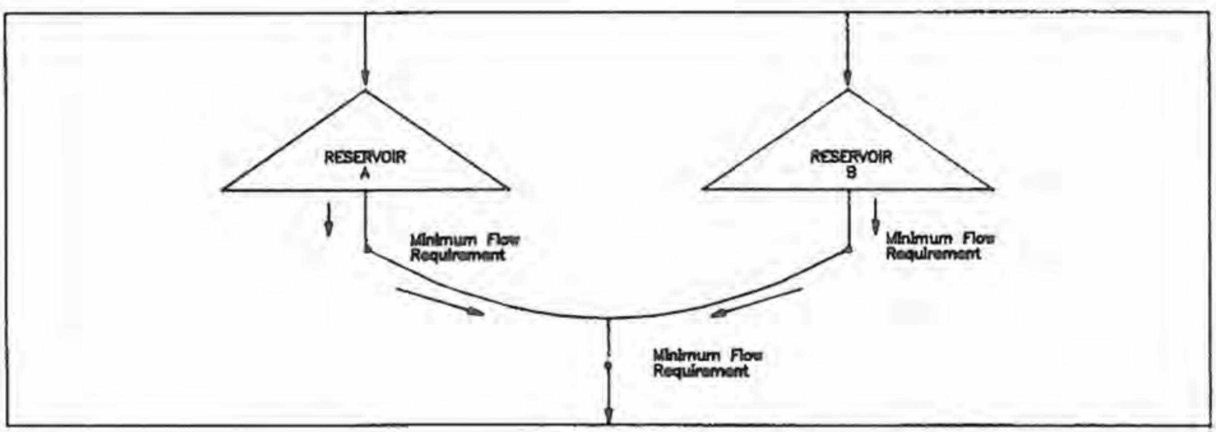


FIGURE 3

# A SURFACE-SUBSURFACE HYDROLOGIC MODEL APPLICABLE TO HILLY TERRAIN UNDERLAID BY WEATHERED SHALES[1]

Roger B. Clapp[2] and Albert Lin
Oak Ridge National Laboratory
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There are no reliable hydrologic models for conditions on the Oak Ridge Reservation (ORR). Thus, all model-based characterizations of contaminant movement from old disposal facilities are suspect and all predictions of the transport of contaminants from proposed future facilities will probably be wrong if conventional models are used.

The reason for the inadequacy of existing models is the complexity of the geology on the ORR. The shale units in the Valley and Ridge Province have a very low hydraulic conductivity, but there is a thin layer (about 1.5 m thick) at the surface with high hydraulic conductivities, as is typical of forested landscapes. Between these two zones the transition in hydraulic properties is very irregular, and is provided mainly by fractures. In the upper soil layer, permeability is provided mainly by macropores associated with root channels, animal holes, and other soil-forming processes.

A key feature of the hydrologic system on the ORR is the close coupling of the surface water and the groundwater regimes. Overland flow on hillslopes is a rare occurrence; yet stream discharge increases within an hour of the onset of rain, indicating rapid subsurface flow paths. Furthermore, transient saturated conditions at shallow depths cause inundation of excavations and buried wastes. As a consequence, buried contaminants are transported to surface water during storms and this movement of contaminants to surface water can easily go undetected by monitoring wells.

To date, very few spatially distributed models depict both surface water flows and water table dynamics, but this is precisely what must be done in order to have a reliable model for conditions on the ORR. The purpose of this report is to provide an overview of the modeling approach that simulates both regimes.

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[1]Research sponsored by Nuclear and Chemical Waste Programs, Office of Energy Research, U.S. Department of Energy under contract DE-AC05-84OR21400 with Martin Marietta Energy System, Inc.

[2]Geosciences Section, Environmental Sciences Division, Oak Ridge National Laboratory, P.O. Box 2008, Oak Ridge, Tennessee 37831-6038 (615/576-6619).

## MODELING GOALS

The goals of model development are to simulate the surface water hydrograph by simulating the flow of water through major hydrologic components of the system, to provide a framework for modeling the movement of dissolved material, to identify data gaps, and to focus future field activities. The initial objectives are to provide estimates of rates of recharge to groundwater and to develop a model of subsurface stormflow, which is needed to design drainage systems. The modeling approach can be classified by hydrologic scale. The macroscale requires flow routing within the channel network, the mesoscale includes the rate of delivery of water to the streams (i.e., the hillslope and overland flow regimes), and the microscale consists of the exchange of water between the macropores or fractures and the soil matrix in the shallow stormflow zone.

## MACROSCALE: CHANNEL ROUTING

For large basins the routing of flows via the channel network is important. Given the magnitude of watersheds on the ORR and the complexities of the ORR hydrologic system, a computationally simple method, such as the geomorphic instantaneous unit hydrograph (IUH), for flow routing should suffice. The IUH requires a linearized advective-dispersive equation for flow in a single channel link, channel parameters, and a time-varying input from the adjacent hillslope. The effects of the channel network can be incorporated into the IUH via the channel width function, which is obtainable from a map of the channel network. In the envisioned computer model, calculation of the stream hydrograph is separate from the calculation of hillslope runoff.

## MESOSCALE: HILLSLOPE FLOW PRODUCTION

The hillslope portion of the model is the major focus of model development. The hillslope hydrologic regime is conceptualized as a tilted layer cake. As shown in Fig. 1, flows are routed laterally to the stream in the stormflow zone, which is estimated to be about 1-1.5 m thick. Below that zone the water moves vertically into the vadose zone, which is nominally 8 m deep at the ridge line and tapers to nothing at the toe of the slope. Below the vadose zone is the shallow groundwater zone bounded above by the water table and below by a low-permeability boundary at a depth specified by the modeler.

### Storm Flow Zone (SFZ)

A significant portion of infiltrated water moves laterally in the shallow subsurface stormflow zone, as described by Moore (1988, 1989) based largely on the ongoing field work of R. Luxmoore and his past and present colleagues K. W. Watson, G. V. Wilson, and P. M. Jardine. In the model, flows are simulated by applying the extended Dupuit-Forchheimer approximation in which flow lines are assumed to be parallel to the surface slope $\alpha$. For hydraulic conductivity (K), the flow (Q) is given by:

$$Q = Kh(\sin \alpha - dh/dx \cos \alpha)$$

where x is oriented along the slope and h is the height of the water surface. This expression can be simplified by assuming that dh/dx is negligible, yielding a linear kinematic wave equation. Where h exceeds the depth of the stormflow zone the soil is fully saturated and the surface becomes a dynamic contributing area for overland flow. Water in the SFZ is removed by downward seepage, lateral discharge, and flow into (or out of) the soil matrix.

Vadose Zone (VZ)

A layer of lower permeability beneath the SFZ will cause unsaturated conditions at greater depths. Although the soil is fractured within the vadose zone, flow in these fractures and in large pores may be infrequent, and it may be assumed that conventional unsaturated porous-medium relationships prevail. Flows can thus be depicted by a single vertical soil column. Flux to the capillary fringe is evaluated at the appropriate depth in the modeled soil column. As shown in Fig. 1, when the soil column depicts a wetting front moving downward in VZ, the front arrives at the water table earliest at the toe of the slope and latest beneath the ridge line. This conceptualization will lead to transient groundwater mounding near the stream and a quick response from the shallow groundwater zone. The capillary fringe can be assumed to have a fixed elevation at any location because changes in water table are seldom greater than about 2.5 m beneath the ridge lines.

Shallow Groundwater Flow Zone (SGWZ)

Movement of water below the water table is modeled by the same extended Dupuit-Forchheimer assumption. The depth of the bottom boundary (the depth of the hydrologically active groundwater zone) is a debated issue among geohydrologists at Oak Ridge. Evidence from a small area surrounding the Center Seven watershed shows that hydraulic conductivity decreases as depth increases and thus that most lateral flows occur near the water table. This relationship between depth and hydraulic conductivity is not evident in data collected regionally. The implications can be investigated via the model, but resolution will come only from data collection and interpretation.

## MICROSCALE: FLOW IN THE SOIL MATRIX

In the SFZ lateral flows occur in macropores, but changes in water storage within soil clods may significantly affect the accumulation of water in the macropores. In the Matrix Submodel the soil clods are conceptualized as a series of thin slabs having two parallel edges. Absorption and drainage at the edges of the slabs are simulated as bulk flow by using the classic Green-Ampt equation for both infiltrability and drainability. The application of the Green-Ampt wetting front principles, which are developed from soil physics theory, to dual domain flow (macropore-micropore flow) is innovative. In the model, water within the matrix system is depicted by two state variables: an average water content (W) and an infiltration parameter (F). The model can be modified to investigate variability in hydraulic properties and the size distribution of the soil slabs.

## DISCUSSION

In addition to the flow systems identified above the proposed model will provide for interception and a deep groundwater storage. The state variables are then I(t), h(x,t), F(t), W(t), Θ(z,t), H(X,t), S(t), as shown in Fig. 1. These variables are calculated through time by numerical integration by using variable time-step Runge-Kutta methods. The model is being developed to simulate single rainfall events but will be extended later to be a continuous simulator by including losses to evapotranspiration.

The proposed model is a computational tool to explore the concepts of flow outlined by Moore (1988). It may provide the best estimates of rates of groundwater recharge, critically needed information for detailed geohydrologic studies. The SFZ submodel may be adapted as a practical engineering tool for the hydrologic isolation of buried wastes. The dual hillslope conceptualization (lateral flows in both the SFZ and the SGWZ) is unique in hydrology, as is the proposed conceptualization of soil physics within the matrix submodel. In summary, the model provides many opportunities to investigate flow mechanisms that prevail in the shaley hills of the ORR and elsewhere, and it has high potential for producing long-sought information needed in many environmental protection programs at the Reservation.

## REFERENCES

Moore, G. K. 1988. Concepts of Groundwater Occurrence and Flow Near Oak Ridge National Laboratory, Tennessee. ORNL/TM-10969. Oak Ridge, Tennessee.

Moore, G. K. 1989. Groundwater Parameters and Flow Systems Near Oak Ridge National Laboratory, Tennessee. ORNL/TM-11368. Oak Ridge, Tennessee.

ORNL DWG 90M-11435

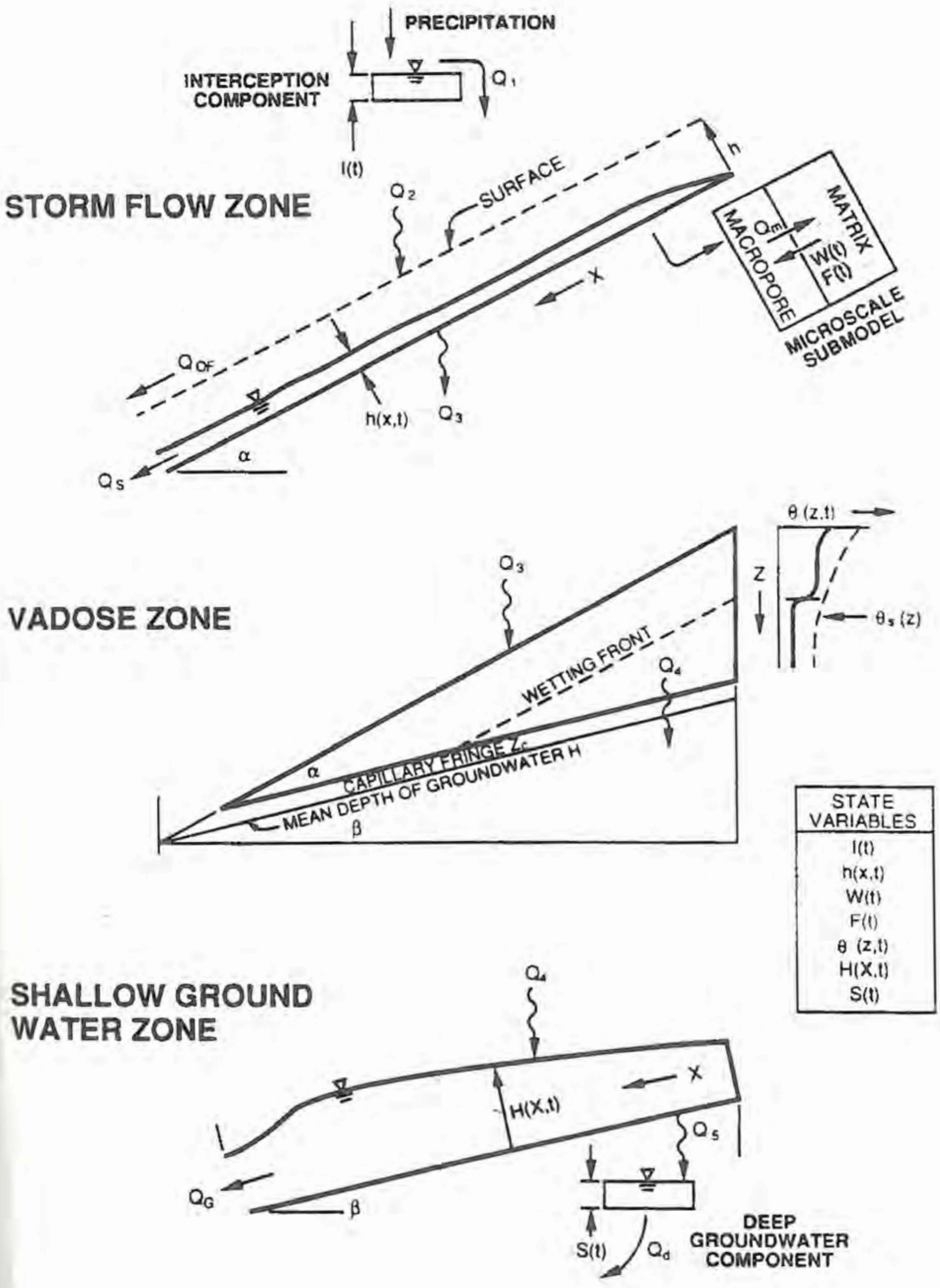


Fig. 1. Flow systems with the hydrologic model.

# PHYSICAL HABITAT EFFECTS OF SELECTIVE CLEARING AND SNAGGING

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## INTRODUCTION

Clearing and snagging (removal of bank vegetation and large woody debris (LWD) from channels) may be used for reducing the stage and duration of high frequency. Complete clearing and snagging can have detrimental effects upon stream habitat. Thus, a balance between habitat considerations and channel conveyance is necessary. Selective removal of bank and near-channel floodplain vegetation and channel obstructions is a means of accomplishing this balance.

This paper describes a study conducted on a reach of the South Fork Obion River (SFOR) in west Tennessee while a clearing and snagging project sponsored by the Obion-Forked Deer Basin Authority (OFDBA) was in progress. The study was part of a larger program to develop techniques to quantify and predict incremental physical and biological effects of LWD removal. Long-term research objectives are to relate the densities and types of LWD formations in streams to specific biotic parameters (species type and densities); flow conveyance; and longitudinal dispersion as an indicator of the tendency of a channel reach to trap and hold fine particulate matter. The principal objective of this SFOR study was to investigate effects of selective clearing and snagging on physical conditions and aquatic habitat in a sand bed river.

## DATA COLLECTION AND ANALYSES

Data were collected from a reach of the SFOR near Bradford, Tennessee in Gibson, Weakley and Carroll Counties. Implementation of the project was in strict compliance with selective clearing and snagging guidelines (GWTNRTF 1985) for Class III work. The work consisted of bank clearing and snag removal along six miles of the main channel and four miles of smaller tributaries.

The SFOR is a straight stream with a uniform, trapezoid-shaped channel resulting from channelization in the early 1910's. Old disposal piles are still evident along the edges of the main channel, attesting to channel stability. Present channel widths are 60-75 ft near top bank with maximum depths in the range of 12-15 ft. At normal low flow conditions, water surface widths are 40-55 ft with depths in the 2-5 ft range.

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[1]Currently on Intergovernmental Personnel Act assignment from Memphis State University to Environmental Laboratory, Waterways Experiment Station, 3909 Halls Ferry Road, Vicksburg, Mississippi 39180 (601/634-3047).

Four field trips have been made to the SFOR site in order to collect physical data. Three types of data were collected: snag density counts, dye tracer tests, and physical habitat (depth, velocity, bed type, and cover) at selected transects. Snag density was the important independent variable, while the dye tracer and physical habitat data were used to study macro- and micro-scale effects of LWD, respectively.

Three different reaches (each one mile or longer) were established for snag density counts and fluorescent dye flow tests. Temporary staff gages were established at the upstream and downstream ends of each reach. One reach was located in the lower cleared portion and the other two reaches were established in the middle and upper uncleared portions of the SFOR site.

Snag Density - Several methods have been devised (Wallace and Benke 1984; Platts et al. 1984) for quantifying LWD. For the purposes of this study, LWD formations were described using the National Marine Fisheries Service (NMFS) classification system (Platts et al. 1984) supplemented with a size criteria we developed.

Debris position in the stream affects stability of LWD and its use by fish and macroinvertebrates. The NMFS system describes LWD in terms of stability and association with the bank.

Two different LWD density parameters were computed. They represent the LWD cross-sectional area and volume per unit channel water volume, respectively.

Dye Tracer Tests - Computation of discharge and associated channel hydraulic parameterms were made using the dye dilution method for each of the three selected reaches. An appropriate volume of Rhodamine WT dye was instantaneously released at the upper end of each reach from a small boat. A flow-through fluorometer at the lower end of the reach was used to measure the dye concentration with time.

Dye tracer data and concomitant stage measurements were used to compute the mean and variance of time of travel, mean velocity, discharge, mean cross-sectional area, mean hydraulic depth, and channel roughness. Gross longitudinal mixing and aquatic habitat heterogeneity is manifested in dye tracer curves by the spread of the base. This spread also reflects all of the mixing processes including irregularity of channel shape and cross section, pool and riffle sequences, and effects of temporary storage. Longitudinal dispersion coefficients and several other statistics were computed to quantify the dye curve spread; the travel time variance is presented herein.

Physical Habitat - Physical habitat measurements were made along selected transects and diversity was determined using methods similar to those described by Gorman and Karr (1978). Velocity, depth, substrate, and cover were measured at 3-ft intervals along each transect using a tagline to locate sampling points. Depth and velocity measurements were later converted into integer (i.e., category) values and bed material and cover were visually categorized in the field. Samples of bed material were also collected at each transect for laboratory sieve analyses. Shannon diversity indices were calculated for the combination of the four physical habitat variables.

Frequency histograms were generated for each physical variable and each reach. Means and standard deviations of the physical habitat variables and median bed material size were computed for each reach. Analysis of variance (ANOVA) were performed for depth, velocity, and bed material size data grouped by reach and grouped into sampling points with cover and without cover.

Snag Density vs. Roughness - Friction factors for each of the three reaches were calculated using a procedure summarized by Smith and Shields (1990) for predicting roughness in a sand bed stream where LWD plays a major role in the flow resistance. In this procedure, the total friction factor is assumed to equal the sum of a boundary friction factor excluding LWD effects and a friction factor due to the LWD. The boundary friction factor is a function of the average hydraulic parameters in the channel and the median bed sediment grain size. The added friction due to the presence of LWD is a function of the snag density and mean hydraulic depth.

## RESULTS AND SUMMARY

Selective clearing and snagging had measurable effects on the physical habitat characteristics of the South Fork Obion River. This becomes evident by comparison of the computed parameters from snag density counts, dye tracer tests, and physical habitat data. Habitat diversity results are shown in Table 1 and physical habitat frequency histograms are shown in Figure 1.

Computed snag densities (in units of sq ft snag area per cu ft water volume) under low flow conditions ranged from 0.003 to 0.005 for uncleared reaches and for cleared reaches 0.000 to 0.002. Although the method for quantifying snag density used in this study was crude, the resulting values are in rough agreement with others (Wallace and Benke 1984; Gorman and Karr 1978).

Relative to the uncleared reaches, the cleared reach had shallower mean depth, higher mean velocity, and slightly coarser substrate at low flow--conditions less preferable to many warm water sportfishes. Shannon indices and dispersion coefficients indicated that snags created eddies and areas of local acceleration and deceleration of flow and attendant zones of scour and deposition. Removal of the snags evidently produced more uniform conditions and reduced physical habitat diversity. Figure 2 is a plot of travel time variance vs. discharge obtained from the dye tracer tests. Evidently LWD increases physical heterogeneity at low flows, but has diminishing influence with increasing discharge. The biological significance of the above findings is currently under investigation.

The ratio of predicted to measured Darcy-Weisbach friction factor varied in the ranges of 0.40-1.30 for low flows, 0.65-1.25 for mid-bank flows, and 0.80-1.15 for high flows. The procedure for predicting roughness using snag density and median sediment grain size appears to work reasonably well for flows near bankfull conditions. However, additional energy losses due to abrupt expansions and contractions (neglected in the above procedure) should be included in the simulation of low-flow hydraulics.

Effects of clearing and snagging upon hydraulic roughness are shown in Figure 3. As the flow approaches bankfull conditions, the snags become deeply submerged and the friction factors for cleared and uncleared conditions tend to converge (range 0.13-0.18). The net positive drainage benefits will depend upon duration and frequency analyses of the in-bank flows for both uncleared and cleared conditions.

## ACKNOWLEDGEMENTS

The author would like to express his sincere thanks to the staff members of OFDBA and Continental Engineering, Inc. for their cooperation and assistance in the data collection efforts. The tests described and the resulting data presented herein, unless otherwise noted, were obtained from research conducted under the Environmental Impacts Research Program of the United States Army Corps of Engineers by the Environmental Laboratory, Waterways Experiment Station, Vicksburg, Mississippi. Technical reviews of this paper were provided by Drs. F. Douglas Shields and Andrew Miller of the Environmental Laboratory and Mr. David Mueller of the Hydraulics Laboratory. Permission was granted by the Chief of Engineers to publish this information.

## REFERENCES

Gorman, O. T. and J. R. Karr, 1978. Habitat Structure and Stream Fish Communities, Ecology, Vol. 59.

Governor's West Tennessee Natural Resources Task Force (GWTNRTF), 1985. Stream Renovation and Maintenance Guidelines for the Obion-Forked Deer River System. (Part of a Federal Court Agreed Order for the OFDBA).

Platts, W. S., et al., 1987. Methods for Evaluating Riparian Habitats with Applications to Management. Gen. Tech. Report INT-221, U.S. Dept. of Agric., Forest Service, Intermountain Research Station, Ogden, Utah.

Smith, R. H. and F. D. Shields, Jr., 1990. Effects of Selective Clearing and Snagging on Physical Conditions of Rivers, Proceedings of Twentieth Mississippi Water Resources Conference, Jackson, MS.

Wallace, J. B. and A. C. Benke, 1984. Quantification of Wood Habitat in Subtropical Coastal Plain Streams, Can. J. Fish. Aquat. Sci., Vol. 41.

TABLE 1 - HABITAT DIVERSITY RESULTS

| Reach | Shannon Index | $D_{50}$ (mm) | $V_{mean}$ (ft/s) | $h_{mean}$ (ft) |
|---|---|---|---|---|
| Lower cleared | 2.17 | 0.60 | 1.3 | 2.8 |
| Middle uncleared | 3.02 | 0.44 | 1.1 | 2.7 |
| Upper uncleared | 2.51 | 0.27 | 0.9 | 3.0 |
| Points with cover (all reaches)[2] | | 0.31 | 0.7 | 2.4 |
| Points without cover (all reaches)[3] | | 0.47 | 1.3 | 3.1 |

[2,3]ANOVA showed the median sediment grain size, velocity, and depth differences were significant at the 95 percent confidence level.

LEGEND

| BAR NUMBERS → | | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|---|
| DEPTH | RANGES | 0 TO 0.164 | > 0.164 TO 0.656 | > 0.656 TO 1.640 | > 1.640 TO 2.624 | > 2.624 | — |
| VELOCITY | RANGES | 0 TO 0.0328 | > 0.0328 TO 0.1641 | > 0.1641 TO 0.6562 | > 0.6562 TO 1.3124 | > 1.3124 | — |
| $D_{50}$ | RANGES | 0 TO 0.065 | > 0.065 TO 0.30 | > 0.30 TO 0.40 | > 0.40 TO 0.60 | > 0.60 TO 2.0 | — |
| COVER TYPE | | 1 | 2 | 3 | 4 | 5 | 6 |

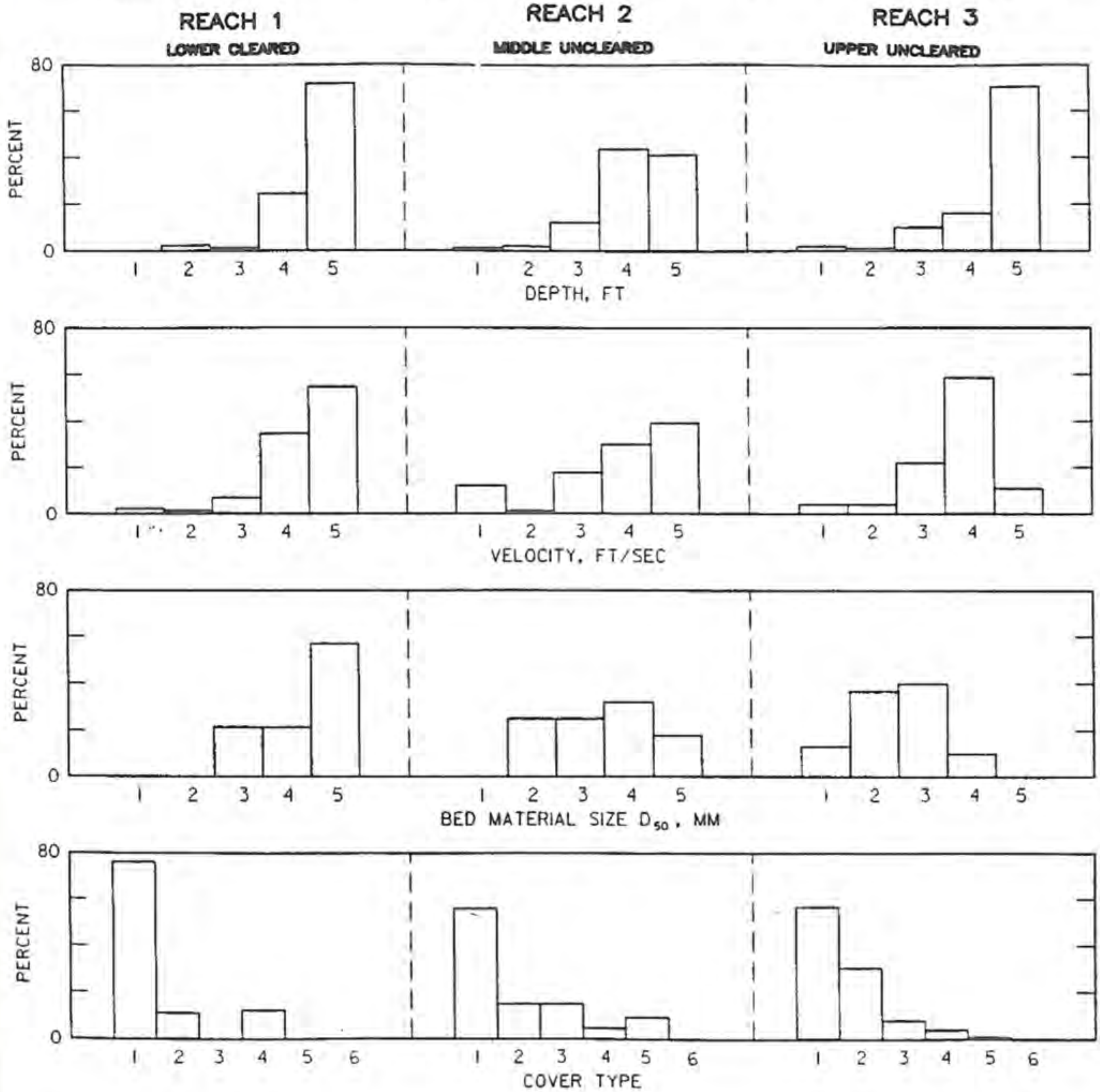


Figure 1. Physical Habitat Frequency Histograms

FIG. 2 — LWD EFFECTS ON TRAVEL TIME VARIANCE

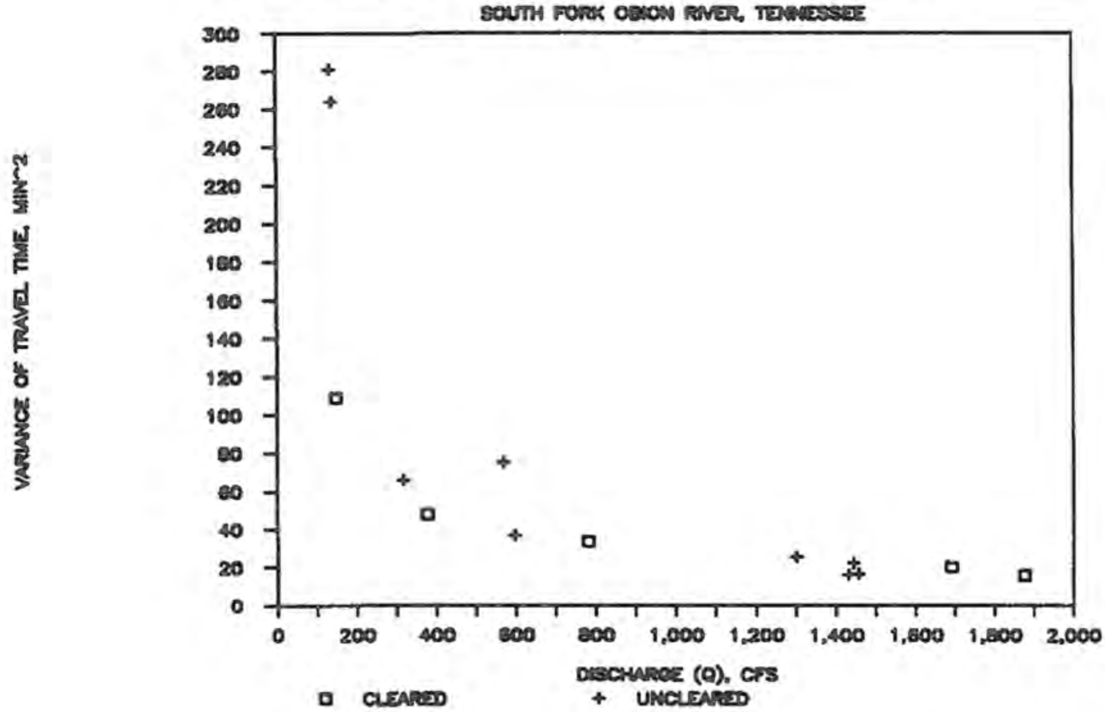


FIG. 3 — LWD EFFECTS ON ROUGHNESS

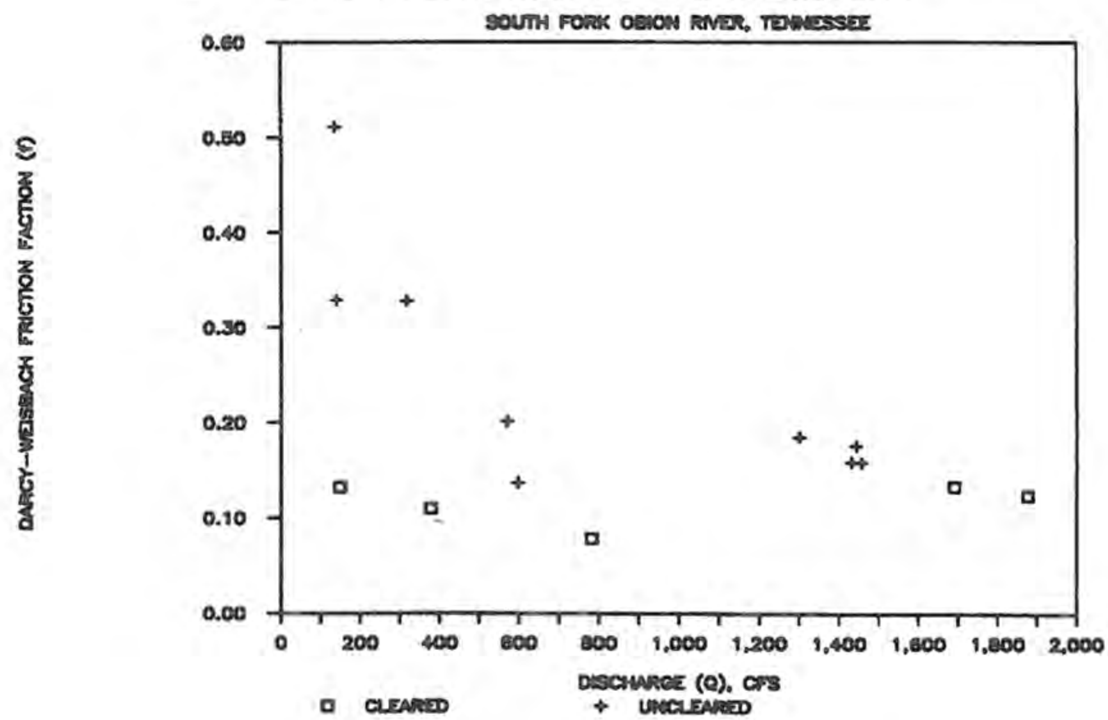


Figure 2. LWD Effects on Travel Time Variance

# SURFACE WATER MODELS AND GEOGRAPHIC INFORMATION SYSTEMS FOR INTEGRATED RISK ANALYSIS OF POLLUTANTS IN WATTS BAR RESERVOIR[1]

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## INTRODUCTION

Decades of operation of the U.S. Department of Energy Oak Ridge facilities have resulted in the chemical contamination of Watts Bar (Olsen et al. 1990), a mainstem reservoir created and managed as part of the Tennessee River system by the Tennessee Valley Authority. Watts Bar is an important resource for flood control, hydropower, navigation, sport fishing, recreation, and potable water. The Clinch River remedial investigation (RI), a major DOE effort, is underway to characterize the nature and extent of the contamination, as well as to examine the feasibility and implications of potential off-site remedial actions. The remedial investigation will also explore the implications of waste management and clean-up activities performed on the Oak Ridge Reservation with regard to their potential impacts on the future environmental integrity of Watts Bar Reservoir.

The complex nature and magnitude of the problem will be addressed through a coordinated, multi-phased approach that organizes existing information, collects additional data, uses process-oriented models of Watts Bar reservoir, and provides a probabilistic framework for human health and environmental assessment. The Clinch River RI will rely significantly on the integration of dynamic models, geographic information systems, and risk analysis. This document outlines these components of the investigation.

## SURFACE WATER MODELING

Differently scaled, spatially explicit hydrodynamic models (e.g., Yeh 1983) will be modified and implemented for Watts Bar Reservoir. These models will be used to simulate velocity profiles, pool volumes, water quality, sediment transport and the distribution of contaminants within the reservoir. The modeling efforts will address the environmental implications of regional meteorology, seasonal reservoir operations, and possible remediation activities. Simulated discharge and water velocity profiles will be used to estimate the transport and distribution of dissolved contaminants, suspended particulates, nutrients, and plankton. Reservoir hydrodynamics influence both the basic ecological production and the environmental chemistry of contaminants within the reservoir. For

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[1]Research sponsored by the Office of Environmental Restoration and Waste Management, U.S. Department of Energy under contract DE-AC05-84OR21400 with Martin Marietta Energy Systems, Inc.

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example, wind and water currents determine plankton distribution and the rate of loss of volatile organic compounds; vertical light intensities influence photosynthesis by plants and photolytic degradation of chemical contaminants. Water quality parameters such as dissolved oxygen, nutrient concentrations, and temperature constrain biological production, which in turn determines the potential for accumulation of dissolved and particulate-bound contaminants by the reservoir biota. Hydrodynamics also influences the net suspension and redistribution of sediments in relation to reservoir operations, local storms, and dredging. Models of these phenomena will be used together with data to increase our quantitative understanding of the ecological and toxicological structure and function of Watts Bar Reservoir.

## GEOGRAPHIC INFORMATION SYSTEMS

One or more geographic information systems (GIS) will be developed for Watts Bar and its regional watershed. The Watts Bar GIS has three primary purposes: One, the GIS will contain the hydrological, ecological, toxicological, meteorological, and sediment data from historical and current studies of this reservoir. These data will be used to directly characterize the ecological and toxicological status of the system, to explore remediation alternatives, and to develop and evaluate models. Two, the initial and boundary conditions for the reservoir simulation models will be maintained as components of the GIS. Certain boundary conditions including surface runoff and non-point source contamination will derive in part from the topographical and meteorological data in the GIS. These conditions will serve to integrate the hydrodynamic models with the GIS. Three, selected spatial results from the reservoir models will be stored and displayed using the GIS capabilities.

## INTEGRATED RISK ANALYSIS

Risk analysis provides a conceptual and methodological framework for integrating contaminant fates and effects in the regional Watts Bar system. Risk is estimated as a conditional probability of occurrence for a specified endpoint. For human health, a common endpoint is the likelihood of excess cancer during a lifetime of exposure. Examples of ecological endpoints include the probability of exceeding water quality standards for individual pollutants or that of reducing the population size of an endangered or commercially valuable species by some fraction (Bartell et al. 1987). The historical and newly acquired data and model results will be used to estimate human health and ecological risks in relation to measured or modeled exposure concentrations for selected radionuclides, trace metals, and organic chemicals. These chemicals have been identified and assigned priorities using the results of preliminary human health (Hoffman et al. 1990) and ecological (Suter 1990) screening calculations.

Risk is a function of exposure, effects, and uncertainties. Sources of uncertainties include imprecise estimates of exposure, inaccurate extrapolations of laboratory assays to field conditions, sampling errors, natural variability, and incomplete understanding of system structure and function (O'Neill et al. 1982, 1983). Therefore, sensitivity and uncertainty analyses will be used to quantify the contributions of these different uncertainties to risk estimates.

The results of these analyses will then be used to judiciously allocate limited experimental and monitoring resources to reduce uncertainties and refine risk estimates.

## REFERENCES


Bartell, S. M., R. H. Gardner, and R.V. O'Neill, 1988. An Integrated Fates and Effects Model for Estimation of Risk in Aquatic Systems, pp. 261-274. In: Aquatic Toxicology and Hazard Assessment: 10th Volume, ASTM STP 971, Philadelphia, Pennsylvania.

Hoffman, F. O., B. G. Blaylock, M. L. Frank, L. A. Hook, E. L. Etnier, and S. S. Talmage, 1990. Preliminary Screening of Contaminants in the Off-Site Surface Water Environment Downstream of the U.S. Department of Energy Oak Ridge Reservation. Oak Ridge National Laboratory, ORNL/ER-9. Oak Ridge, Tennessee.

Olsen, C. R., I. L. Larsen, P. D. Lowry, C. R. Moriones, C. J. Ford, K. C. Dearstone, R. R. Turner, and B. L. Kimmel, 1990. Transport and Accumulation of Cesium-137 and Mercury in the Clinch River and Watts Bar Reservoir System. Oak Ridge National Laboratory, ORNL/ER-7. Oak Ridge, Tennessee.

O'Neill, R. V., R. H. Gardner, L. W. Barnthouse, G. W. Suter, S. G. Hildebrand, and C. W. Gehrs, 1982. Ecosystem Risk Analysis: A New Methodology. Environmental Toxicology and Chemistry, 1: 167-177.

O'Neill, R. V., S. M. Bartell, and R. H. Gardner, 1983. Patterns of Toxicological Effects in Ecosystems: A Modeling Study. Environmental Toxicology and Chemistry, 2: 451-461.

Suter, G. W., II, 1990. Screening Level Risk Assessment for Off-Site Ecological Effects in Surface Waters Downstream from the U.S. Department of Energy Oak Ridge Reservation. Oak Ridge National Laboratory, ORNL/ER-8. Oak Ridge, Tennessee.

Yeh, G. T, 1983. CHNTRN: A Channel Transport Model for Simulating Sediment and Chemical Distribution in a Stream/River Network. Oak Ridge National Laboratory, ORNL-5882. Oak Ridge, Tennessee.

# A SIMPLIFIED THORNTHWAITE CLIMATIC-INDEX MODEL TO STUDY POTENTIAL CHANGES IN RUNOFF IN TENNESSEE INDUCED BY THE GREENHOUSE EFFECT

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The impact of the "greenhouse effect" on the hydrology of Tennessee was studied with a simplified model based on Thornthwaite's climatic index. The simplified model was selected as a tool to screen potential climate-change scenarios in Tennessee due to the uncertainty in the precision and validity of global models. Average annual temperature and precipitation data throughout the state were used to compute values of the index representing long-term conditions. The index was then re-computed utilizing potential scenarios representing a range of changes in temperature and precipitation. Scenarios were tested representing changes in temperatures from + 1.0 to + 4.5 degrees Centigrade, and changes in precipitation from - 20 percent to + 20 percent. Values of the climatic index were calculated for 15 different scenarios. The scenarios representative of the extreme changes in temperature and precipitation were related to the average annual runoff throughout Tennessee. The results of the computations show that significant changes in the amount of runoff would result under several of the scenarios. These changes in runoff could result in significant flooding in several areas, and for the extreme drier conditions, a reduction in the available water supplies.

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# EFFECTS OF CLIMATIC CHANGE ON TVA ACTIVITIES

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## ABSTRACT

TVA is conducting an analysis to define how climate change may affect the various programs of the agency. The reservoir system, the power system, and most of the agricultural and natural resource systems in the Valley are susceptible to upset by climatic extremes. Operational experience during the drought of the 1980s clearly showed the vulnerability of these systems and the Valley itself to extremely hot and dry weather. It also revealed the strong interrelationship among many system elements and agency activities that can make it difficult to isolate adverse effects and correct them. The TVA plan is to perform an integrated examination of potential impacts Valley wide. This assessment will draw upon TVA experience during extreme weather conditions and expertise in integrated assessment as well as results of other Federal research. Plans for mitigation of potential impacts will be developed considering a wide range of program activities.

In addition to internal TVA support, this assessment is supported with funds provided by the Electric Power Research Institute, U.S. Environmental Protection Agency, and U.S. Bureau of Reclamation. These agencies are not only interested in results from the program, but in learning from the approach for performing such an integrated assessment as well.

## INTRODUCTION

The probability of climate change presents a challenge for natural resource management agencies, and TVA is no exception. The reservoir system, the power system, and most of the agricultural and natural resource systems in the Valley are susceptible to upset by climatic extremes. Operational experience during the drought of the 1980s clearly demonstrated that our systems and the Valley itself is vulnerable to extremely hot and dry weather. It also revealed the strong interrelationship among many system elements and agency activities that make it difficult to isolate adverse effects and mitigate them.

Predicted changes in weather patterns include changes in precipitation, storm tracks, and temperature extremes. Potential consequences of these altered patterns include increased incidents of floods, droughts, forest fires, crop losses, fish kills, and power shortages. However, it should be noted that resource management agencies have always been required to address these problems,

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and they have amassed a wealth of experience. The concern is that we may be forced to operate with weather extremes which are outside of our range of experience. For example, almost a century of weather data reveals that annual precipitation across the Tennessee Valley has varied no more than %30 percent. Experience indicates that the Valley is susceptible to floods or droughts when precipitation approached these extremes. Although estimates of changes in precipitation are very tentative at this time, it is easy to postulate weather patterns for the Tennessee Valley which will exceed this relatively narrow range of precipitation. For this reason, it is important to examine potential impacts of climate change, and to incorporate these factors into long-range planning.

Miller and Brock (1988) presented results of a TVA examination of the sensitivity of the reservoir system to climatic shifts. That analysis was included in Smith and Tirpak (1988), which presented to Congress an assessment sponsored by the Environmental Protection Agency of the potential effects of climate change in the United States. Although the analysis of the TVA reservoir system was limited in scope, it provided the basis for developing the fully integrated plan for the study presented herein.

## INTEGRATED ASSESSMENT STRATEGY

TVA has undertaken a systematic, integrated examination of potential impacts Valley wide. The integrated approach was chosen because many of the required analyses are common to more than one program. Furthermore, neither potential adverse effects nor corrective actions can be disaggregated. This assessment will draw upon experience gained during weather extremes, expertise in integrated analysis, and information from other Federal research activities.

The project plan, presented in Figure 1, begins with an examination of results of General Circulation Models (GCMs) as well as 40 years of meteorological records to define more clearly the changes in weather patterns most likely to occur throughout the Tennessee Valley. Although all GCMs predict warmer temperatures, they disagree significantly as to precipitation changes in our region. TVA atmospheric scientists are attempting to develop methodology required to provide a better understanding of regional atmospheric processes and their relationship with global scale processes. This is necessary to allow application of global model results to a regional scale. Results of this analysis will also be useful for estimating future trends of regional air quality, and for developing strategies for environmental protection.

Simultaneously, a hydrologic model is being applied to four representative watersheds across the Valley. This model will simulate the effects of changes in precipitation, evapotranspiration, and other appropriate hydrologic variables on basin runoff, soil moisture, and ground water recharge. Results from this hydrologic model will also provide the basis for assessing changes in nonpoint source pollution and flood frequencies for unregulated streams. Changes in soil moisture coupled with atmospheric temperatures and airborne pollutants will be used to assess impacts on the terrestrial ecosystem. Such factors as the shifts in forest species ranges, changes in productivity of crops and forest resources, heat stress to crops, and changes in agricultural practices will be assessed.

Results of this assessment will be fed back into the hydrologic model to account for changes in land cover and evapotranspiration.

Mathematical models of several TVA reservoirs will incorporate incremental changes in runoff predicted by the hydrologic model along with results from the meteorological analysis. Two-dimensional reservoir models, such as the one described by Hauser, et al. (1987), will be used to evaluate how stratification, dissolved oxygen, and other water quality parameters may be affected by changes in climate. These models will provide predictions of suitable habitat (i.e., regions of acceptable temperature and dissolved oxygen) for fish and other important aquatic species.

The effects of incremental changes in reservoir discharge flow rates, temperatures, and dissolved oxygen will be evaluated using mathematical models of the entire reservoir system. Although these systemwide models do not provide the two-dimensional resolution of the individual reservoir models, they are adequate for integrated assessments of water resource demands on a systemwide basis. A model of the TVA reservoir system (Shane and Gilbert, 1981) is currently used for systemwide planning and operation for flood control, hydropower, navigation, recreation, water supplies, and other flow-related factors. An enhanced model to predict water temperatures and dissolved oxygen throughout the river system is under development. This model will facilitate systemwide planning to assure optimum operation of the reservoirs to promote multiple objectives. Effects of climate change on river flow and water quality can then be evaluated throughout the entire river system.

## CURRENT STATUS

The assessment of potential effects of climate change on the various program interests of TVA is designed as a multiyear program to draw upon other analyses whenever possible. Because a better understanding of climate changes for the Tennessee Valley is vital to developing a coping strategy, this is an area of immediate and continuous concern. Development of a methodology for improving the credibility of climate change predictions for the Valley is now being studied. Similar activities of other research programs will be monitored to assure that we are using the best methods possible.

Defining effects of climate change upon hydrology and other aspects of water resources is fundamental to describing effects upon natural resources. For this reason, computer models designed to provide a better definition of streamflow, lake levels, and water quality throughout the system are under development. Such models will have immediate use by TVA in planning and operation as well as in evaluating potential effects of climate change.

A quantitative assessment of effects on most areas of natural resources cannot be evaluated until results of regionalized climate changes and enhanced hydrologic and reservoir models are available. While awaiting results from those development efforts, TVA has begun to examine many of the agency activities to identify vulnerabilities to climatic extremes. For example, TVA is currently defining vulnerabilities of the hydroelectric, fossil, and nuclear plants and the transmission system to extremely hot and either wet or dry weather. The critical thresholds, defined by environmental, safety, or equipment maintenance limits, as well as degradation of thermal efficiency will be established for each plant. These thresholds will be expressed as a function of localized meteorology, river temperature, and streamflows. Results from assessments of the individual generating facilities and transmission system will be combined to evaluate the ability of TVA to generate and deliver power throughout the power service region during periods of extreme weather. The regionalized climate studies will define the severity and frequency of such extreme events. The Electric Power Research Institute and Environmental Protection Agency are supplementing this TVA study as the results will have broader applicability than the TVA power system.

The assessment of impacts upon most other natural resources is still in the planning stage. These plans will draw upon experience attained during implementation of the power study and from other research programs as well. The assessment of the vulnerability of forest species will be given a high priority because of the potential impact on hydrology. Likewise, changes in agricultural practices should be evaluated soon because of the possibility of increased use of irrigation. Extensive irrigation could severely deplete ground water in some regions which could impact local potable water supplies and base flow to streams and rivers.

Eventually, most areas of natural resources of interest to TVA will be included in this analysis. The goal is to develop a comprehensive computer model or series of models with the capability of performing an integrated assessment of climate change on natural resources throughout the Tennessee Valley. Model results will be used to develop plans for mitigating potential impacts considering a wide range of program and power activities. Segments of this modeling effort currently exist or are under development, but the pace will depend partly upon the success of the international effort in defining the global atmospheric processes associated with climate change. Such analyses are never considered complete since continuous refinement of planning and operational methods are a fundamental responsibility of agencies like TVA.

## REFERENCES

Hauser, G. E., M. D. Bender, M. C. Shiao, and R. T. Brown, 1987. Two-Dimensional Modeling of Water Quality on Cherokee Reservoir, TVA Engineering Laboratory, Technical Report No. WR28-1-590-131, Norris, Tennessee.

Miller, B. A., and W. G. Brock, 1988. Sensitivity of the Tennessee Valley Authority Reservoir System to Global Climate Change. TVA Engineering Laboratory, Technical Report No. WR28-1-680-101, Norris, Tennessee.

Shane, R. M., and K. C. Gilbert, 1981. A Weekly Time Step Scheduling Model for the TVA Operated Reservoir System, International Symposium on Real-Time Operation of Hydro Systems, Waterloo, Ontario, Canada, pp245-262, Water Resources Publications, Littleton, Colorado.

Smith, J. B., and D. A. Tirpack, 1988. The Potential Effects of Global Climate Change on the U.S., Draft Report to Congress, Executive Summary, EPA Office of Policy, Planning and Evaluation, Washington, D.C.

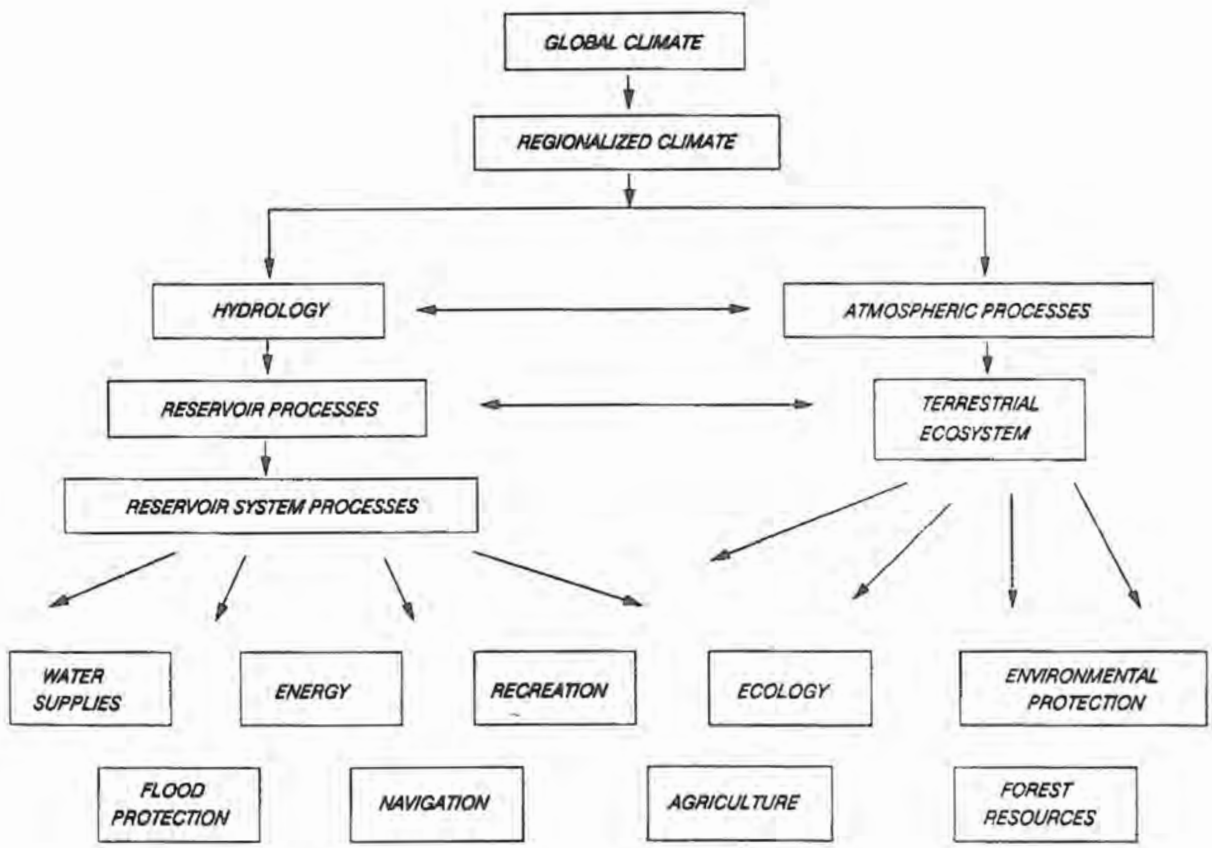


Figure 1. TVA Program for Assessment of Global Climate Change Impacts.

# TENNESSEE CROP WATER REQUIREMENTS IN A GREENHOUSE WORLD

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## INTRODUCTION

Many climatologists agree that a doubling of atmospheric $CO_2$ within the next 50 years will cause a 2 to 5 °F increase in global temperature, with greater warming at the higher latitudes (Schneider, 1989). There is also much speculation about the effects of increased atmospheric $CO_2$ and temperatures on crop water requirements. It has been found that higher amounts of $CO_2$ alone result in increased water use efficiency of many crop plants (Idso, 1989). However, the concomitant warming may increase the evaporative demand of the atmosphere and rates of photosynthesis, evapotranspiration and respiration.

Pans of many shapes and sizes have been used to measure free water evaporation. In humid regions pans may give realistic estimates of potential evapotranspiration ($ET_p$). Crop water use in such climates is usually 60-90% of $E_{pan}$ (Rosenberg et al., 1983). Pan evaporation data have been used to develop curves for potential consumptive use of water by crops (Hatfield, 1990). Irrigation can be scheduled from crop water use data and amount of water required to recharge the root zone at a selected depletion level (Rhoads and Bennett, 1990).

At present, less than 1% of the grain crop acreage in Tennessee is irrigated. It has not been proven that any economical return would be realized by irrigating grain crops.

## OBJECTIVES

The objectives of this research were to 1) to select the three warmest years from 1975 to 1989 for six locations in Tennessee as representative of impending "greenhouse" years and 2) to use daily pan evaporation and precipitation from these locations to drive a maize water use/yield reduction model to predict irrigation requirements during the reproductive period and crop yield reduction for the "greenhouse" years.

## METHODS AND PROCEDURES

Daily Class A pan evaporation and precipitation from 1975-1989 for six locations (counties) in Tennessee - Knoxville (Knox), Crossville (Cumberland), Spring Hill

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(Maury), Ames Plantation (Fayette), Jackson (Madison), and Martin (Weakley) were analyzed for the period June 15 - August 16. This 9-week period was selected as being representative of the range of dates of anthesis and pollination in maize in Tennessee. Maize was used as the sample grain crop because its yield response models are well documented and it is one of the most sensitive to water shortage during the reproductive period.

Correlation analysis was used to examine the relationships between 1) daily pan evaporation and maximum air temperature, 2) weekly pan evaporation and mean maximum air temperature, and 3) seasonal (June 15-August 16) pan evaporation and mean maximum air temperature. Regression analysis was used to determine the regression coefficient with seasonal pan evaporation as the dependent variable and season average maximum air temperature as the independent variable.

A BASIC program (von Bernuth and Logan, 1988) was used to compute estimated consumptive water use by maize during the reproductive period starting on June 15, using daily potential evapotranspiration and effective rainfall. Potential evapotranspiration was estimated as 70% of actual pan evaporation. Soil water holding capacity was estimated as 60 mm at Knoxville, Crossville, and Martin, and 100 mm at Spring Hill, Jackson, and Ames Plantation. It was assumed that the available soil moisture was at 75% of maximum at the beginning of the reproductive season (June 15). When daily rainfall exceeded 2 inches, only 75% of the excess was considered to be effective rainfall. This procedure was repeated for each of the six locations and 15 years. The results focus on yield reductions in maize during the seasons with highest average daily maximum temperature in the 15-year record. Actual county maize yields were used to compare the estimated vs. actual yield reductions.

## RESULTS

On a day to day or week to week basis, the correlation between pan evaporation and maximum daily air temperature was not very high. Daily pan evaporation values tend to fluctuate greatly, and are influenced by many factors other than daily maximum air temperature (i.e. wind over the pan, vapor pressure deficit, solar radiation). The seasonal values of pan evaporation tend to "even out" the day to day discrepancies. The correlations between seasonal (June 15 - August 16) pan evaporation and average daily maximum air temperature over 15 years were highly significant (Table 1). Regression analysis over years (Table 1) indicates that for every 1 °F increase in average seasonal daily maximum air temperature, seasonal pan evaporation will increase 0.40 (Crossville) to 0.58 (Ames Plantation) inches.

Tables 2 and 3 show that 1980, 1988, and 1986 had the warmest seasons for the period 1975-89 at all locations except Jackson, where 1977 turned out to be warmer than 1988. 1980 had the warmest season at all locations except Spring Hill, where both 1988 and 1986 were warmer than 1980. The average seasonal pan evaporation recorded over the three warmest years was considerably higher than the 15-year average, ranging from 14.42 inches at Crossville to 17.20 inches at Spring Hill. Seasonal rainfall was variable over the three warmest seasons, but tended to be below the 15-year average.

The results from the consumptive water use model are also shown in Tables 2 and 3. The range in irrigation requirements was 1.89 inches at Spring Hill (1980) and Ames Plantation (1988), to 6.61 inches at Knoxville (1986), Ames Plantation (1980 and 1986), and Martin (1983). The range in estimated yield reductions was from 0.39 at Knoxville (1986) to 0.97 at Spring Hill (1980). The actual yield reductions ranged from 0.34 for Weakley county (Martin) in 1983 to 0.87 for Fayette county (Ames Plantation) in 1988. On a year to year basis, there was not a very good one-to-one correspondence between estimated and actual yield reductions. County yields are not good estimates of yields at a specific location within the county because the variation between farms is very high. Climatic anomalies do occur, as in the case of Spring Hill in 1980, when significantly greater amounts of rainfall fell from June 15 to August 16 than in the rest of the state. However, the 3-year (warmest) averages of estimated and actual yield reductions were very similar for Knoxville (0.52 vs. 0.53), Crossville (0.53 vs. 0.56), Ames Plantation (0.65 vs. 0.67), and Martin (0.55 vs. 0.54).

Over all locations, the three warmest seasons had an average daily maximum temperature 3 °F above, total rainfall 20% below, and total pan evaporation 11% above the 15-year averages. The estimated and the actual maize yield reductions were 0.63 and 0.57, respectively, over the three warmest years at all locations. Irrigation would have significantly increased maize yields in nearly all of the warmest seasons at all locations. The economic profitability cannot be determined at this point. However, a 60-70% yield reduction would be considered crop failure. An average of only 5.74 inches of total irrigation applied 6 times (capacity of approximately 1 inch over 3 days) would have resulted in maximum yield when estimated yield reduction was 0.70 or less.

## SUMMARY AND CONCLUSIONS

Analysis of temperature, rainfall, and pan evaporation data from six locations in Tennessee (Knoxville, Crossville, Spring Hill, Ames Plantation, Jackson, and Martin) was used to determine the three warmest seasons, ranging from 1.9 to 4.5 °F above normal maximum air temperature, and 53% below to 56% above average rainfall. These warmer and generally drier years can be used as examples of the "greenhouse" years that are predicted for the near future. An average total irrigation of only 5.74 inches (ranging from 3.78 to 6.61 inches) during the critical reproductive period of maize (June 15 - August 16) in these warm years would have resulted in substantial yield increases.

## REFERENCES

Food and Agriculture Organization of the United Nations, 1979. Yield Response to Water. FAO Irrigation and Drainage Paper No. 33. Rome, Italy.

Hatfield, J. L., 1990. Methods of Estimating Evapotranspiration. In: Stewart, B. A., and D. R. Nielsen, (eds). Irrigation of Agricultural Crops. Agronomy Monograph Series No. 30. American Society of Agronomy, Inc., Crop Science Society of America, Inc., and Soil Science of America, Inc., Madison, WI. pp. 435-474.

Idso, S. B., 1989. Carbon Dioxide and Global Change: Earth in Transition. IBR Press, Tempe, AZ.

Peterson, D. F., and A. A. Keller, 1990. Effects of Climate Change on U.S. Irrigation. J. Irrig. Drain. Eng., 116(2):194-210.

Rhoads, F. M., and J. M. Bennett, 1990. Corn. In: Stewart, B. A., and D. R. Nielsen, (eds). Irrigation of Agricultural Crops. Agronomy Monograph Series No. 30. American Society of Agronomy, Inc., Crop Science Society of America, Inc., and Soil Science of America, Inc., Madison, WI. pp. 569-596.

Rosenberg, N. J., B. L. Blad, and S. B. Verma, 1983. Microclimate, the Biological Environment. John Wiley & Sons, New York, NY.

Schneider, S. N., 1989. Global Warming. Sierra Club Books, San Francisco, California.

von Bernuth, R. D., and J. Logan, 1989. Drought in Tennessee: Implications on Irrigation System Capacity. Proceedings of National Water Conference, IR and WR Divs/ASCE, Newark, DE.
pp. 465-470.

Table 1. Results of correlation analysis between pan evaporation and maximum air temperature.

| | Knoxville | Crossville | Spring Hill | Ames Plantation | Jackson | Martin |
|---|---|---|---|---|---|---|
| Over days | 0.44 | 0.53 | 0.89 | 0.52 | 0.45 | 0.50 |
| Over weeks | 0.60 | 0.74 | 0.65 | 0.71 | 0.60 | 0.62 |
| Over years | | | | | | |
| Correlation coefficient | 0.88 | 0.88 | 0.89 | 0.86 | 0.68 | 0.84 |
| Regression coefficient | 0.57 | 0.40 | 0.55 | 0.58 | 0.42 | 0.54 |
| R-square | 0.77 | 0.77 | 0.80 | 0.74 | 0.46 | 0.71 |

Table 2. Seasonal maximum air temperature, minimum air temperature, rainfall, pan evaporation, estimated potential evapotranspiration (0.7 X pan evaporation), estimated irrigation requirement, and estimated and actual maize yield reductions, for Knoxville, Crossville, and Spring Hill for the three warmest years compared to 15-year averages (1975-89).

| Year | Max Air Temp °F | Min Air Temp °F | Precip. inches | $E_{pan}$ inches | $ET_p$ inches | Irrig. required inches | Estimated yield reduction | Actual yield reduction[1] |
|---|---|---|---|---|---|---|---|---|
| Knoxville | | | | | | | | |
| 1980 | 91.5 | 65.7 | 8.37 | 15.56 | 10.89 | 4.72 | 0.58 | 0.46 |
| 1986 | 90.9 | 66.2 | 5.91 | 17.42 | 12.19 | 6.61 | 0.39 | 0.55 |
| 1988 | 89.7 | 65.9 | 6.87 | 15.93 | 11.15 | 5.67 | 0.59 | 0.57 |
| Mean | 90.7 | 65.9 | 7.05 | 16.30 | 11.41 | 5.04 | 0.52 | 0.53 |
| 1975-89 Avg. | 87.6 | 64.9 | 9.20 | 14.03 | 9.82 | 3.15 | 0.76 | |
| Crossville | | | | | | | | |
| 1980 | 88.2 | 62.0 | 4.66 | 15.03 | 10.52 | 5.67 | 0.43 | 0.41 |
| 1988 | 87.3 | 61.6 | 9.40 | 13.55 | 9.48 | 3.78 | 0.70 | 0.62 |
| 1986 | 86.1 | 63.0 | 4.54 | 14.09 | 9.86 | 5.67 | 0.46 | 0.64 |
| Mean | 87.2 | 62.2 | 6.20 | 14.22 | 9.95 | 5.04 | 0.53 | 0.56 |
| 1975-89 Avg. | 83.6 | 61.6 | 8.97 | 12.78 | 8.95 | 2.68 | 0.80 | |
| Spring Hill | | | | | | | | |
| 1988 | 93.0 | 66.3 | 5.59 | 18.62 | 13.03 | 5.67 | 0.59 | 0.54 |
| 1986 | 91.1 | 64.3 | 4.69 | 16.91 | 11.84 | 5.67 | 0.61 | 0.63 |
| 1980 | 90.6 | 66.6 | 13.29 | 16.08 | 11.26 | 1.89 | 0.97 | 0.49 |
| Mean | 91.6 | 65.7 | 7.86 | 17.20 | 12.04 | 4.41 | 0.72 | 0.55 |
| 1975-89 Avg. | 88.5 | 64.9 | 8.50 | 14.93 | 10.45 | 2.00 | 0.89 | |

[1] Actual yield reductions were based on the 10-year (1979-89) maximum yield for each county.

Table 3. Seasonal maximum air temperature, minimum air temperature, rainfall, pan evaporation, estimated potential evapotranspiration (0.7 X pan evaporation), estimated irrigation requirement, and estimated and actual maize yield reductions, for Ames Plantation, Jackson, and Martin, for the three warmest years compared to 15-year averages (1975-89).

| Year | Max Air Temp °F | Min Air Temp °F | Precip. inches | $E_{pan}$ inches | $ET_p$ inches | Irrig. required inches | Estimated yield reduction | Actual yield reduction[1] |
|---|---|---|---|---|---|---|---|---|
| Ames Plantation | | | | | | | | |
| 1980 | 94.3 | 69.9 | 4.70 | 18.12 | 12.68 | 6.61 | 0.53 | 0.32 |
| 1986 | 92.0 | 67.9 | 3.63 | 16.65 | 11.65 | 6.61 | 0.50 | 0.82 |
| 1988 | 91.7 | 67.9 | 8.34 | 14.78 | 10.35 | 1.89 | 0.93 | 0.87 |
| Mean | 92.6 | 68.6 | 5.56 | 16.52 | 11.56 | 5.04 | 0.65 | 0.67 |
| 1975-89 Avg. | 89.8 | 67.1 | 7.74 | 14.42 | 10.09 | 2.44 | 0.87 | |
| Jackson | | | | | | | | |
| 1980 | 91.7 | 71.0 | 11.93 | 17.41 | 12.19 | 2.83 | 0.92 | 0.42 |
| 1986 | 91.0 | 69.4 | 6.49 | 16.79 | 11.75 | 3.78 | 0.72 | 0.73 |
| 1977 | 90.5 | 70.2 | 8.02 | 17.29 | 12.10 | 3.78 | 0.79 | 0.56 |
| Mean | 91.1 | 70.2 | 8.81 | 17.16 | 12.01 | 3.46 | 0.81 | 0.57 |
| 1975-89 Avg. | 88.5 | 68.8 | 9.00 | 15.72 | 11.00 | 2.28 | 0.90 | |
| Martin | | | | | | | | |
| 1980 | 92.5 | 69.8 | 7.33 | 17.52 | 12.26 | 5.67 | 0.59 | 0.55 |
| 1988 | 92.0 | 66.3 | 5.82 | 16.08 | 11.26 | 5.67 | 0.58 | 0.73 |
| 1983 | 91.7 | 68.2 | 5.39 | 15.51 | 10.86 | 6.61 | 0.47 | 0.34 |
| Mean | 92.1 | 68.1 | 6.18 | 16.37 | 11.46 | 5.98 | 0.55 | 0.54 |
| 1975-89 Avg. | 89.3 | 67.3 | 7.84 | 14.98 | 10.48 | 4.36 | 0.68 | |

[1] Actual yield reductions were based on the 10-year (1979-89) maximum yield for each county.

# SURFACE WATER QUALITY TRENDS AT SEVEN STATIONS IN TENNESSEE, 1980 TO 1989

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Water-quality records from two sampling networks operated by the U.S. Geological Survey, the National Stream Quality Accounting Network and the National Hydrologic Benchmark Network, were analyzed for long-term water-quality trends at seven stations on Tennessee rivers. Trends were determined by a statistical method known as the seasonal Kendall test. Constituent-concentration data were adjusted for streamflow to preclude identifying trends in concentrations that were the result of trends in streamflow. Trend results for eight water-quality characteristics were compared with ancillary data on basin conditions.

Partial evidence for the effects of atmospheric deposition on water quality is contained in the trends for dissolved arsenic and sulfate. The decreasing trend in arsenic concentrations at all but one of the stations closely corresponds to observed declines in atmospheric arsenic in Tennessee for the period 1980 to 1989. Sulfate concentrations increased at six of the seven stations, despite decreases in sulfate deposition during the period. The increases in sulfate concentrations may, however, represent a delayed response the steady increase in sulfate deposition during the period 1965 to 1975.

The pattern of observed trends in stream nitrate and phosphorus concentrations during the period 1980 to 1989 appear to be related to the pattern of fertilizer use in the State during this period. Nitrate concentrations decreased at four stations and showed no trend at three stations. Phosphorus concentrations increased at only one station (Obion River at Obion), and showed no trend at the remaining six stations. Concentrations of dissolved oxygen, fecal coliform bacteria, dissolved solids, and suspended sediment showed no significant trend at most, if not all, of the stations.

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# GROUNDWATER AND SURFACE WATER INTERACTION IN A SHALLOW KARST SYSTEM AND IMPLICATIONS FOR CONTAMINANT TRANSPORT[1]

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Identifying a source of contamination in a karst flow system may be complicated by non-coincidental surface-water and ground-water flow boundaries, mixing of ground water from adjacent formations, and the interaction of ground water and surface water. Remediation of the affected ground water depends on the identification of all sources of contamination. The purpose of this paper is to present the findings of a tracer test conducted in a stream where complex interactions with the underlying ground-water flow system were known to exist.

The study area is located in East Tennessee within the Valley and Ridge Physiographic Province where thrust faulting has resulted in steeply inclined strata which have weathered to form the linear ridges and intervening valleys for which the province is named. Surface runoff and ground-water flow is generally from recharge areas on the the ridges to the valleys where flow parallels the longitudinal axis of the valley. Ground water discharge to the surface drainage is common at the valley floor and, in those valleys bounded by carbonate strata, ground water commonly discharges from large springs.

In upper Bear Creek Valley, Bear Creek overlies the middle to lower portion of the Maynardville Limestone, and in places the upper portion of the underlying Nolichucky Shale (Figure 1). Solution cavities are common in the Maynardville Limestone and the hydraulic connection of Bear Creek to these cavities is significant as illustrated by the discharge profile in Figure 2 which shows the creek to be alternately losing and gaining along different reaches.

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[1]The work described in this report was conducted by a contractor of the U.S.Government under Contract DE-AC04-84OR21400. Accordingly, the U.S.Government retains a nonexclusive, royalty-free license to publish or reproduce the published form of this contribution, or allow others to do so, for U.S.Government purposes.

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[3]Operated by Martin Marietta Energy Systems, Inc., for the U.S. Department of Energy under Contract DE-AC05-84OR21400.

A perennial spring, SS-5, issuing from the Maynardville Limestone was found to contain above background levels of nitrate and conductivity. The spring is located at the base of a steep north-facing ridge formed by highly solutioned dolostones of the Knox Group (Figure 1). The ridge was thought to be the recharge area for the spring; however, no sources of nitrate contamination are known to exist along the ridge. The nearest known source of nitrate was at the S-3 Waste Management Area located approximately 3 kilometers upstream near the headwaters of the creek. Different scenarios have been proposed to explain contamination of the spring, including a connection to losing reaches of Bear Creek via solution channels in the Maynardville Limestone, and undiscovered source areas on the ridge.

Spring SS-5 discharges to a large pool that forms a tributary to Bear Creek. The response of discharge at Spring SS-5 to precipitation is flashy, with measured peak flows exceeding base flow by a factor of up to 200. This suggests that flow to the spring is supplied primarily by conduits, rather than through diffuse flow.

Upstream of Spring SS-5, Bear Creek is contaminated by nitrate originating from the S-3Waste Management Area. The conductivity of Bear Creek is high as a result of the elevated levels of nitrate. The average conductivity at BCK 12.46, located at the head of Bear Creek (Figure 1), ranged between 530 and 7,260 µmhos/cm and averaged 2,820 µmhos/cm during 1987. Average conductivity at BCK 10.41, located approximately 2 kilometers downstream, ranged from 180 to 1,470 µmhos/cm and averaged 670 µmhos/cm. The conductivity of Spring SS-5 during 1987 was in the range 110 to 1,000 µmhos/cm and averaged 530 µmhos/cm, while the conductivity of an uncontaminated spring in the Knox Group averaged only 270 µmhos/cm.

A tracer test was conducted during a period of extreme low flow to determine if water from influent reaches of Bear Creek migrating via solution channels in the Maynardville Limestone was responsible for the elevated levels of nitrate in Spring SS-5. The tracer test was conducted by injecting a solution of 10,220 liters of water containing 3 grams per liter sodium chloride and fluorescein dye into the stream channel at BCK 10.41 during a period of no flow. The concentration of both tracers was below levels which might cause adverse effects to organisms living in the aquatic system where the tracer emerged. The conductivity of the tracer solution was approximately 4,000 mhos/cm. It was thought that the sodium chloride tracer would cause a measurable increase in the conductivity where it emerged. Conductivity measurements were collected at 30 minute intervals at Spring SS-5 and twice daily at a number of other stations along Bear Creek and tributary streams.

Activated charcoal detectors were also placed at several locations along Bear Creek and at Spring SS-5. The detectors were collected and replaced at one-week intervals. The detectors were eluted with a solution of potassium hydroxide and isopropyl alcohol. The solutions were then analyzed with a Turner Model III flourometer for the presence of fluorescein dye.

Fluorescein dye was visually observed at Spring SS-5 on October 18 approximately 5-1/2days after the tracer was released into Bear Creek. Based on the visual detection of the tracer, the average ground-water travel time from Bear Creek to

Spring SS-5 was approximately 180 meters/day. However, this estimate is likely to have been affected by adsorption of the tracer as it infiltrated clay-rich bottom sediments in Bear Creek channel. Thus, the travel time for non-reactive constituents should be shorter. The tracer probably traveled along the contact between stream sediments and bedrock until a solution conduit was reached, at which time the transport velocity may have increased.

Conductivity data collected at Spring SS-5 suggests that the sodium chloride tracer may have emerged on October 17. The appearance of the sodium chloride tracer is indicated by the increase in conductivity which began October 17 (Figure 3) before which, the conductivity had been gradually decreasing for a period of approximately eight days. However, a small rainfall event occurred at the same time which may have been responsible for the rise in conductivity.

Based on the large amount of dilution that the sodium chloride tracer underwent, it is thought that a second, uncontaminated source of water also discharges to Spring SS-5. The conductivity measured at Spring SS-5 was approximately 1050 µmhos/cm when the fluoroscein tracer emerged; whereas, the initial concentration of the sodium chloride tracer was 4,000 µmhos/cm. A second water source may also be suggested by the manner in which the fluorescein dye emerged at Spring SS-5. The initial emergence of the fluorescein tracer was observed in the center of the pool, indicating that either the pool is fed from a single spring located beneath the pool, or a second spring is present near the head of the pool.

The tracer test conclusively demonstrated the connection between an influent portion of Bear Creek and the Spring SS-5, thereby demonstrating that contamination in the spring was due to this influent water. However, a second source of water that is probably uncontaminated, possibly discharging from the ridge to the south, also appears to supply the spring. To conclusively demonstrate a connection between recharge areas on the ridge and Spring SS-5, a second tracer test would have to be conducted. This study demonstrated that in karst systems, springs may be fed by recirculated streamflow as well as ground-water discharge.

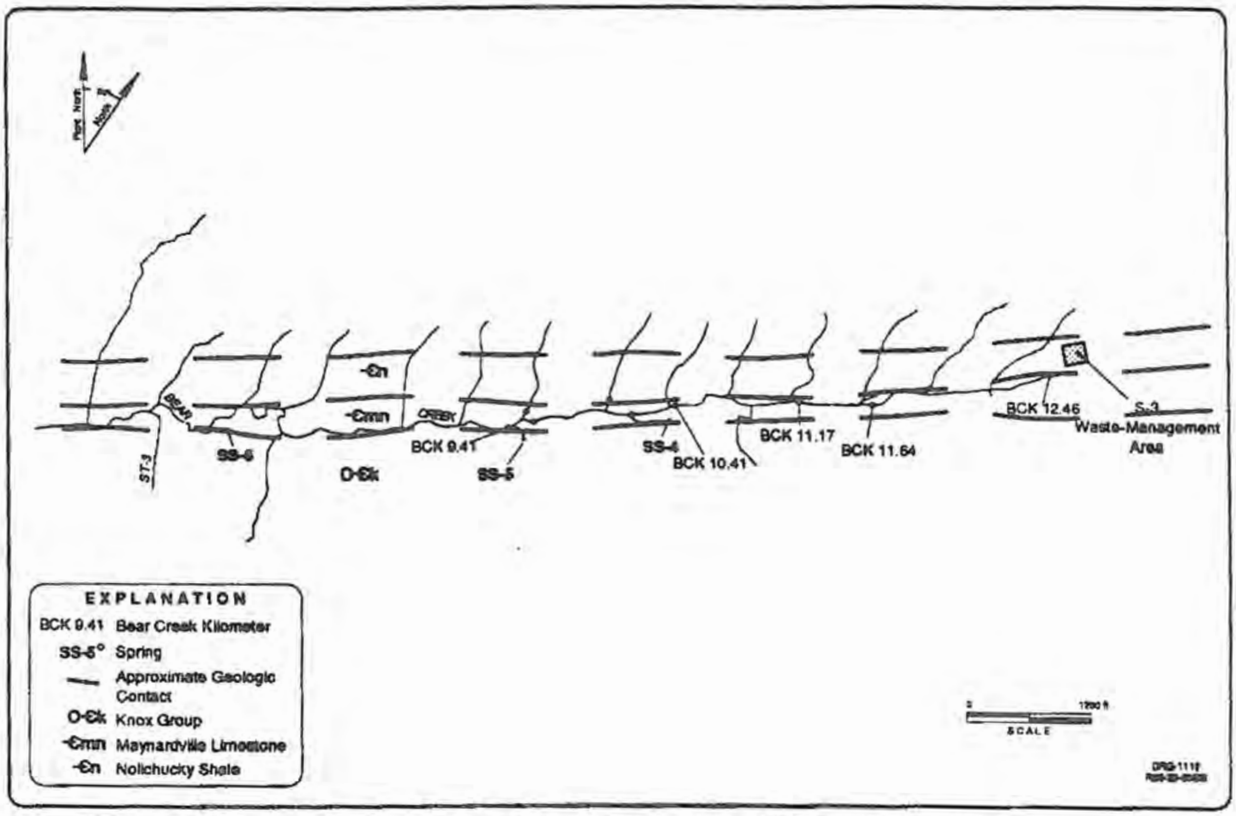


Figure 1. Bear Creek Drainage Basin

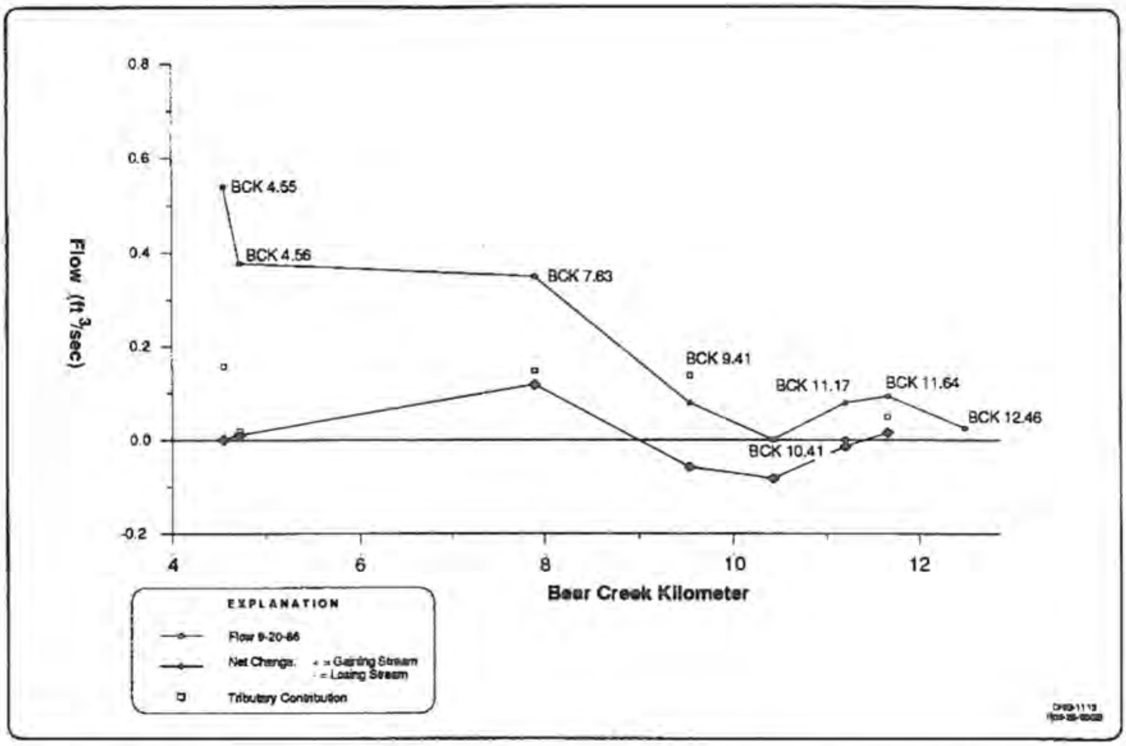


Figure 2. Discharge Profile of Bear Creek During Low-Flow Conditions

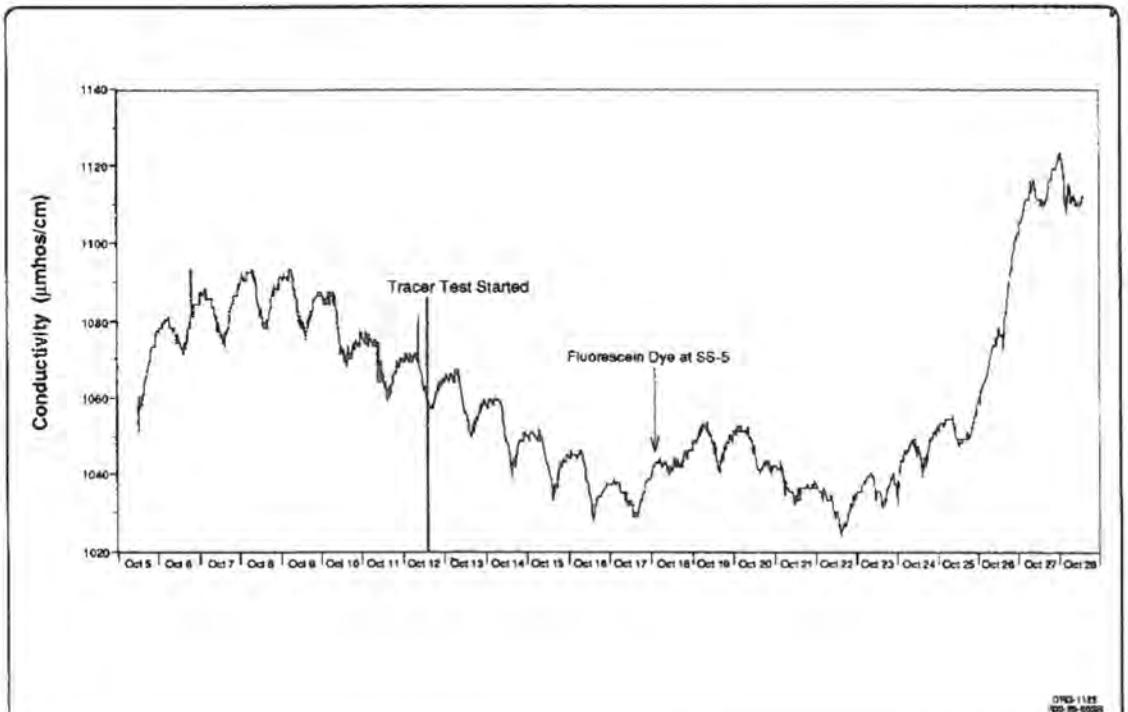


Figure 3. Conductivity at SS-5

# TIME-SERIES ANALYSIS OF SPRING WATER QUALITY FOR WELLHEAD PROTECTION PURPOSES IN EAST TENNESSEE[1]

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First Tennessee Development District
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## INTRODUCTION

Many of the public water supplies of eastern Tennessee utilize springs that emerge from cavernous limestones and dolomites. The cavernous rock does not provide significant filtration of contaminants compared to other rock types. The vulnerability of these municipal water supplies prompted the First Tennessee Development District (Matthews, 1986 and Brown, 1987) and TVA (Foxx, 1981) to make a survey of potential pollution sources within close proximity of the springs. 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 groundwater quality. Documentation of groundwater flow paths and velocities is needed to better define sources of pollution and delineate areas that need protection.

In 1976, Congress amended, the Safe Drinking Water Act to provide for the delineation of wellhead protection areas surrounding public municipal water supplies. The Environmental Protection Agency (1988) in their national wellhead protection plan recommends dye tracing as the means to determine recharge areas and travel times for karstic spring water drinking supplies. Therefore, the primary objective of this study was to define the recharge areas and groundwater flow velocities of several municipal used springs around Johnson City. In addition, monthly water samples were taken to ascertain chemical changes associated with recharge events. Utilizing this information, it is possible to distinguish springs in which groundwater flow is diffuse versus conduit (Shuster and White, 1972; Hess, 1974; Ogden, 1988). The basic premise is that conduit flow type springs are more likely to be adversely affected by contaminant spills, leaky underground storage tanks, septic tanks, or land application of agrichemicals.

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[1]This research was funded jointly by the UT-Water Resources Research Center and the TTU - Center for the Management, Utilization and Protection of Water Resources.

[2]Tennessee Technological University, Center for the Management, Utilization and Protection of Water Resources, Box 5033, Cookeville, Tennessee 38505.

## METHODS

Nine municipal-used springs were sampled monthly for one year and before, during, and after a storm event. The following springs were studied: 1) Carter County - Blue Spring, Big Spring, Hampton Springs, and Rockhouse Cave Spring; 2) Unicoi County - Johnson City Springs (2 sites); 3) Washington County - Jonesborough Spring; and 4) Hawkins County - Hamilton Spring and Lee Spring. Groundwater tracing was conducted using fluorescein dye and optical brighteners. Tracing was performed qualitatively using charcoal packets to absorb the dye and untreated cotton balls to absorb the optical brighteners. Temperature, pH, conductivity, dissolved oxygen, and ORP were measured in the field. Samples taken were analyzed in the laboratory for nitrate, sulfate, chloride, calcium, magnesium, alkalinity, TOX, MTBE, fecal coliform, and fecal streptococcus bacteria. The following discussion will present a subset of the results.

## RESULTS

Figure 1 shows the dye trace results for the Rockhouse Cave Spring which is utilized by the City of Elizabethton. Two wells have been drilled into the cave stream and submersible pumps bring the water up to the treatment facility. A dye trace was conducted from Dry Creek which sinks into its bed. The tracer was detected at the Rockhouse Cave stream, flowed through Carter Saltpetre Cave, and emerged at Taylor Spring on Buffalo Creek. Discharge measurements at Dry Creek and in Rockhouse Cave show that essentially all flow in the cave is from Dry Creek. Two other traces were conducted in the area that went to Taylor Spring via Carter Saltpetre Cave, but did not go to the Rockhouse Cave stream. Figure 1 shows the "wellhead" protection area for this water supply based on these groundwater traces.

Tracing was also successful for delineating recharge areas for Big and Blue springs. Successful tracing has not occurred for the other sites at this time.

Figures 2, 3, 4, and 5 present the results of sampling for temperature, fecal coliform, nitrate, and calcium, respectively for some of the springs. Rockhouse Cave Spring and Blue Spring show the most temperature variability. This is expected since flow is through open conduits as demonstrated by tracing results. Dye emerged in less than one week after injection. The temperature at Blue Spring is colder than the other sites. Streams sink at the base of Iron Mountain immediately upon reaching the limestone. Cold mountain runoff, short distance of surface flow, and deeper circulation are believed to contribute to the lower temperatures. The temperature at Hampton Springs is the most constant suggesting diffuse flow within the alluvium of the Doe River. The low calcium content of Hampton Springs also suggests that the source of the water is from non-carbonate rock. Similarly, Johnson City Spring, Davis Spring, and Blue Spring have low calcium content. Davis Spring is piped from Limestone Cove in Unicoi County to nearby Johnson City Spring along Indian Creek. Both springs are shown on the geology map to emerge from the Shady Dolomite as does Blue Springs. This could account for the significantly lower calcium levels than Big and Rockhouse springs which emerge from the Knox Group. Davis and Johnson City springs likely receive significant recharge from flow through Pre-Cambrian crystallines and Cambrian sandstones and their associated colluvium. The thick colluvium covers all outcrops of the Shady Dolomite near the springs.

Fecal coliform bacteria and nitrate levels were lowest for Davis and Johnson City springs since recharge occurs from undeveloped National Forest Service land on Unaka Mountain. Low turbidity of these springs during storms suggests diffuse flow. Davis Spring become slightly turbid during large storms which may be a result of recharge from alluvium along Indian Creek since the spring is less than fifty feet from the creek bed and is in the flood plain.

Big and Rockhouse Cave springs have the highest nitrate levels, although far below the 10 mg/L health limit. Sinking water from Gap Creek supplies a significant amount of recharge to Big Spring, as demonstrated by a successful dye trace. Many deep sinkholes and housing developments with septic tanks occur within the recharge areas for both springs. Higher levels of fecal coliform bacteria are also commonly found in these springs compared to the others.

## CONCLUSIONS

Dye tracing has proven an effective tool in delineating "wellhead" protection areas for springs where significant surface karst has developed. A time-series analysis of spring water quality enables further distinction of diffuse versus conduit flow types. Low variability of measured parameters are indicative of diffuse flow systems. This information will be useful to the cities for protecting recharge areas, searching for sources of contamination, responding to highway spills, and preparing to meet new U.S. EPA guidelines requiring filtration of springs rapidly affected by surface waters.

## REFERENCES

Brown, T., 1987. Survey of Public Ground Water Supplies. First Tennessee Development District, Johnson City, TN, 217 p.

Foxx, Suzanne A., 1981. A Summary of the Geohydrology and of the Spring Recharge Zones within the Tennessee Portion of the 208 Study Area of the First Tennessee-Virginia Development District. Report No. WR28-1-520-116, TVA Office of Natural Resources, Division of Water Resources.

Hess, J. W., 1974. Hydrochemical Investigations of the Central Kentucky Karst Aquifer System: Office of the Water Research and Technology, NTIS Pub. Pb-238, V. 366, 218 p.

Matthews, M. R., 1986. Groundwater Situation Assessment of the Tennessee Valley Region: TVA Office of Natural Resources and Economic Development, Division of Air and Water Resources, Chattanooga, TN.

Ogden, A. E., 1988. Distinguishing Flow Regimes of Springs in Carbonate Rock Terranes: Proc. Natl. Water Well Assoc. Conf. on Ground Water Chemistry, Dublin, Ohio, pp. 53-73.

Shuster, E. T. and W. B. White, 1972. Source Areas and Climatic Effects in Carbonate Groundwaters Determined by Saturation Indices and Carbon Dioxide Pressures. Water Resources Res., 8:1006-1073.

U.S. Environmental Protection Agency (EPA), 1988. Guidelines for Delineation of Wellhead Protection Areas. EPA Office of Ground Water Protection, Washington, D.C.

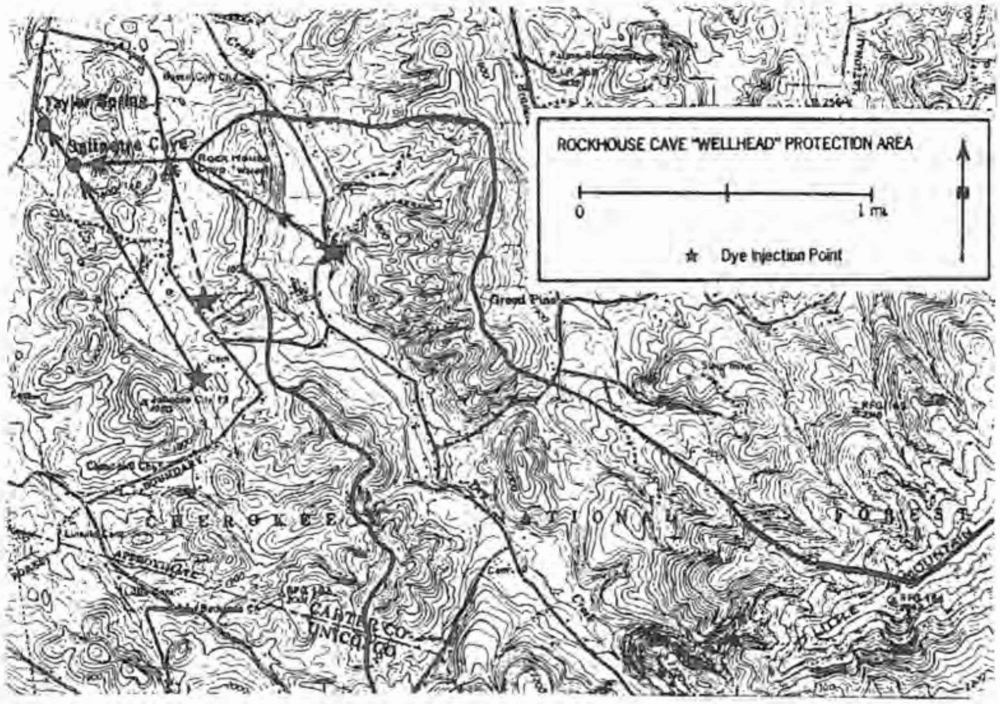


Figure 1. Dye trace results for Rockhouse Cave.

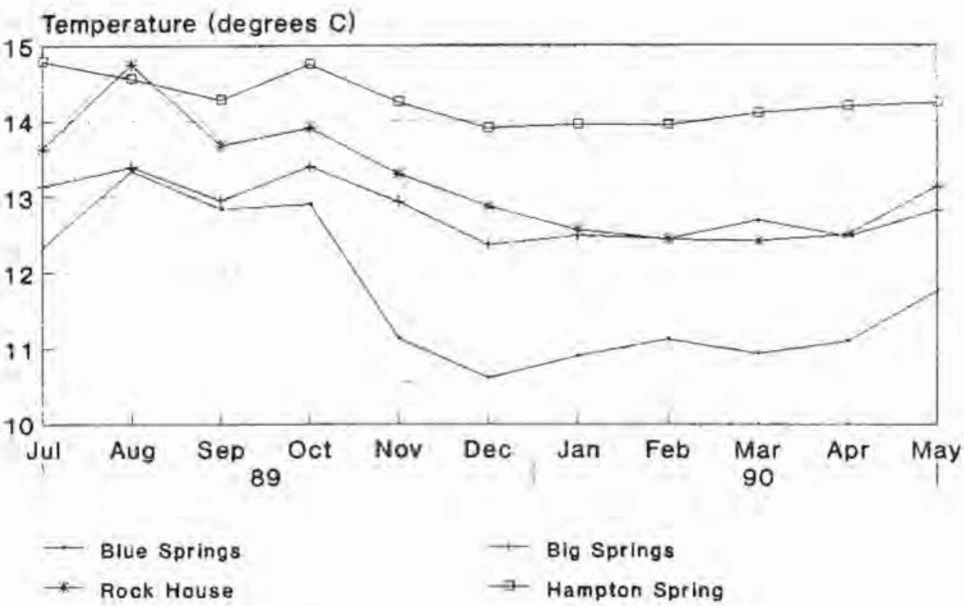


Figure 2. Temperature versus time.

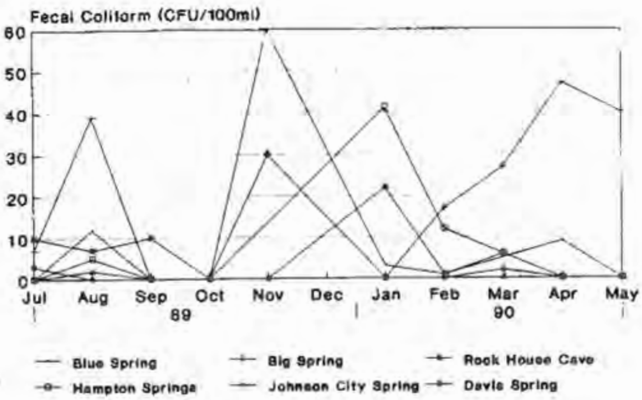


Figure 3. Fecal coliform bacteria levels versus time.

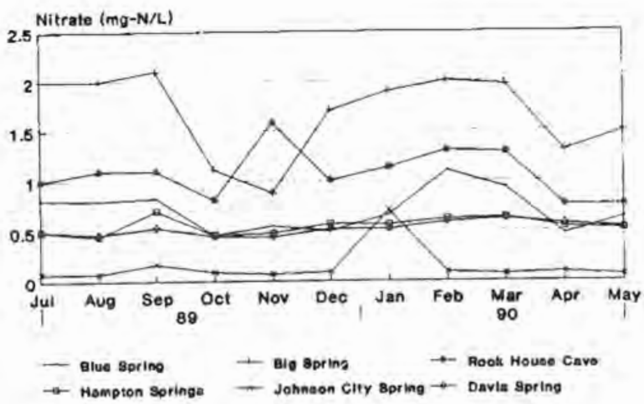


Figure 4. Nitrate concentrations versus time.

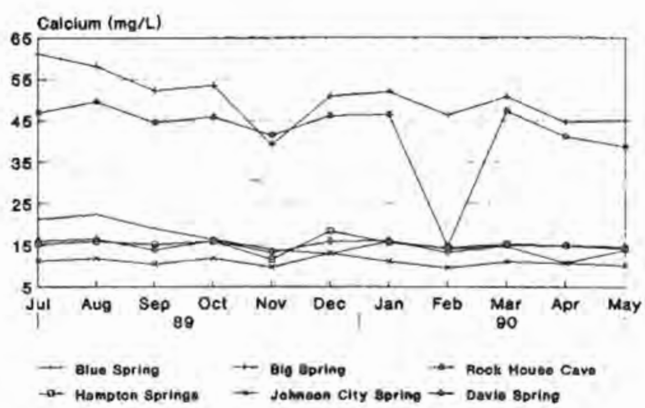


Figure 5. Calcium concentrations versus time.

# RADON, RADIONUCLIDES, AND GEOCHEMISTRY OF GROUNDWATER IN HICKMAN AND MAURY COUNTIES

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The U.S. Geological Survey, in cooperation with the Tennessee Department of Health and Environment, conducted a reconnaissance-level investigation of groundwater geochemistry including radon-222 and selected other radionuclides in Hickman and Maury Counties, Tennessee. Devonian-age black shale and Ordovician-age phosphatic limestones in these two counties contain uranium and are potential sources of radionuclides in groundwater.

Water samples from twenty wells or springs in each county were collected for chemical and radionuclide analyses in 1989. All radionuclide concentrations in these water samples were less than the U.S. Environmental Protection Agency drinking water standards. Radon-222 concentrations determined using the Lucas cell technique ranged from 85 to 1,486 picocuries per liter. Most of the waters samples were calcium bicarbonate type although some were calcium sulfate type waters. Concentrations of dissolved solids ranged from 26 to 1,160 milligrams per liter, and were generally higher in water samples from phosphatic limestones than in waters associated with black shales. Little correlation existed between radionuclide concentrations and major ion concentrations or physical properties of the waters samples.

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# DESCRIBING CONTAMINANT-TRANSPORT IN KARST AQUIFERS

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## INTRODUCTION

Karst aquifers are particularly vulnerable to contamination by accidental spills and poor waste-management practices. A thorough understanding of the contaminant-transport characteristics of karst ground-water flow is needed for development and application of effective well-head and spring-head protection programs. Use of quantitative dye-tracing techniques can contribute significantly to this understanding. These techniques were evaluated and refined during a hydrologic study of a mature karst aquifer in the Elizabethtown area, Hardin County in central Kentucky (Mull, Smoot, and Liebermann; 1988).

Water supply for the study area is obtained from springs and wells that derive water from a nearly horizontal limestone aquifer. The area, typical of much of the karst in Tennessee and Kentucky, is characterized by thin soils, numerous sinkholes, springs, karst windows, and losing and sinking streams (Mull and Lyverse, 1984). Potentially, contaminants of many types may be introduced to the aquifer from activities associated with a variety of residential, commercial, industrial, and agricultural land uses and major rail and highway transportation corridors.

Because a conservative water-soluble tracer behaves in the aquatic environment much the same way that a conservative water-soluble contaminant does, much can be learned from introducing a harmless tracer to a karst aquifer in ways to simulate contaminant spills during various hydrologic conditions and monitoring its arrival at locations used for water supply such as springs and wells. Quantitative dye tracing procedures provide tools for understanding, analysis, and interpretation from which management strategies and engineering solutions can be developed.

## METHODS DEVELOPMENT

During the study, an underground connection was identified between a sinkhole, that receives surface runoff from several types of land uses, and a water- supply spring using simple qualitative dye tracing (identifying point-to-point flow connections). Seven repeat quantitative traces were made between these points over a wide range of flow conditions using rhodamine-WT dye to simulate inputs

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from a chemical spill or non-point contamination during storm runoff under avariety of wet and dry antecedent conditions. Each of the quantitative traces generally followed these steps:

1. Measure accurately a known quantity of the dye for injection.

2. Introduce dye to the flow path in the manner desired and record time.

3. Sample automatically or manually at all known or suspected outlets (wells and springs) at short time intervals and record time of each sample before, during, and after passage of dye cloud (a minimum of 15-25 samples to define the time-concentration plot of the dye passage). Samples must be collected so as to closely represent the mass of dye moving through the cross section of flow. Sampling at the mid-point of flow is usually done, but this assumes that nearly complete cross-sectional mixing of the dye has occurred. In practice, mixing is frequently incomplete. Thus, samples must be collected from three or more points in the cross section in order to insure representative cross-sectional sampling.

4. Measure accurately the flow rate from each well and spring sampled before, during, and after passage of dye cloud for dye mass computation and flow-condition characterization.

5. Determine the background concentration or fluorescence of the dye.

6. Determine and record the dye concentration in each sample using temperature-equilibrated standards and samples and a calibrated fluorometer.

7. Plot the time-concentration (breakthrough or response) curves for each sampled outlet.

8. Measure the area under the breakthrough curve and multiply by the average flow rate of that outlet to determine the quantity of dye recovered from that outlet.

9. Normalize the ordinates of the breakthrough curve for a unit quantity injected and for dye conservancy (assuming that dye not recovered either decayed or was adsorbed to aquifer material).

10. Compute centroid of the breakthrough curve (from which the mean travel time and apparent groundwater velocity can be determined) and the dispersion coefficient.

11. From the breakthrough curve, determine characteristics such as elapsed time to arrival of leading edge, trailing edge, peak concentration, and peak concentration.

The time-concentration response typically gives a right-skewed, bell-shaped curve that is steeper on the rising limb (left side) than on the falling limb (right side). The falling limb or tail portion of the response curve is longer and flatter than the leading edge due to dispersion properties of the flow. The dispersion properties and hence the shape of the time-concentration curve commonly vary for different locations and under different flow conditions (Hubbard, and others, 1982). Thus, repeat dye traces between the same injection and recovery points are desirable for determining the variation in the time-concentration curve under different flow conditions (Mull, Smoot, and Liebermann; 1988).

Normalized peak concentration, mean travel time, and standard deviation of travel time were computed for each of the breakthrough curves and were determined to be closely related to the flow of the outlet by use of statistical regression. The coefficient of determination (R-square) for the three regressions were in the range 0.739 to 0.995, indicating good regression fits.

The shapes of the time-concentration curves at the outlet resulting from dye releases at the injection sinkhole under different flow conditions were similar. However, because the scales of each were different the curves could not directly be compared or combined. Therefore, the abscissa and ordinate of each was first made dimensionless so the curves could be compared and composited. The ordinate was converted to the ratio of dye concentration to peak dye concentration and the abscissa was converted to the ratio of the elapsed time from dye injection minus mean travel time to the standard deviation of travel time. The curves were then composited into one dimensionless type curve which was the average of the others. This curve could then be used to simulate the dye breakthrough curve (or contaminant breakthrough curve) for flow conditions other than those prevailing during the traces. The time and concentration scales of the curve can be estimated, for any selected outlet flow rate, from the previously described regression relations between normalized peak concentration, mean travel time, and standard deviation, with flow rate. This synthesized curve of concentration versus time could be used to estimate arrival time, peak concentration, and persistence of dye (or a spilled contaminant) at the outlet (Smoot and others, 1989).

Repeat dye traces from the injection sinkhole to the outlet, a straight-line distance of about 914 meters (3,000 feet), showed that the arrival time of the leading edge of the dye cloud ranged from 24 to 5 hours and that the mean travel time given by the centroid of the dye cloud, ranged from 31 to 6 hours for a range of discharge from 0.015 and 0.13 cubic meters per second (0.53 and 4.6 cubic feet per second), respectively.

## EXAMPLE APPLICATION

To illustrate the application of quantitative dye tracing techniques for water management purposes using the previously studied karst connections, assume that 57 liters (15 gallons) of a 5-percent copper sulfate solution is accidentally spilled and enters the sinkhole that drains to the outlet. In order to respond to the spill, the water manager needs to know the effect at the outlet, which supplies public drinking water. Copper sulfate is a common agricultural chemical; the quantity spilled contained 1.13 kilograms of copper. Assume that the drinking-water supply quality criteria for copper is 1.0 milligram per liter and that the flow at the outlet for this example is determined to be 0.025 cubic meters per second (0.9 cubic feet per second). From the statistical regressions developed, the following values are estimated: mean travel time, 20.1 hours;

standard deviation of travel time, 3.67 hours; normalized peak concentration, 2.31 milligrams per liter per kilogram of contaminant. Because 1.13 kilograms of copper was spilled, the estimated peak concentration is 2.61 milligrams per liter. Using these values in conjunction with the dimensionless type curve, the time-concentration response to the spill at the outlet can be simulated. On the basis of this simulation, the leading edge of the contaminant mass will arrive about 14 hours after the spill, reach peak concentration of 2.6 milligrams per liter 4 hours later, and decrease to 2-percent of the peak concentration 16 hours later. The contaminant is virtually nondetectable at the spring about 40 hours after the spill. From the simulated curve, the drinking-water supply quality criteria would be exceeded for approximately 4.8 hours, beginning 16 hours after the spill. These values are only estimates, but these techniques may be especially useful to a water manager in areas where public water is supplied from karst springs or wells.

## SUMMARY AND CONCLUSIONS

Dye tracing is a useful tool for better understanding of the contaminant transport characteristics of ground-water flow in karst aquifers. Quantitative analysis of dye tracing results can provide interpretive and predictive information that is not readily obtainable using other techniques. This information is especially useful to water managers who need to respond to the introduction of potential contaminants into the ground-water system. The maximum benefit from this type of dye tracing is most likely realized, however, when performed as an adjunct to detailed geologic, hydrologic, and topographic investigations of a study area.

## REFERENCES

Aley, T. and M. W. Fletcher, 1976. The Water Tracers Cookbook: Missouri Speleology, 16(3):1-32.

Fischer, H. B., 1968. Dispersion Predictions in Natural Streams: Journal of the Sanitary Engineering Division, American Society of Civil Engineers, 94(SA5):927-943.

Hubbard, E. F., F. A. Kilpatrick, L. A. Martens, J. F. Wilson, Jr., 1982. Measurement of the Time of Travel and Dispersion in Streams by Dye Tracing: U.S. Geological Survey Techniques of Water-Resources Investigation Book 3, chapter A9, p.44.

Kilpatrick, F. A. and E. D. Cobb, 1985. Measurement of Discharge Using Tracers. U.S. Geological Survey Techniques of Water-Resources Investigation Book 3, chapter A16, p.52.

Mull, D. S., T. D. Liebermann, J. L. Smoot, L. H. Woosley, Jr., 1988. Application of Dye-Tracing Techniques for Determining Solute-Transport Characteristics of Ground Water in Karst Terranes. U.S. Environmental Protection Agency, EPA 904/6-88-001, p.103.

Mull, D. S. and M. A. Lyverse, 1984. Ground Water Hydrology of the Elizabethtown Area, Kentucky. U.S. Geological Survey Water-Resources Investigation Report 84-4057, p.59.

Mull, D. S., J. L. Smoot, and T. D. Liebermann, 1988. Dye Tracing Techniques Used to Determine Groundwater Flow in a Carbonate Aquifer System Near Elizabethtown, Kentucky. U.S. Geological Survey Water-Resources Investigations Report 87-4174, p.95.

Smart, P. L. and I. M. S. Laidlaw, 1977. An Evaluation of Some Fluorescent Dyes for Water Tracing. Water Resources Research, 13(1):15-33.

Smoot, J. L., D. S. Mull, and T. D. Liebermann, 1989. Quantitative Dye-Tracing of Karst Ground-Water Flow. In: Environmental Engineering, Proceedings of the 1989 Specialty Conference, American Society of Civil Engineers, New York, New York, pp.677-684.

# MANAGEMENT OF HERBICIDE PLACEMENT FOR NONPOINT POLLUTION REDUCTION IN THE PRESENCE OF CROP RESIDUE

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## INTRODUCTION

No-till agriculture, a cultural practice in which the soil is not tilled and residues of previous crops are left on the soil surface when a new crop is planted, has been shown to be very effective in reducing erosion and associated sediment pollution (Shelton and Bradley, 1987; Shelton et al., 1983; Larson, 1981; Phillips et al., 1980). However, elimination of tillage makes effective use of herbicides for weed control essential; and some studies have suggested that off-site movement of agricultural chemicals when reduced tillage is used may be greater than when conventional tillage practices are followed (Maas, et al., 1984).

Some herbicides currently used in no-till agriculture, metolachlor for example, are water soluble and, thus, have the potential of moving off-site with water. Metolachlor itself is one of the most widely used herbicides in the United States [only four pesticides are used in larger quantity (Anonymous, 1987)], and the chemical has been found in groundwater samples (Ritter, 1986). There is, therefore, a need to understand the impact of crop production practices on the mobility of metolachlor.

Plant residues on the soil surface when no-till practices are followed interfere with placement of herbicides and may impact their off-site movement. Banks and Robinson (1986) have shown that the quantity of applied metolachlor reaching the soil surface varies inversely with the quantity of wheat straw on the surface. Their data suggest that a substantial amount of metolachlor may bind to the straw and become unavailable for release to the soil. Response of metolachlor on wheat straw to off-site movement forces is not well understood and may be considerably different from that of metolachlor on soil.

Application of herbicide beneath the plant residue in no-till operations should overcome much of the problem of residue interference. Conceivably, a chemical application machine could be developed that would remove straw from the surface, apply chemical, and redeposit the vegetative material in a once-over operation.

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## OBJECTIVES

Objectives of the project described herein were to:

1. compare the impact of no-till and conventional tillage practices on the off-site movement of metolachlor, and

2. compare the application of herbicide beneath and over-the-top of wheat straw from the standpoint of water-borne off-site movement of metolachlor.

## MATERIALS AND METHODS

The study was conducted with soil columns in the laboratory. Nine 254-mm diameter monolithic soil cores were taken in close proximity from an agriculturally productive field site. Cores were obtained by forcing sections of PVC pipe 600 mm deep into the soil with a tractor-powered hydraulic cylinder and extracting the pipes so that a minimally-disturbed soil core was retained inside each pipe section (hereafter referred to as columns).

After being transported to the laboratory, each column was equipped with the necessary flow lines for collection and segregation of all surface runoff and deep seepage. Provisions were made for collecting the deep seepage under tension. The columns were then mounted in a support rack under a rainfall simulator. The support rack held the columns at a slight angle from vertical to accommodate collection of surface runoff.

The soil surface in three columns was tilled and left bare. Surfaces of the other six columns were not tilled. Metolachlor was applied to the six columns with bare soil surfaces (three tilled and three untilled) at a rate of 2.2 kg of active ingredient per hectare. The six untilled column surfaces were then covered with wheat straw at the rate of 4480 kg per hectare and metolachlor was applied over the top of the straw in the three columns which had not previously received herbicide.

All columns were subjected to simulated rainfall events with a mean intensity of 26.5 mm per hour for either 2.5 or 3 continuous hours (two events had durations of 3 hours and four events had durations of 2.5 hours). The rainfall events occurred at 4, 48, 168, 504, 1008, and 2016 hours after the one-time chemical application. Surface runoff and deep seepage from each column was collected and analyzed by established protocols for detection and quantification of metolachlor.

## RESULTS

Metolachlor was found to be mobile, and a small amount of it moved off-site in water from both runoff and deep seepage. From all columns, the total off-site movement of metolachlor ranged from a low of 1.6% to a high of 7.9% of that applied to the columns. As can be seen from the observed concentrations presented in Figure 1, the concentration of metolachlor in water leaving all columns was low. The highest concentrations were produced by the rainfall event occurring four hours after chemical application. For the 4-hr rainfall event, only 8 of the 18 samples had metolachlor concentrations above the Lifetime Health Advisory (LHA) value of 0.1 mg/liter (EPA, 1988). The rainfall event at 48 hours after application produced only one sample with a concentration above the LHA, and no concentrations were higher than the LHA after 48 hours.

Analysis of variance indicated that there were no significant differences in the observed metolachlor concentrations in water from deep seepage among the three treatments. There were, however, significant differences in concentrations in runoff water among the three treatments (see Table 1). Metolachlor concentrations were greatest in runoff from columns where the herbicide was applied over-the-top of straw, and lowest from columns where it was applied to bare tilled soil.

There was also a significant interaction at the 0.032 probability level in the runoff concentration data between surface treatment and time. The plot of mean metolachlor concentration values against time for the three surface treatments in Figure 2 show that the response from columns where the herbicide was applied over-the-top of the straw was different from the response from other columns. Between the 48-hr and 168-hr rainfall events, the metolachlor concentration in runoff from the herbicide-over-straw treatment did not decline as it did in runoff from the straw-over-herbicide treatment. Therefore, in addition to there being more metolachlor in runoff from the herbicide-over-straw treatment, the period of time for off-site movement to occur was also extended.

Table 1. Rank mean values[2] for metolachlor concentrations in deep seepage and runoff water from columns with three different surface treatments.

| Surface Treatment | Deep Seepage Rank Mean[3] | Runoff Rank Mean[4] |
|---|---|---|
| Bare Soil | 41.5 | 33.8 |
| Herb. Over Straw | 41.7 | 50.2 |
| Straw Over Herb. | 39.6 | 39.4 |

[2]A greater rank mean value indicates a greater number of observations for which a greater metolachlor concentration was observed.

[3]Probability that differences of the magnitude observed among means could have occurred solely by chance is 0.856.

[4]Probability that differences of the magnitude observed among means could have occurred solely by chance is 0.004.

## CONCLUSIONS

Results of this study suggest that:

1. No-till practices may lead to increased concentrations of herbicide in surface runoff from agricultural land, and

2. Application of herbicide beneath the residue of previous crops in no-till situations may significantly reduce both the concentration of herbicide in runoff and the time period during which off-site movement of the herbicide is likely to occur.

## REFERENCES


Anonymous, 1987. Total National Pesticide Usage by Active Ingredient. U.S. Water News, 4(2):7.

Banks, P. A. and E. L. Robinson, 1986. Soil Reception and Activity of Acetochlor, Alachlor, and Metolachlor as Affected by Wheat (Triticum Aestivum) Straw and Irrigation. Weed Science, 34:607-611.

U.S. Environmental Protection Agency (EPA), 1988. Metolachlor Health Advisory. Office of Drinking Water, Washington, D.C.

Larson, W. E., 1981. Protecting the Soil Resource Base. J. Soil Water Cons., 36(1):13-16.

Maas, R. P., S. A. Dressing, J. Spooner, M. D. Smolen, F. J. Humenik, J. W. Meek, and F. N. Swader, 1984. Best Management Practices for Agricultural Nonpoint Source Control - IV. Pesticides. Washington, DC.

Phillips, R. E., R. L. Blevins, G. W. Thomas, W. W. Frye, and S. H. Phillips, 1980. No-tillage Agriculture. Science, 208:1108-1113.

Ritter. W. F, 1986. Pesticide Contamination of Groundwater - A Review. Paper No. 86-2028, ASAE, St. Joseph, Michigan.

Shelton, C. H. and J. F. Bradley, 1987. Controlling Erosion and Sustaining Production with No-till Systems. Tennessee Farm and Home Science, 141:18-23.

Shelton, C. H., F. D. Tompkins, and D. D. Tyler, 1983. Soil Erosion from Five Soybean Tillage Systems. J. Soil Water Cons., 38(5):425-428.

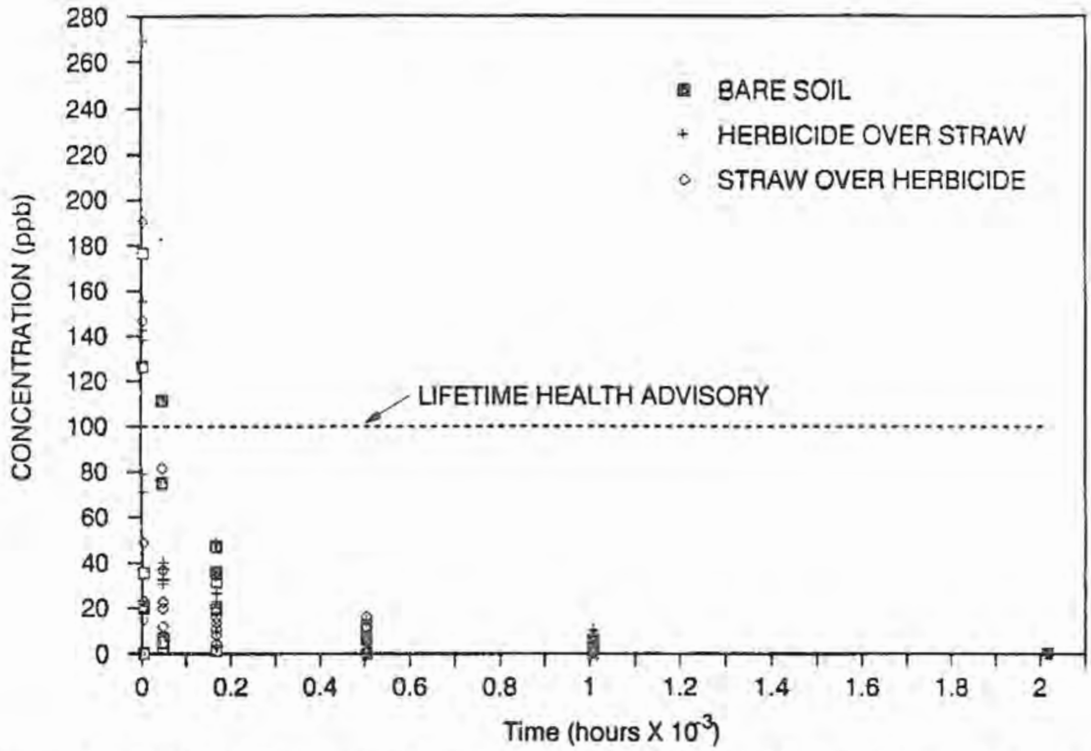


Figure 1. Metolachlor concentrations observed in all runoff and deep seepage samples

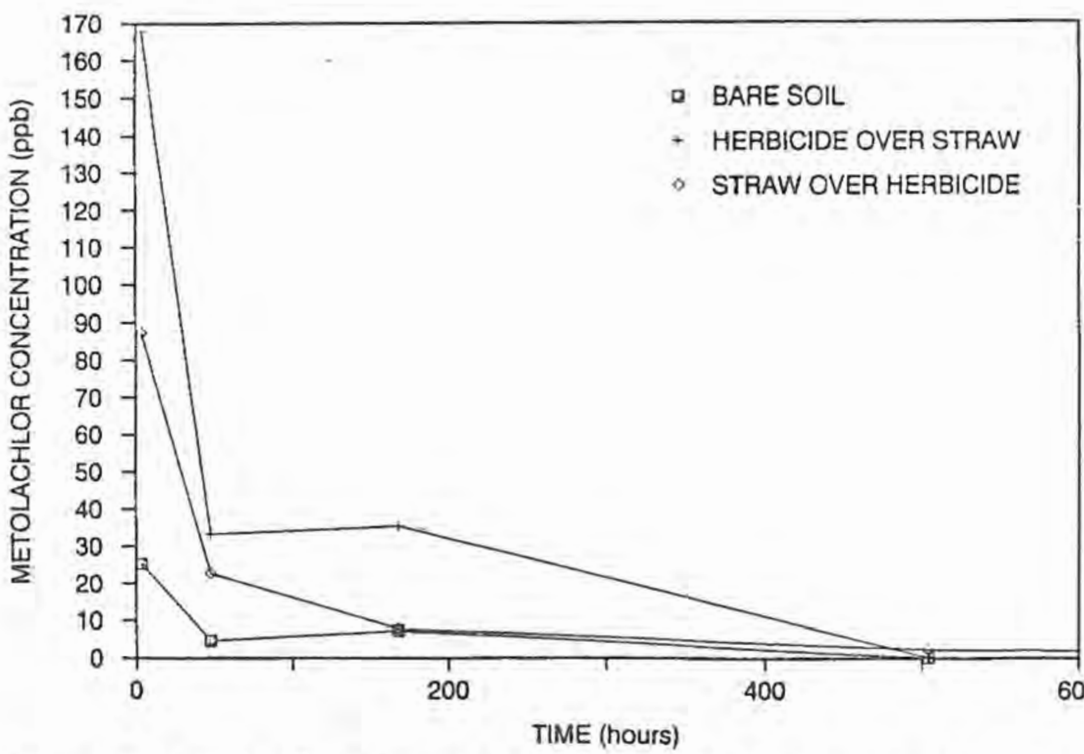


Figure 2. Mean metolachlor concentration in runoff as a function of time after chemical application

# THE OCCURRENCE AND CHARACTERIZATION OF DENSE NONAQUEOUS PHASE LIQUIDS IN BEAR CREEK VALLEY NEAR OAK RIDGE, TENNESSEE[1]

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In January 1990, accumulations of dense, nonaqueous phase liquids (DNAPLs) were discovered in a newly-installed monitoring well (GW-625) at depths of approximately 274 ft below ground surface along the southern border of Burial Ground A-South (Fig. 1) within the Bear Creek Burial Grounds Hazardous Waste Disposal Unit (BCBG) at the U. S. Department of Energy's Oak Ridge Y-12 Plant. Subsequent to this discovery, a preliminary investigation was initiated to obtain information on the mode of occurrence and distribution of the DNAPLs in fractured rock, such as that underlying BCBG. The investigation included installation and sampling of two additional groundwater monitoring wells (GW-628 and GW-629), resampling of existing wells in the vicinity of the discovery well, evaluation of existing groundwater quality data, and review of available information and site conditions.

Movement of DNAPLs in fractured rocks is complex and difficult, if not impossible, to quantify with certainty. Depending on fracture patterns within a rock mass, the pathways of downward movement of DNAPLs may be direct or indirect. At the BCBG site, the pathways are likely indirect. Given the complex geology of the study area and uncertainty as to the quantities and location(s) of DNAPL release(s), the pathways of DNAPL migration and the present subsurface configuration of DNAPL accumulations cannot be determined.

Movement of DNAPLs is driven by gravity in a downward direction, and, as migration proceeds, the body of migrating DNAPL decreases due to loss of material as it becomes entrained and trapped in fractures. Downward migration continues until the DNAPL body has become sufficiently depleted so that it no longer has adequate head to overcome surficial tension and move into fractures. A trail of residual material is left along the pathway of the migrating DNAPL body. The remaining DNAPL body becomes bifurcated into numerous smaller occurrences at

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[1]Research sponsored by the Environmental Surveillance Section of the Environmental Monitoring Department at the Oak Ridge Y-12 Plant, operated by Martin Marietta Energy Systems, Inc. under contract DE-AC05-84OR21400 with the U. S. Department of Energy.

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depth. The depth to which a DNAPL body sinks cannot be predicted with certainty for a site as complex as the BCBG.

Because of the complex behavior of DNAPLs in fractured rocks, only rarely are accumulations of DNAPLs encountered in the subsurface. Samples obtained from wells GW-625 and GW-628, however, provide direct evidence for the occurrence of DNAPLs at depths of at least 270 ft below ground surface. Groundwater samples from these two wells consist of an upper, aqueous phase containing tetrachloroethene (PCE; 930 to 2,500 mg/L), trichloroethene (TCE; 70 to 110 mg/L), and polychlorinated biphenols (PCB; 2.8 to 11 mg/L), and a lower, oil-like phase containing PCE (450,000 to 530,000 mg/L), TCE (11,000 to 19,000 mg/L), and PCB (14,000 to 27,000 mg/L).

Accumulations of DNAPLs in the subsurface generate dissolved contaminant plumes within groundwater. Detection and characterization of such plumes typically provide the only reliable means of determining the extent of DNAPL contamination at a site. The predominance of PCE and TCE among volatile organic compounds in groundwaters from two existing monitoring wells (GW-71 and GW-117), and the occurrence of low concentrations of PCBs in one of them (GW-117) indicate that these wells may sample dissolved plumes associated with subsurface DNAPL occurrences. Such groundwater chemical data, together with the existence of an upward vertical hydraulic gradient at the site and the locations of these wells, suggest that DNAPL occurrences may extend to ~500 ft below ground surface at the BCBG site.

Migration of DNAPLs occurs immediately upon their release into the environment. However, given the history of BCBG, where DNAPLs were disposed of a number of years ago, DNAPL migration is not likely to be occurring at present. Such a static situation will remain as long as the hydrogeologic system in the immediate vicinity of the DNAPL occurrences is not hydraulically disturbed. If the system is disturbed, DNAPL accumulations could become remobilized. The occurrence of approximately 10 ft of DNAPL within well GW-625 indicates that substantial accumulations of DNAPLS do exist in the subsurface. The static nature of the system and the occurrence of DNAPL pools in the subsurface, combined with the complex hydrogeology of the site require that future characterization and remediation activities at the site be undertaken with care and understanding of all factors that can influence DNAPL migration.

## REFERENCES

Haase, C. S. and H. L. King, 1990. Report and Preliminary Assessment of the Occurrence of Dense, Nonaqueous Phase Liquids in the Bear Creek Burial Grounds Hazardous Waste Disposal Unit at the Oak Ridge Y-12 Plant. Y/TS-629.

King, H. L. and C. S. Haase, 1989. Results and Preliminary Interpretation of Hydrogeologic Packer Testing in Core Holes CH-157, GW-131, and GW-132 through GW-135 at the Oak Ridge Y-12 Plant. Y/TS-495.

Kueper, B. H., 1990. The Occurrence of Dense, Nonaqueous Phase Liquids in Fractured Rock at the Bear Creek Burial Grounds Waste Management Area. (Consultants Report submitted to Oak Ridge National Laboratory/Martin Marietta Energy Systems, Inc. on February 23, 1990).

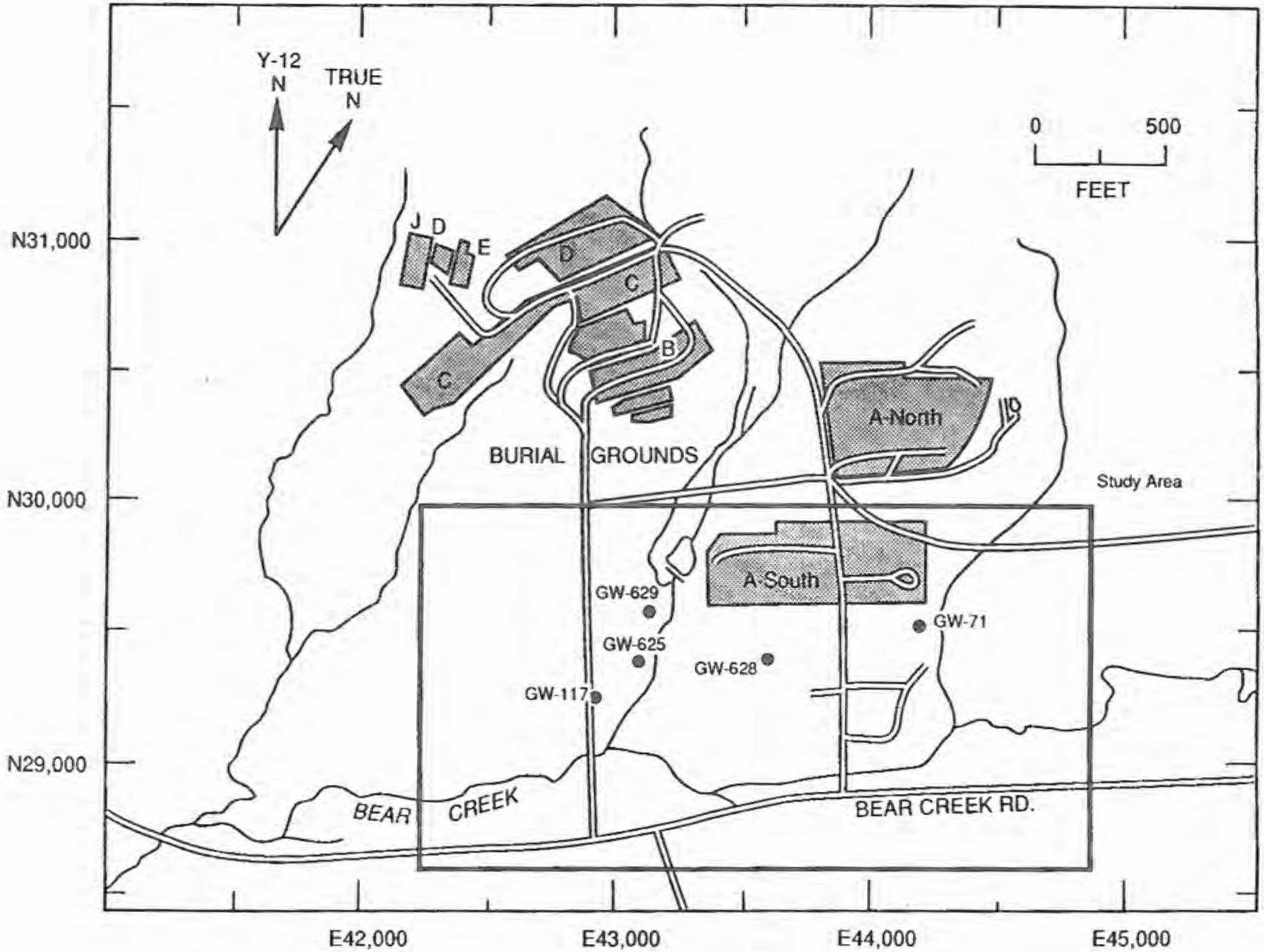


Figure 1. Map of study area showing location of selected groundwater monitoring wells from which direct or indirect evidence of DNAPL accumulations was obtained.

# POTENTIAL GROUNDWATER TRANSPORT OF CHLORDANE AT A HAZARDOUS WASTE SITE

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## INTRODUCTION

Hollywood Dump is a closed municipal/industrial landfill located in north Memphis on the alluvial plains and abandoned channels of the Wolf River (Figure 1). Elevated levels of the insecticide, chlordane (up to 100mg/Kg), have been found in the soils and sediments in the vicinty of the site. In addition, some species of fish from the abandoned dredge ponds have chlordane concentrations of up to 25mg/Kg, far in excess of the FDA 'action level' of 0.3mg/Kg. The general direction of the ground-water flow is radially away from the dump but towards the river. During periods of high flow, there is considerable inundation of the dump from the Wolf River followed by a 'flushing out' as water levels recede, from the dump back into the Wolf River.

As an organochlorine, chlordane, (1, 2, 4, 5, 6, 7, 8, 8-octa chloro-2, 3, 3a, 4, 7, 7a hexahydro-4, 7-methano-indane) is highly stable in the environment with a high octanol-water partition coefficient, log Kow = 5.58 (1). In the sub-surface environment, the mobilization and transport of chlordane will be controlled by its partitioning between the aqueous phase (mobile) and the solid phase (immobile). Such simple partitioning is more complex in the presence of humic substances in the aqueous phase and organic matter associated with the solid phase. In the following study, batch solubility and sorption experiments in combination with column studies were used to determine the potential for chlordane mobilization and transport at the site.

## METHODS

Chlordane solubility and the effect of aqueous humic substances on solubility were determined by coating excess [14C] chlordane to the inside of teflon centrifuge tubes and shaking gently in solutions of different organic carbon concentration. Humic acid was extracted from native geologic material in 0.5N sodium hydroxide, followed by acid precipitation, centrifugation and freeze drying of the solid material. Commercial humic acid was purified in a similar manner using 0.1N sodium hydroxide.

Because of low aqueous solubility and significant adsorption of chlordane onto container walls, sorption experiments were conducted in the presence of a co-solvent. Soils were shaken for 24hr at constant temperature, in solutions of

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1 - 20% methanol in purified water, spiked with 10uL [14C] chlordane. Following centrifugation at 12,000g for 30mins, distribution coefficients (Kd) were determined by comparing the ratio of chlordane in the solid and liquid phases. Kd values of the pure aqueous phase were determined by extrapolation of the plot log Kd vs. fraction of the co-solvent.
The effect of ground-water humic substances on mobilization and transport of chlordane was investigated through a series of column experiments. Glass columns containing native sand spiked with [14C] chlordane were eluted, in duplicate, with humic solutions of varying dissolved organic carbon (DOC) concentration. A constant head of 4cm was maintained on the top of each column from feed carbuoys. Eluant was collected continuously and sampled once every 12hr. In all studies, chlordane concentrations were determined by Liquid Scintillation Counting.

## RESULTS AND DISCUSSION

While chlordane solubility was 32ug/L in pure water, solubility was enhanced to 468ug/L using commercial Aldrich humic acid of 50mg/L as DOC (Figure 2). Chlordane solubility was similarly enhanced in the presence of extracted humic acid (66 and 57mg/L as DOC respectively) to 474ug/L and 366ug/L. Solubility increased from 52ug/L in ground-water collected upgradient of the site (total organic carbon, TOC = 1.8mg/L) to 146ug/L in downgradient ground-water (TOC = 34mg/L). From the slope and intercept of the plots, the effect of humic substances on chlordane solubility can be described by the partition coefficient, log Kdoc (2). In the presence of commercial humic acid a log Kdoc = 5.41 was calculated, while for the ground-water a smaller log Kdoc = 4.86 was obtained, possibly because of the more hydrophilic nature of ground-water humic substances (3).

The Kd values for chlordane (18 - 220mL/g) were indicative of its low aqueous solubility and strong affinity for the solid phase. There was no correlation found between the particle size of the geologic material and the Kd. A correlation of $r^2 = 0.822$ was found between solid-phase organic carbon concentration and the Kd, demonstrating the influence of organic carbon on the partitioning of such a hydrophobic compound.

Chlordane retardation was shown to decrease with increasing concentration of humic substances in the aqueous phase. When eluted with commercial humic solutions of 2.5 and 25mg/L as DOC, chlordane breakthrough decreased from 158 to 50 pore volumes (Figure 3). Observed retardation factors were greater than those predicted (65 and 12 pore volumes respectively) using a modified form of the retardation equation (4), possibly due to the retention of humic substances that form an organic coating on particles in the column.

## CONCLUSIONS

Although batch sorption experiments demonstrated the strong binding of chlordane to the solid-phase, enhanced solubility and the increased mobilization of chlordane with increasing concentration of humic substances in the aqueous phase, show that the potential exists for chlordane to be mobilized and transported, particularly in downgradient ground-water, beneath the hazardous waste site.

## REFERENCES


Conway, R. A., 1982. Environmental Risk Analysis for Chemicals. p. 558. Van Nostrand Reinhold Co.

Chiou, C. T., R. L. Malcolm, T. I. Brinton, and D. E. Kile, 1986. Environ. Sci. Technol., 20:502-508.

Thurman, E. M., 1985. Humic Substances in Soil, Sediment and Water. pp. 87-103. G. R. Aiken, D. M. McKnight, R. L. Wershaw, and P. MacCarthy (eds.), John Wiley and Sons.

Hutchins, S. R., M. B. Tomson, P. B. Bedient, and C. H. Ward, 1985. CRC Critical Reviews in Environmental Control. pp. 355-416, Vol. 15.

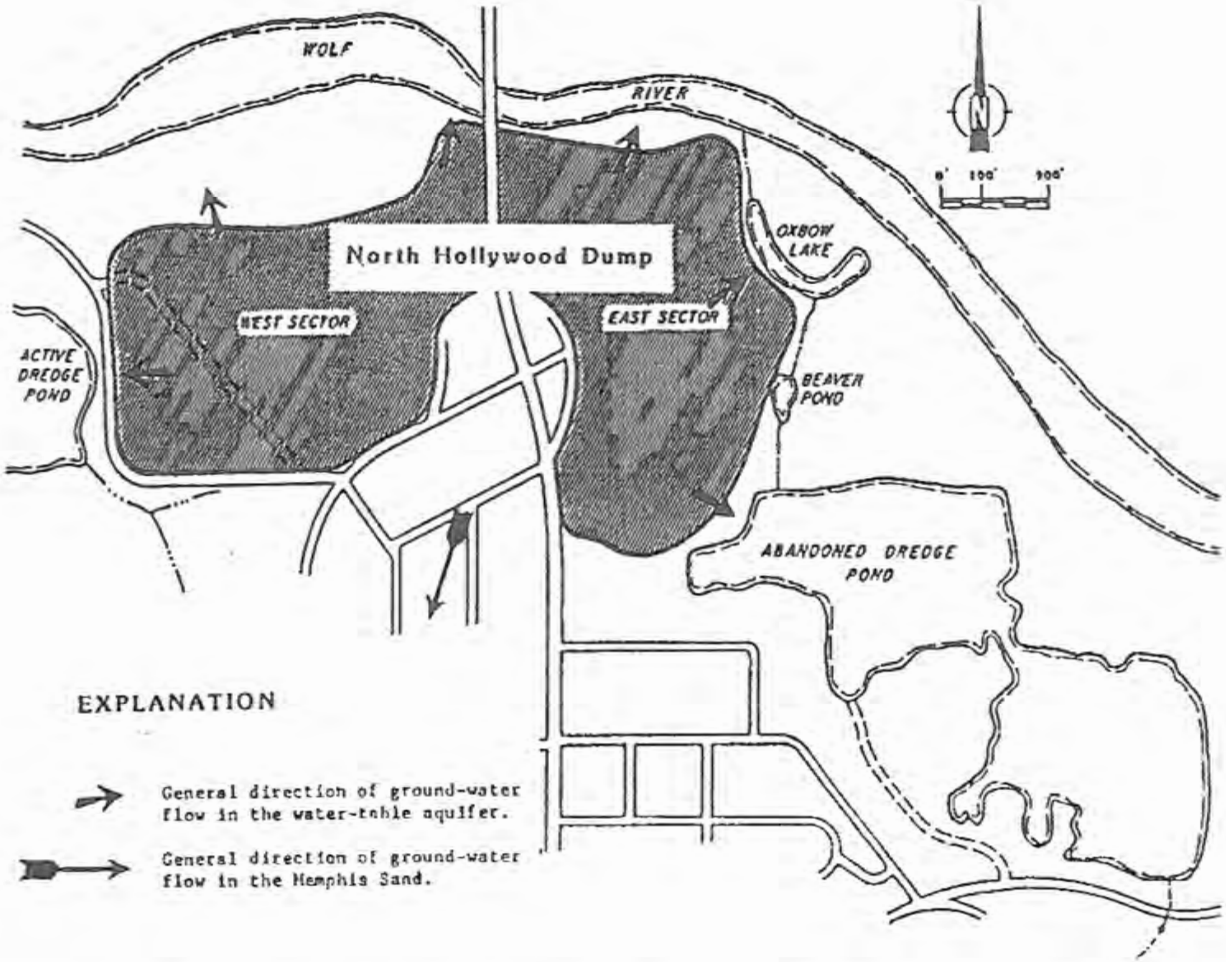


FIGURE 1. North Hollywood Dump showing general direction of groundwater flow

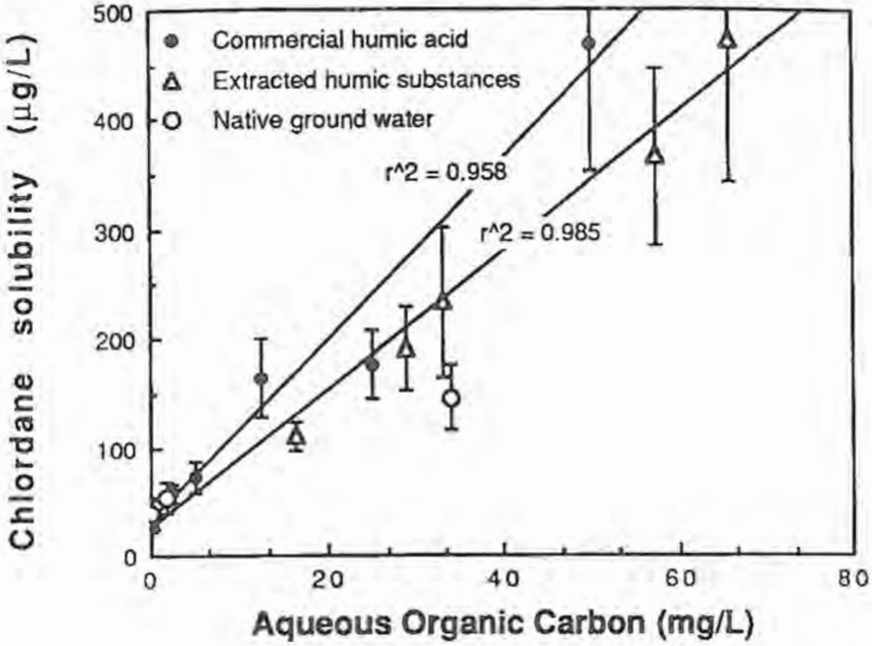


Figure 2. Correlation between chlordane solubility and aqueous organic carbon from different sources (n=3, bars represent standard deviations

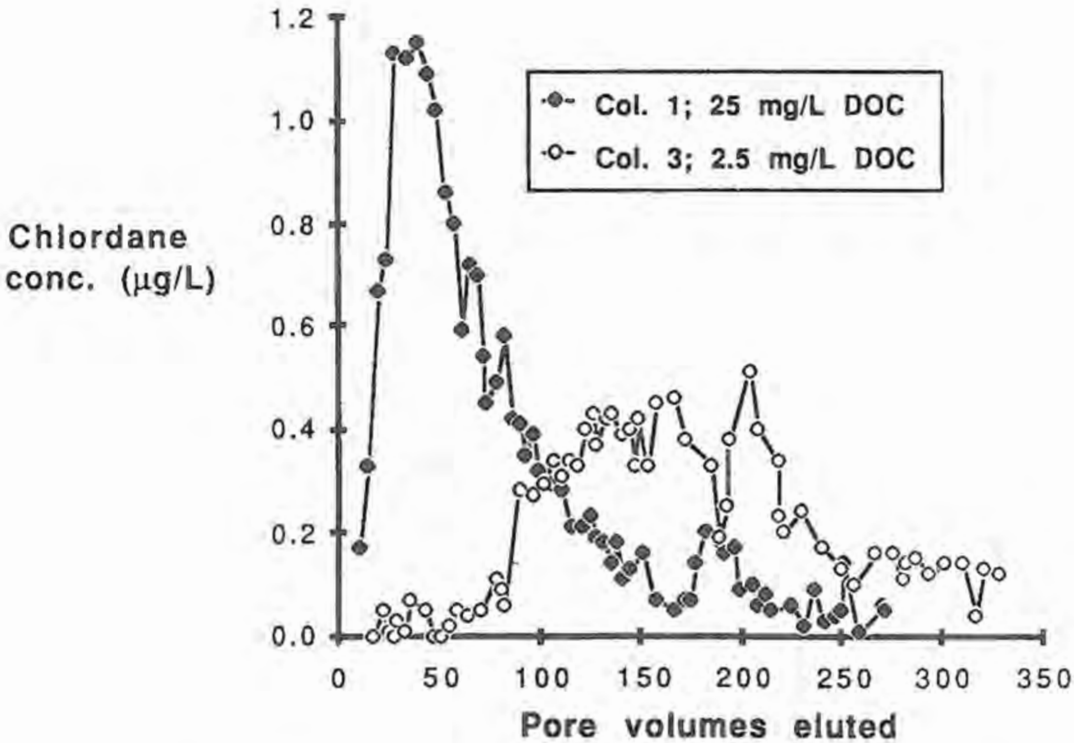


Figure 3. Breakthrough curves for chlordane through columns containing native sand eluted with humic substances of 2.5 and 25 mg/L as DOC.

# MERCURY MONITORING OF WATER AND SEDIMENT IN OAK RIDGE NATIONAL LABORATORY STREAMS DURING 1989

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## INTRODUCTION

As assessment plan was implemented in compliance with the Clean Water Act and the Oak Ridge National Laboratory's (ORNL) National Pollutant Discharge Elimination System (NPDES) Permit to identify, locate, and minimize all sources of mercury contamination in ORNL discharges to the aquatic environment. This plan was designed to identify sources of mercury from past operations and spills through a review of file records and personal interviews. A network of monitoring and sampling stations, based on knowledge of mercury deposits in receiving streams, knowledge of mercury discharges from pipes to streams, and sample collection. The plan was designed to assess the potential for mercury reaching surrounding streams and rivers by placement of sampling sites relative to potential contaminant movement from areas of deposition. This summary report describes the monitoring data for 1989, collected during the first and fourth quarters, while contrasting to the 1988 data. Based on 1988-1989 data, recommendations are proposed to eliminate those sample locations which have not provided quantitative evidence of mercury deposition. Sample locations near sources identified from the monitoring data will be retained in the sample network.

Effluents from the numerous laboratories at ORNL are treated and subsequently monitored before discharging into the receiving streams at permissible concentrations. In previous years, before stringent regulations, some contaminants reached various streams primarily as the result of accidental spills or leaks. The intent of the monitoring effort is to provide evidence that no new sources of mercury have resulted from plant operations. Receiving streams within the ORNL perimeter include White Oak Creek, Fifth Creek, First Creek, and Northwest Tributary. The more remote streams, Melton Branch, White Oak Lake, Clinch River-Melton Hill Reservoir, and Clinch River-Watts Bar Reservoir systems also receive effluents from plant operations. The locations of area streams and reservoirs are depicted in Fig. 1. Unusual incidents at ORNL are routinely reported to the Laboratory Shift Supervisor for entry into a log. Examination of the records from 1954 to 1989 indicate approximately 30,013 kg (66,168 lbs) of mercury have been reported in spills (Alexander, 1989). Three major mercury events are detailed below, the balance of losses is in gram quantities. The OREX process was similar to the METALLEX procedure but was designed to separate lithium isotopes. The lithium was amalgamated, pressed into billets, sintered, and the mercury removed by vacuum distillation leaving the lithium. This process was carried out in the basement of Building 4501 in 1954. The basement floor was

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of concrete construction with tar seams and was flooded with 10 cm (4 in.) of water. The water layer was intended to reduce mercury fumes in the building atmosphere. A steel grate above the water pool supported equipment and personnel. Throughout the process some mercury escaped from the basement at the tar seams as is confirmed by soil analyses (Oakes, 1983b). The condensed mercury was pumped to a tank truck where it was transferred to Building 3592 for cleaning and recycle. It has been estimated that an excess of 22,680 kg (50,000 lb) of mercury may have been lost during the process (Parker, 1986). Most spills were associated with pump failures where amalgam was being pumped from the basement to the upper level of Building 4501.

Mercury distilled from the OREX process was transported to Building 3592 for cleaning by resin exchange columns. Following cleaning, it was placed in containers and later removed to Y-12. A spill occurred due to operator error which involved 400 gal (20,500 kg) of mercury. Approximately 300-350 gal were recovered by vacuum sweeping. The remainder, 50-100 gal (2500 to 5000 kg), was lost to the surrounding soil, subject to transport to White Oak Creek through the Laboratory storm drain system (Dinsmore, 1986). Soil and sediment analyses confirm contamination by mercury (Oakes, 1983a).

## METHODS AND PROCEDURES

As a means of establishing baseline data for environmental concentrations of mercury, water was collected from receiving streams near the various Laboratory outfalls. Areas sampled included selected Category I, II, and III outfalls; NPDES Serial Numbered Sampling sites; and areas surrounding known mercury spills. Category I outfalls receive water from storm drains. Category II outfalls include storage area drains, spill area drains, roof and parking lot drains, and cooling tower blowdown and condensate drains. Category III outfalls receive routine process wastes and periodic laboratory wastes. These systems represent the greatest potential for mercury transport to receiving streams. The Serial Numbered Sampling sites routinely sampled for radiological contaminants are included to provide a broader survey for mercury in the Laboratory's receiving streams.

A total of 92 sites were available for sampling (water) during the study with an additional 14 sites for sediment samples. All water samples consist of three replicate, manual grab samples collected during two sampling periods (dry and wet seasons) during 1989. Samples were collected in 1-L I-Chem high-density polyethylene bottles with teflon caps. I-Chem bottles are proprietary containers, precleaned by the vendor to EPA specifications where microchemical determinations are requested. Samples were preserved immediately upon collection by acidifying with concentrated nitric acid to a pH of <2.0. Sediment samples were collected at selected stations and placed in glass containers. The glass containers were also I-Chem, EPA approved. Generally, samples were analyzed as soon as possible after collection, and no sample analysis exceeded the maximum allowable holding time of 28 days. Sediment samples were first extracted by utilizing SW845 methodology (USEPA, 1982). A modification of Method 245.1 (USEPA, 1983) was utilized for all analyses, and the results of sediment analyses were reported on a dry weight basis.

## RESULTS

In March 1989, 91 stations were sampled for mercury content in water. The lower detection limit for this series of samples was <0.05 µg/L. Only fifteen locations identified quantitative concentrations (mean $\pm$1 SE). Among outfalls along Fifth Creek, Outfall 367 had a concentration of 3.03 + 0.34 µg/L, a factor of three greater than noted in the spring of 1988. This outfall is east of Building 3039, Central Radioactive Disposal Facility. Outfalls 261 and 363 had >0.1 µg/L concentrations; all others along Fifth Creek were below the detection limit. Among 14 outfalls sampled along First Creek, none contained mercury at the detection limit (<0.05 µg/L) in contrast to a single quantitative concentration of 0.5 µg/L from Outfall 341 in the Spring of 1988 (Taylor, 1989). Among 44 outfalls sampled along White Oak Creek in the first quarter, 12 contained mercury concentrations exceeding the detection limit. This is in contrast to 4 stations among 29 sampled during the same period of 1988. The highest concentration observed was 1.83 µg/L from Outfall 304, a factor of 14 greater than observed from the same location during the same period last year. Outfall 304 is approximately 50 m east of the Process Waste Treatment Plant (Building 3544). Along Melton Branch, no outfall among 11 stations had detectable mercury. Similarly, no station sampled in the Spring of 1988 had detectable mercury concentrations. Miscellaneous stations (remote streams) sampled in 1988 identified mercury (0.17 µg/L) at one location, Lower Section of White Oak Creek. This compares to no detectable mercury among the same sites during the first quarter of 1989.

In November of 1989, 88 stations were sampled for mercury content in water. The lower detection limit for this series of samples was <0.05 µg/L. Only 21 locations identified quantitative concentrations (mean $\pm$ 1 SE). Among those outfalls along Fifth Creek, Outfalls 261 and 363 exceeded the analytical detection limit with concentrations of 0.12 $\pm$ 0.08 and 0.10 $\pm$ 0.003 µg/L, respectively. During the first quarter, these outfalls did not have detectable mercury. Outfall 261 is east of Building 3500 and Outfall 363 is west of Building 4501. Among 13 locations along First Creek, only Outfall 341 exceeded analytical detection limits (0.37 $\pm$ 0.01 µg/L). During the first quarter, no outfall along First Creek had detectable mercury. Forty-five locations were sampled along White Oak Creek during the fourth quarter, with 17 showing detectable concentrations of mercury. This compares to 12 locations among 44 during the first quarter sampling. The maximum concentration was observed from Outfall 304, while the minimum concentration (0.04 $\pm$ 0.01 µg/L) was from Outfall 210. Outfall 304 had the maximum concentration of 1.83 $\pm$ 0.73 µg/L during the first quarter. Outfall 304 is located south of the Process Waste Treatment Plant (Building 3544) and the Process Waste Water Treatment Plant (Building 3518). Nine locations were sampled along Melton Branch during the fourth quarter. Mercury was not detectable at any location. This was also the situation during the first quarter. Among miscellaneous locations (remote steams, background locations, etc.) the lower portion of White Oak Creek reported a quantitative concentration (0.04 $\pm$ 0.02 µg/L). While this concentration is less than the analytical detection limit, the standard error (50 %) indicates that one or more replications had a detectable concentration.

Twelve stations were sampled for sediments along the Laboratory's streams during the first quarter 1989. A 1988 report identified mercury in sediments ranging from 0.13 ± 0.02 µg/g (White Oak Creek headquarters) to a maximum of 4874 ± 2556 µg/g near Outfall 261, east of Building 3500. During the first quarter 1989 sampling effort, the same location continued to exhibit elevated mercury concentrations. Outfall 261 had a mean of 555.67 ± 310.38 µg/g, a factor of eight less than the concentration reported in 1988. The decrease is related to sampling variability and does not reflect a reduction of the source. The spatial mercury contamination of ORNL streams is presented in Fig. 2(a). The data are not intended to infer a dilution with distance from the ORNL complex. For concentration data to illustrate a dilution phenomenon, all sediments must be sieved with stones and organic materials removed.

During the Fourth Quarter the 12 stations sampled during the first quarter were sampled with the addition of samples from Melton Branch headwaters (an additional background location) and White Oak Creek downstream from the confluence with Melton Branch. The concentration data are presented in Fig. 2(b). Considering the variability between replications and within locations, the data were similar to the results of 1988 and first quarter 1989. The minimum concentration (0.003 µg/g) observed was at White Oak Creek downstream of the confluence with Melton Branch, while the maximum (7427 µg/g) was near Outfall 261. The background concentration from White Oak Creek headwaters (0.02 ± 0.003 µg/g) is less than the average concentration (0.17 µg/g) reported for the eastern (0.17 µg/g) reported for the eastern conterminous United States (Schacklette et al, 1971). The major difference from the spring quarter was the decreased concentration (5.67 µg/g) noted downstream from the containment box below Outfall 362. The spring quarter concentration was 155.81 µg/g. During the fourth quarter, it was noted that the box did not have a significant accumulation of sediment, indicating the probability of washout by heavy stream flow following storm events. The mean concentration near Outfall 261 was much higher during the fourth quarter (7427 µg/g), where the between-replication error was 95% in contrast to a 55% between-replication error associated with the first quarter mean of 555.67 µg/g. The sediment data are considered important indicators in the selection of locations for continued water sampling sites.

The monitoring report for 1988 (Taylor, 1989) identified a maximum sediment concentration of 4874 µg/g (top 5 cm) below Outfall 261. The discharge plume to Fifth Creek is 20 cm wide and 0.5 m long. The unusually high concentration suggested that a more detailed sample procedure include soils within depth profiles along the distance gradient to the creek. Samples were collected from 0-5, 5-10, 10-15, and 15-20 cm depths at 0, 0.25, and 0.5 m from Outfall 261. The concentrations nearest Fifth Creek indicate a dilution with distance, and the concentration similarity within the depth profile (0 to 20 cm) suggests a single release (spill) event (Fig. 3). The youngest deposition is nearest the surface, such that the greater concentrations noted between 5 and 15 cm deep indicate no new or recent deposition. The greatest concentration noted (21,500 µg/g) occurred within the top 5 cm at a distance of 0.25 m from the outfall. The soil bulk density (1.4 g/cm$^3$) was determined gravimetrically and incorporated into calculations of the total soil mass present in each 5-cm-thick layer of the outfall to creek plume. Mercury concentrations from 0, 0.25, and 0.5 m (n = 3) along the plume were averaged to calculate the total mercury present. It is estimated that 52 g are present in the 0-5 cm layer, 3 g in the 5-10 layer, 5 g in the 10-15 cm layer, and 2 g in the 15-20 cm layer for a total of 62 g.

## CONCLUSIONS AND RECOMMENDATIONS

The water chemistry data from a 1987 scoping survey, the 1988 annual mercury monitoring data, and the 1989 annual mercury monitoring data have clearly identified potential sources of mercury to ORNL streams. Sediment analyses identify pools of residual mercury as potential release sources during heavy runoff following storm events. A criterion for reducing the water sampling efforts during 1990 is proposed, based on the absence of any evidence of mercury discharges relative to the State of Tennessee, Criteria for Water Conditions, Domestic Water Supply, Tennessee Rule 1200-4-3-.03, 1983. That guideline is 0.2 µg/L (0.2 ppb). As a conservative measure, it is proposed that any outfall source or stream location having mercury concentrations equivalent to 50% (0.1 µg/L) of the state rule during two or more sampling periods be continued as a sampling location. All others would be deleted as a cost-effective measure to attain the goals of the annual mercury monitoring plan. With these recommendations, 18 locations are recommended for annual sampling. On the other hand, mercury in sediments has the potential to be methylated and become bioavailable. Concentrations greater than 1 µg/g (A Conservative Estimate of Mercury in Surficial Materials in the Conterminous United States, Schacklette et al, 1971) have a potential to influence stream concentrations. Therefore, nine sediment locations are proposed for continued sampling. It is recommended that a remedial action project be initiated to remove the soil/sediment plume from Outfall 261 to Fifth Creek. This would eliminate the area of highest mercury contamination.

It is a general conclusion that mercury contamination of ORNL waters is not yet an environmental concern. This conclusion is based on a good record of spill or accident events and three years of detailed water and sediment chemistry data.

## REFERENCES

Personal file, W. A. Alexander, 1989, History of Chemical Spills at ORNL.

Letter, T. W. Oakes to J. F. Wing, Department of Energy/Oak Ridge Operations, "Mercury in Soil and Sediment at ORNL," (1983a).

Personal communication, George W. Parker to F. G. Taylor, Jr., October 1986, Oak Ridge National Laboratory.

Personal Communication, George B. Dinsmore to F. G. Taylor, Jr., October 1986, Oak Ridge National Laboratory.

U.S. Environmental Protection Agency (EPA), 1982. Test Methods for Evaluating Solid Wastes, Physical/Chemical Methods. EPA-SW-846. Office of Solid Waste and Emergency Response, Washington, D.C.

U.S. Environmental Protection Agency (EPA), 1983. Methods for Chemical Analysis of Water and Wastes. EPA-600/4-79/020. Environmental Monitoring and Support Laboratory, Office of Research and Development, Ohio.

Taylor, Fred G. Jr., 1989. Mercury Assessment for Water and Sediment in Oak Ridge National Laboratory Streams. ORNL/M-713, Martin Marietta Energy Systems, Inc., Oak Ridge National Laboratory.

Shacklette, H. T. et al., 1971. Mercury Content of Surficial Materials in the Conterminous United States. U.S. Geological Survey Circular, 644:1-5.

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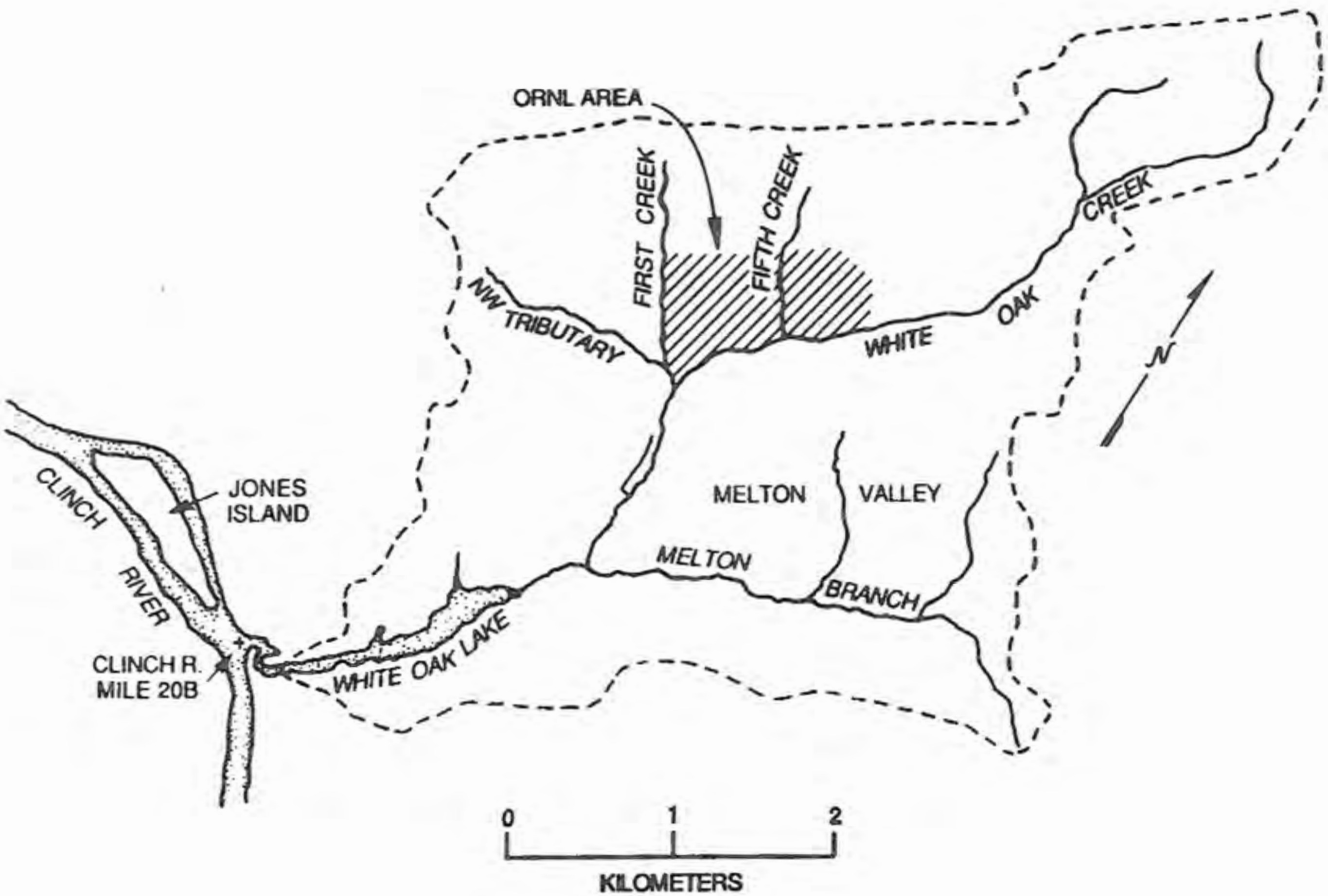


Figure 1. Locations of ORNL streams, rivers, and impoundments.

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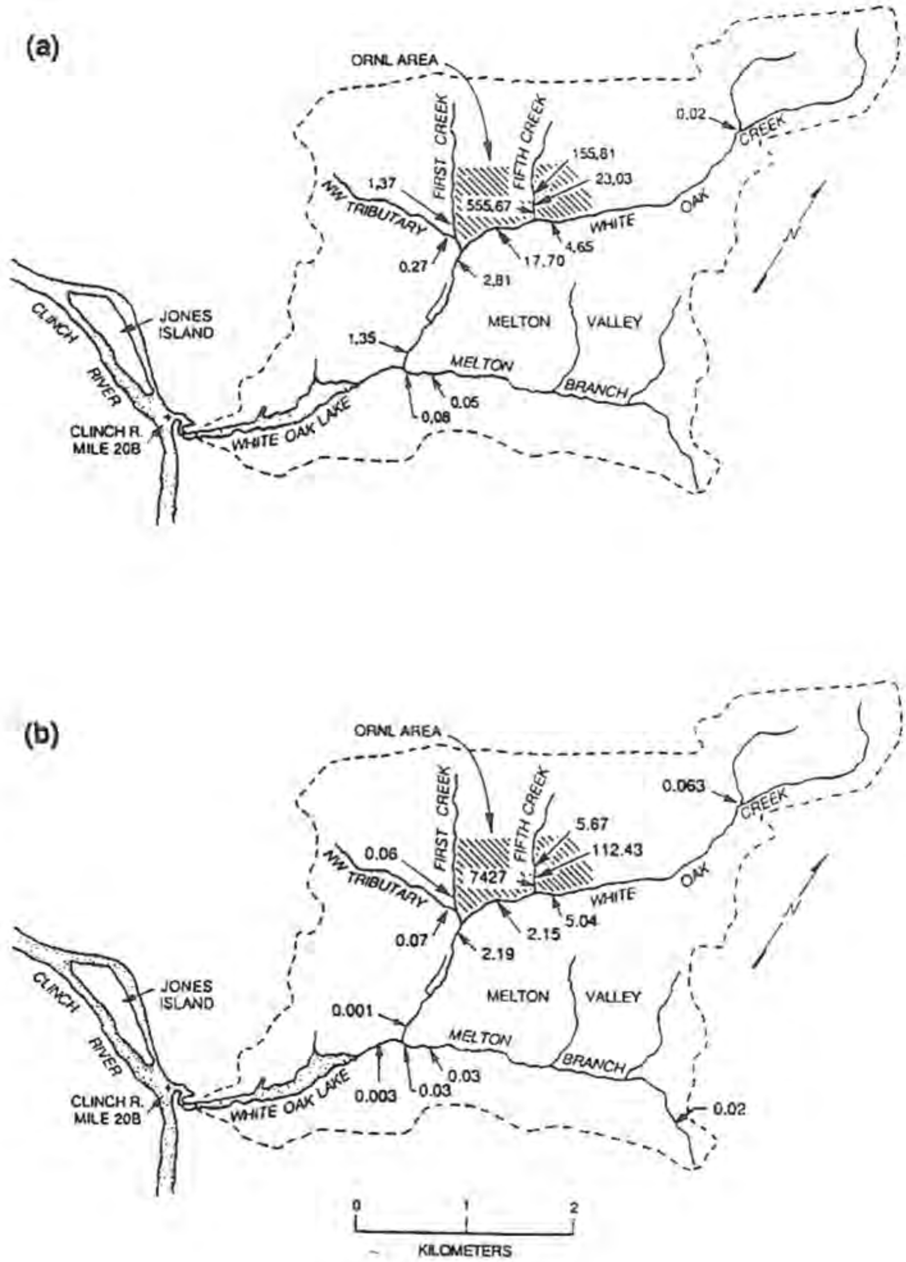


Figure 2. Locations in ORNL stream with excess mercury in sediments: (a) First Quarter, (b) Fourth Quarter. Concentrations are ug/g dry weight.

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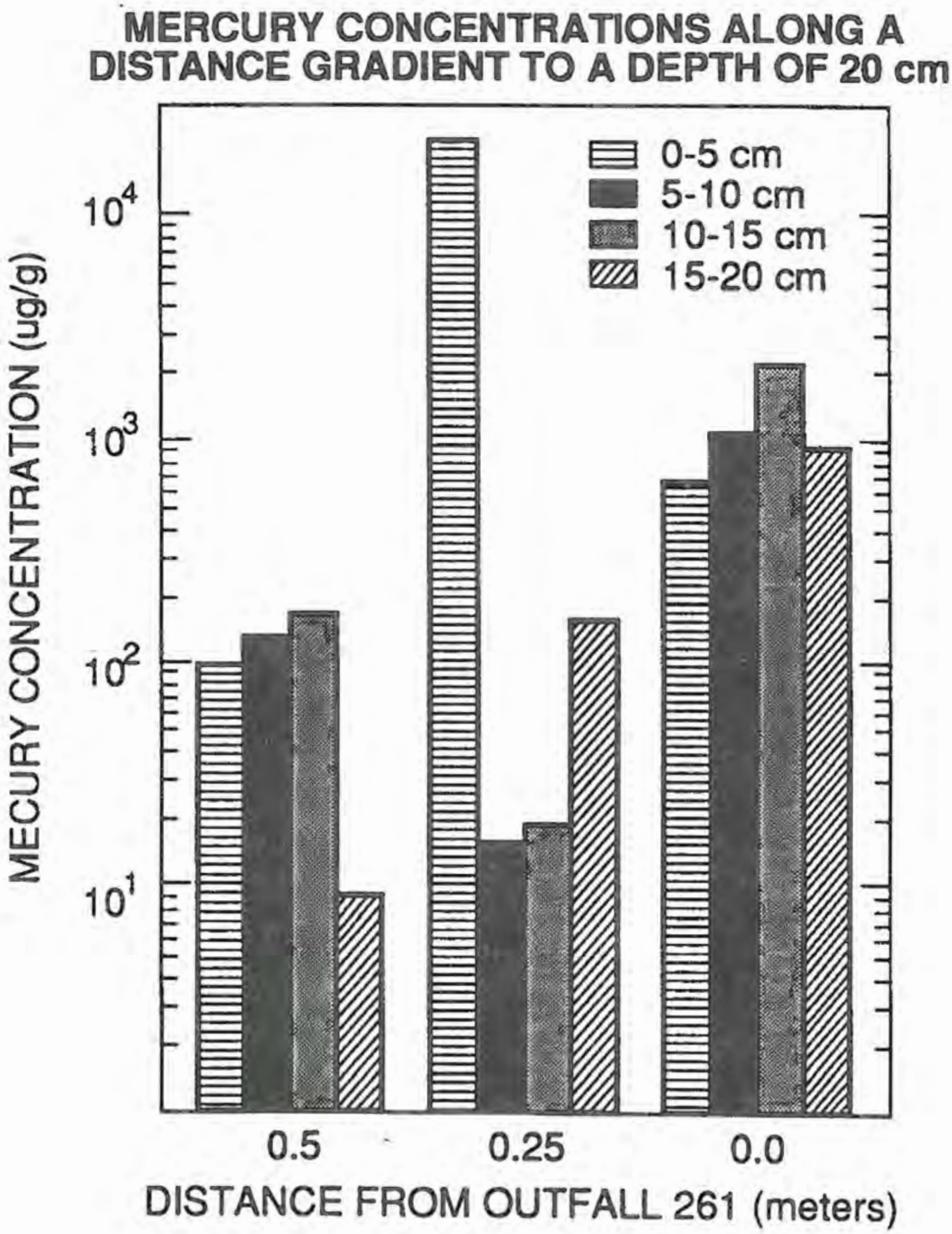


Figure 3. Mercury concentrations in sediments along a distance gradient to a depth of 20 cm downstream of Outfall 261.

# NASHVILLE DISTRICT COE INVOLVEMENT IN THE DEFENSE ENVIRONMENTAL RESTORATION PROGRAM

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The Defense Environmental Restoration Program (DERP) is the Department of Defense (DoD) program for remediating environmental problems which have occurred as a result of DoD activities. This program was authorized in Section 120 of the Superfund Amendments and Reauthorization Act (SARA) of 1986. DERP is divided into two categories. The Installation Restoration Program (IRP) covers military installations which are still active or under DoD ownership. The Formerly Used Defense Sites (FUDS) covers properties which were formerly owned and utilized by DoD but have since been excessed to other public or private owners.

Congress funds DERP on an annual basis as part of DoD's budget. The IRP and FUDS are funded separately as subsets of DERP. Since Superfund money may not be used at federal facilities, this funding is the only source of money available to clean up past DoD contamination. These funds can also be used at sites where DoD has been named as a potentially responsible party (PRP) by the Environmental Protection Agency.

The remediation process under DERP parallels the process which is followed under the Environmental Protection Agency's Superfund Program. A preliminary assessment is initially done to determine if the potential for contamination exists. This assessment usually consists of a site visit, a historical records search,real estate records search, and interviews with current or former personnel. An Inventory Project Report is generated from this information. The Inventory Project Report can be positive or negative. The negative projects reports require no further action and are filed. The positive reports are considered for either a Site Investigation (SI) or Remedial Investigation/ Feasibility Study (RI/FS). If significant environmental contamination is found, then the Remedial Design (RD) and Remedial Action (RA) are conducted for the site.

The Corps of Engineers has been given the responsibility for conducting the Defense Environmental Restoration Program for Formerly Used Sites (DERP-FUDS) for the Department of Army. There are currently 7,083 sites nationwide which are being considered under DERP-FUDS. In order for a site to be eligible for clean-up under DERP-FUDS, it must meet the following criteria:

1. The site was formerly owned/used by DoD.
2. A hazard exists at the site which was a result of DoD presence at the site.

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3. The site meets requirements of current DoD policy.
4. The current owner agrees to allow access to the site.

As of April 1990, the preliminary assessment of 1,934 sites was complete. Of these sites, 325 were considered eligible for remediation under DERP-FUDS. More sites will be added to this list as the remaining sites on inventory are completed. The Nashville District is responsible for the DERP-FUDS within the geographical boundaries of the Ohio River Division. This area is shown in Figure 1.

A wide variety of environmental problems are encountered at former DoD facilities. Common contaminants include solvents, metals, and petroleum hydrocarbons. Another common contaminant which is almost exclusive to DoD facilities is explosives. DERP-FUDS projects are divided into the following five categories based on the type of potential contamination present: Hazardous and Toxic Waste, Hazardous and Toxic Waste - Potentially Repsonsible Party, Hazardous and Toxic Waste - Containerized, Ordnance, and Debris.

The Nashville District currently has several containerized Hazardous and Toxic Waste (HTW) projects. Underground Storage Tanks are included under this category. These projects are coordinated very closely with the state regulatory agencies. Projects under this category which have been completed in Tennessee include the Former William Northern Airfield in Tullahoma and the Former Bradyville Gap Filler Radar Annex in Bradyville. Both of these sites had steel underground storage tanks which were installed in the 1940's. These tanks were in surprisingly good condition when they were removed in 1988. Very little contamination was found in the soil. Another underground storage tank project is scheduled to begin this year at the Former Sewart Air Force Base in Smyrna. The Nashville District currently has several other underground storage tank remediation projects in Ohio. PCB transformers are also considered containerized HTW. The Nashville District recently completed a project of this type in Indiana.

The Nashville District is also conducting several site investigations under DERP-FUDS. These sites include former ordnance plants and former Nike missile sites. The site investigation at the Former Chickasaw Ordnance Works in Millington, TN was completed this past spring. DoD acquired the 5,500-acre reservation in the early 1940s. An ammunition plant was operated there during World War II. After the war, the land was excessed and is currently owned by private citizens and Shelby County, TN. A geophysical survey was done at this site and soil, ground water and surface water samples were collected to determine if any contamination existed at the site. Existing records indicated that residual explosives contamination was likely, however, no explosive contaminants were found during the site investigation. Moderate levels of metals contamination were observed in one of the disposal areas used by DoD, and this area is being recommended for further study. With the exception of this disposal area, no contamination resulting from past DoD activities was observed. The Nashville District has found explosives contamination at former ordnance plants in Ohio and Indiana.

The Defense Environmental Restoration Program - Installation Restoration Program (DERP-IRP)is centrally managed by the U.S. Army Toxic and Hazardous Materials

Agency (USATHAMA). USATHAMA became part of the Corps of Engineers in 1989. The design phase of these projects is currently being brokered to the Hazardousand Toxic Waste Field Operating Agencies (FOA) within the Corps. DERP-IRP focuses on cleanup of contamination associated with past activities at active military installations. Eligible projects under DERP-IRP include remedial actions to protect or restore areas damaged by contamination from past HTW disposal activities; research, development and technology demonstrations necessary to conduct cleanups; operation and maintenance costs for the first ten years of remedial and monitoring systems; remediation of underground storage tanks not used since January 1984; and response actions under CERCLA/SARA including sites where DoD is a PRP. Projects are not eligible for DERP-IRP if they are covered under the Resource Conservation and Recovery Act (RCRA).

As the designated FOA for the Ohio River Division, the Nashville District is anticipating an increased workload in design for DERP-IRP projects. Current DERP-IRP projects include an interim remedial action at Ft. Campbell, KY. The groundwater in the vicinity of the airfield has been contaminated with JP-4 jet fuel. USATHAMA conducted the remedial investigation and the Nashville District is currently designing a system to prevent the release of petroleum hydrocarbons into surface waters or groundwater aquifers as well as remove free phase petroleum hydrocarbons from existing montoring wells around the airfield.

The workload for the Nashville District under DERP is constantly increasing. Future projects will involve more remedial investigation and remedial design activities.

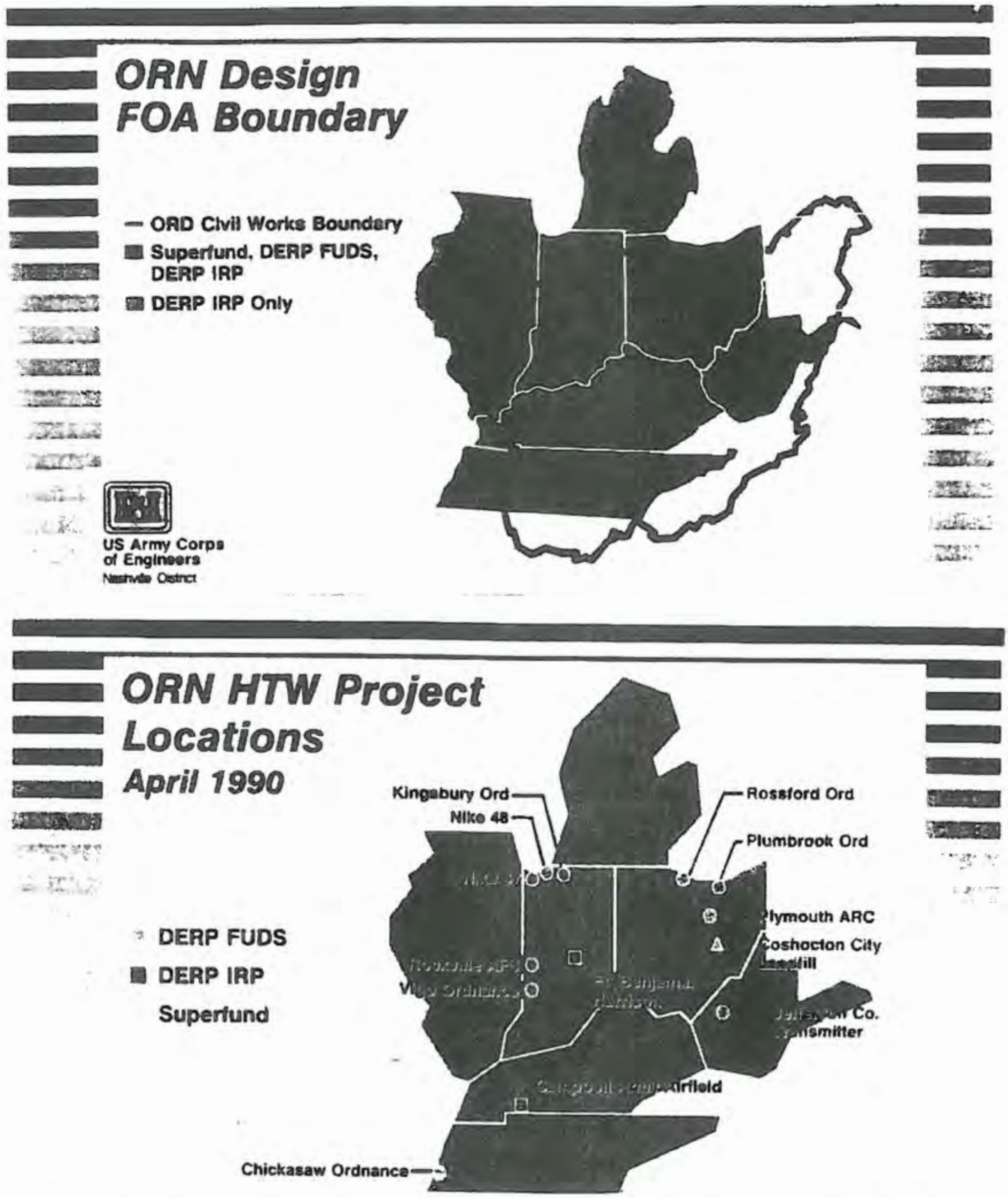


Figure 1. Ohio River Division Boundaries and Locations of Nashville District DERP Projects.

# THE OPINIONS OF TENNESSEE LEADERS ON WATER QUALITY ISSUES

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As part of our program planning process, the Tennessee Agricultural Extension Service (AES) conducted a mail survey of community leaders throughout the state during the fall of 1988. The objectives were to gather opinions about water quality issues and to identify leading local concerns to help county AES staff develop stronger programs.

The purpose of the survey was to provide county level information for local planning and program development. However, the results also provide some insight into opinion from across the state. This paper presents highlights of the state-level summary. A more detailed state report is available from the author.

## PROCEDURE

Fourteen water quality issues were identified through a review of current literature and discussions with knowledgable professionals. A survey form asking three types of questions was developed. Respondents were asked:

1. To rate the severity of the fourteen issues in their home county on a one to five scale (critical, major, minor, no problem, uncertain).

2. To prioritize the importance of work on each issue in their county on a one to four scale (high, average, low, uncertain).

3. To list the three water quality issues AES should target for action in their county.

Each county AES office mailed questionnaires to leaders the county staff identified from 13 categories. Twelve groups were identified in the instructions to county staff and are listed in Table 1. Other locally important groups such as chambers of commerce, farm organizations and special interest groups typically exist. The option of mailing questionnaires to leaders in other organizations was given; these are identified as "other" in Table 1. Envelopes were provided to return completed forms to the county office.

Past research has shown that the opinions of leaders identified through a local nomination process are generally representative of community opinion. The survey may then offer some sense of the opinions of the average Tennessean as well as the individuals who responded to the survey.

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## RESULTS

A total of 3,257 questionnaires were returned. The number and percentage of the total returned by each group are reported in Table 1.

Table 2 reports (a) the number and percent of respondents rating each issue as a critical or major concern; (b) the number and percent of respondents rating work on each item as a high priority; and (c) the number and percent of respondents listing an issue as a priority for AES programming.

## DISCUSSION

As shown in Table 2, a few items were consistently identified as important issues throughout the survey. These include lack of awareness of the nature and importance of water resources; understanding the effects of current practices on future supplies of safe water; lack of knowledge and use of water conservation methods; improper disposal of potential contaminants; soil erosion on farms; and inadequate supplies of safe water. This kind of consistency might be expected. Issues rated as locally important would, in most cases, also be leading priorities for action.

One possible criticism of the survey is that local opinion may be uninformed and not based on objective fact. The point is valid. However, citizen perception is a factor in dealing with issues of public concern. Awareness is a prerequisite to marshalling support for addressing an issue. Opinion can also help shape actions taken. Public perception is a reality which merits consideration in making and carrying out public policy

However, while this approach may provide insight, many other sources of information exist. They should also be considered in creating and implementing programs. In fact, lack of awareness of significant issues may in itself suggest the need for increased public education efforts.

Educational programs are one means of addressing priority issues identified in this survey. The AES is currently developing and delivering such programs in cooperation with several other agencies and organizations.

## REFERENCE

Smith, G. F. and T. H. Klindt, 1976. The People in Tennessee's Title V Counties: A Summary Report on Characteristics and Attitudes. Bulletin 558, Agricultural Experiment Station, University of Tennessee, Knoxville, Tennessee.

Table 1

1988 Water Quality Survey Respondents

| Group | Number | Percent |
| --- | --- | --- |
| Bankers | 178 | 5.5 |
| County agricultural Extension committee | 249 | 7.6 |
| County government Officials | 248 | 7.6 |
| Community clubs | 80 | 2.5 |
| Homemaker clubs | 478 | 14.7 |
| Merchants | 170 | 5.2 |
| Ministers | 159 | 4.9 |
| Newspaper editors | 70 | 2.1 |
| School principals | 379 | 11.6 |
| Senior 4-H clubs | 64 | 2.0 |
| Service clubs | 207 | 6.4 |
| Rural development Committees | 444 | 13.6 |
| Other | 319 | 9.8 |
| Not identified | 212 | 6.5 |

Table 2

1988 Water Quality Survey Responses

| Issues | Number (percent) of Responses | | |
| --- | --- | --- | --- |
| | Critical or Major Local Issue | Merits High Priority | Priority for AES |
| Soil erosion on farms | 1489(8.1) | 1125(8.7) | 336(7.7) |
| Soil erosion from construction sites | 1093(6.0) | 674(5.2) | 116(2.7) |
| Soil erosion from road banks, parks and public property | 938(5.1) | 581(4.5) | 72(1.6) |
| Misuse of Agricultural chemicals | 1108(6.0) | 901(7.0) | 374(8.6) |
| Misuse of household chemicals | 681(3.7) | 497(3.9) | 36(0.8) |
| Misuse of chemicals by local businesses | 1046(5.7) | 807(6.3) | 223(5.1) |
| Improper disposal of leftover chemicals, empty containers, used motor oil, etc. | 1164(6.3) | 1266(9.8) | 463(10.6) |
| Farm livestock wastes contaminating water | 585(3.2) | 389(3.0) | 141(3.2) |
| Farmers lack of knowledge and use of best management practices | 1131(6.2) | 697(5.4) | 143(3.3) |
| Lack of citizen awareness of the nature and importance of water resources | 2279(12.4) | 1457(11.3) | 541(12.4) |
| Lack of knowledge and use of water conservation methods | 1941(10.6) | 1172(9.1) | 559(12.8) |
| Inadequate supplies of safe water | 1279(7.0) | 935(7.2) | 528(12.1) |
| Lack of information about water testing | 1427(7.8) | 830(6.4) | 326(7.5) |
| Understanding effects of current practices on future supplies of safe water | 2191(11.9) | 1567(12.2) | 503(11.5) |

# OPPORTUNITIES AND EFFORTS FOR MULTIOBJECTIVE MANAGEMENT OF RIVER CORRIDORS IN TENNESSEE
## (A Floodplain Management Perspective)

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## INTRODUCTION

Tennessee's 20,000 miles of rivers and streams and the lands adjacent to them contain some of the State's most important assets. Comprising about 9½ percent of the total land area, they provide a variety of valuable resources and contain substantial capital investments in structures for those who live, work, or engage in business as well as infrastructure for transportation and commerce. It is estimated that up to 20 percent of land in the State's urban areas is subject to flooding and that about half of this flood-prone land is development.

Management of the State's river and stream corridors involve a variety of disciplines, governments, programs, and activities. The most prevalent in terms of numbers and resources involve those for reducing the economic losses resulting from flood events that visit the State from time to time often with devastating results. Rarely is the entire State spared this annual calamity.

Over 225 of Tennessee's 340 flood-prone localities are currently participating in the National Flood Insurance Program (NFIP) established by the Congress in 1968. Individuals in participating counties and communities may purchase flood insurance, intended as a substitute for postflood disaster assistance and other forms of Federal aid for flood recovery. In return for the availability of flood insurance, localities are required to enact and enforce land-use and control measures for future development in flood-prone areas.

The State's floodplains in their natural or relatively undisturbed condition possess a variety of environmental values, often of great but unrecognized or under-recognized value. Most of the state's riverine wetlands and environmentally sensitive and critical areas are located within its floodplains, significantly contributing to the array of natural and beneficial values. These values and benefits include fish and wildlife habitat, aquatic productivity, water quality maintenance, natural flood storage, erosion control, groundwater supply and balance, natural products, cultural resources, opportunities for scientific study and outdoor education, and recreational opportunities and aesthetics. One study has grouped these values into three broad categories: water resources values, living resources values, and cultural resources values (Exhibit 1).

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Exhibit 1

## Floodplain Natural and Cultural Values

### Water Resources Values

Natural Flood and Erosion Control
- Reduce flood velocities
- Reduce flood peaks
- Reduce wind and wave impacts
- Stabilize soils

Water Quality Maintenance
- Reduce sediment loads
- Filter nutrients and impurities
- Process organic and chemical wastes
- Moderate temperature of water
- Reduce sediment loads

Maintain Groundwater Supply and Balance
- Promote infiltration and aquifer recharge
- Reduce frequency and duration of low flows

### Living Resources Values

Support Flora
- Maintain high biological productivity of floodplain and wetland vegetation
- Maintain productivity of natural forests
- Maintain natural crops

Provide Fish and Wildlife Habitat
- Maintain breeding and feeding grounds
- Create and enhance waterfowl habitat
- Protect habitat for rare and endangered species

### Cultural Resources Values

Maintain Harvest of Natural Products
- Create and enhance agricultural lands
- Provide areas for cultivation of fish and shellfish
- Protect silvaculture

Provide Recreation Opportunities
- Provide areas for active and consumptive uses
- Provide areas for passive activities
- Provide open-space values
- Provide aesthetic values

Provide Scientific Study and Outdoor Education Areas
- Provide opportunities for ecological studies
- Provide historical and archaeological sites

Source: U.S. Water Resources Council. A Unified National Program for Floodplain Management. Washington, D.C. 1979.

Source: A Status Report on the Nation's Floodplain Management Activity, An Interim Report, Prepared for the Interagency Task Force on Floodplain Management, L. R. Johnston Associates, April 1989.

## EXPANDING MANAGEMENT OBJECTIVES

Most floodplain management measures carried out at the local level are in response to the requirements of the NFIP for flood loss reduction and are not designed or intended to achieve other objectives. Strategies for flood loss reduction may be grouped into those designed to modify flooding, those that modify the impact of flooding, and those that reduce the susceptibility of flood damage ( Exhibit 2). Although widely viewed as floodplain management, most measures implemented at the local level involve only floodplain regula- tions and eligibility for individuals to purchase flood insurance. The degree of commitment to end effectiveness of local regulatory measures varies considerably.

Among many floodplain managers, there is a growing awareness of the importance along with an increased interest in maintaining the natural and beneficial resources and resource values within the State's river and stream corridors. Existing land-use and control measures for flood loss reduction and public safety now in place in over 225 communities provide a limited degree of resource protection; but, more importantly, they provide a framework for multiobjective river corridor management.

## INTEREST IN RIVER CORRIDOR MANAGEMENT

There are several causes or reasons for this growing (but not yet widespread) interest.

Over the past few decades better procedures have been developed to identify and document the natural and beneficial resources and values contained within river corridors including impact analysis, i.e., not only the extent of benefits these resources provide but also how they can be impacted, impaired, or even lost.

The passage of the National Environmental Protection Act of 1968, and Executive Orders 11988 - Floodplain Management and 11990 - Protection of Wetlands have helped to give formal recognition to the environmental aspects of floodplain management and the broader river corridor and to establish a conceptual and management framework.

There are increasing (although yet limited) levels of interdisciplinary and intergovernmental cooperation. Most riverine wetlands are located within floodplains, and wetlands managers, floodplain managers, and other natural resources managers are discovering that they are involved in the same areas and have many common interests and needs. But much remains to be done. There is still a significant polarization of positions, missions, and attitudes.

Some managers have already recognized, and others are discovering that single-objective management approaches are not working well or as effectively as they could or should be working. Most floodplain management objectives and practices use existing watershed and floodplain conditions to determine areas of involvement; generally deal with controlling future development; do little for existing problems; look at only a small portion of the floodplain and river corridor, i.e., the area that would be inundated by the 1-percent-annual-chance flood; and are typically single purpose, i.e., flood loss reduction.

Exhibit 2

## Strategies and Tools for Flood Loss Reduction

### Strategy A. Modify Susceptibility to Flood Damage and Disruption

1. Floodplain regulations
   a) State regulations for flood hazard areas
   b) Local regulations for flood hazard areas
      1) Zoning
      2) Subdivision regulations
      3) Building codes
      4) Housing codes
      5) Sanitary and well codes
      6) Other regulatory tools
2. Development and redevelopment policies
   a) Design and location of services and utilities
   b) Landrights, acquisition, and open-space use
   c) Redevelopment
   d) Permanent evacuation
3. Disaster preparedness
4. Disaster assistance
5. Floodproofing
6. Flood forecasting and warning systems and emergency plans

### Strategy B. Modify Flooding

1. Dams and reservoirs
2. Dikes, levees, and floodwalls
3. Channel alterations
4. High-flow diversions
5. Land-treatment measures
6. Onsite detention measures

### Strategy C. Modify the Impact of Flooding on Individuals and the Community

1. Information and education
2. Flood insurance
3. Tax adjustments
4. Flood emergency measures
5. Post-flood recovery

Source: Interagency Task Force on Floodplain Management. A Unified National Program for Floodplain Management. Federal Emergency Management Agency. Washington, DC 1986.

Source: A Status Report on the Nation's Floodplain Management Activity, An Interim Report, Prepared for the Interagency Task Force on Floodplain Management, L. R. Johnston Associates, April 1989.

Community and citizen interest in floodplain management for flood loss reduction is often lukewarm at best. Most programs exist because of a requirement to participate in the NFIP. Because of the perceived unlikelihood of occurrence of the regulatory ("100-year") flood and the prevailing view that after a flood government at a higher level will bail them out by restoring everything to preflood conditions or more likely even better, bigger, and stronger, any support is often tacit. Putting together multipurpose manage- ment plans and programs to meet a number of community needs helps broaden the political and public support needed for success. This includes being able to put together funding packages where resources and support are not adequate for single-objective approaches.

## SOME EXISTING EFFORTS

Several Tennessee communities are carrying out multiobjective planning and implementation of river corridor management plans for park, parkway, wildlife, conservation or other environmental or social uses. The most notable of these are Chattanooga and Kingsport.

At the State level, there is interest in a Statewide assessment to identify the most significant rivers or river reaches for various resource categories. Such an assessment would provide a valuable data base for future decision- making. The State has already designated some scenic rivers reaches for special management.

The Governor's Wetlands Task Force (formed about a year ago and comprised of representatives of State and Federal water resources agencies, private conservation and recreation groups, and the Tennessee Farm Bureau) is to attempt to develop a consensus wetlands acquisition and conservation plan for Tennessee. The work of this Task Force is important to river corridor management in that, as stated earlier, most of the State's wetlands are within its river corridors.

At the Federal level, EPA has taken the lead role in sponsoring several symposiums on the subject of multiobjective management of river corridors over the past 2 years. The purpose of these symposiums was to heighten awareness and interest in management opportunities and to build broader support of all levels.

The President's Commission on Americans Outdoors, created by President Reagan in 1984 and chaired by Governor Lamar Alexander of Tennessee, took due note of the importance of these stream corridors for outdoor recreation. The Commission's report (released in 1987) called for wide-ranging programs to be implemented at the local level that would clean up and protect rivers and wetlands, preserve floodplains, and educate the public about these valuable recreation resources. The report also envisioned a "living network of green- ways" that could be established on streams and other corridors which would link community to community and eventually span the Nation.

Support is growing at the congressional level. In the early part of 1989, Congressmen McDade (PA) and Udall (AZ) sponsored a series of fact-finding workshops across the Nation to gather information and reach a consensus to more clearly define the goals, attitudes, and alternatives available for coordinating the multiple uses of river corridors, including finding ways for achieving balances between conservation and economic uses. Proposed legislation to facilitate better multiobjective river corridor planning and management was introduced in this session of the Congress. Hearings will be held at a later date.

## SOME SOURCES OF ASSISTANCE

Several years ago EPA created an Office of Wetlands Protection within the agency in response to the continuing losses and degradation of wetlands to look at nonregulatory initiatives and ways of protecting wetlands. The Atlanta regional office of EPA, through this program, can provide technical assistance on how to maintain and/or restore the natural and beneficial values of flood-plains and other river corridor lands. This assistance includes identification of these values, how they will be impacted by proposed development or use, and prevention or mitigation actions and techniques.

In addition the U.S. National Park Service (NPS), within the U.S. Department of the Interior, has assisted communities in planning to protect greenway values. One community that has been assisted is Chattanooga. This has been accomplished primarily through the NPS State and Local Rivers Conservation Assistance Program. Assistance usually consists of Statewide river assessments and river corridor plans. As reported earlier, there is interest in a Statewide assessment but State funding is a problem at present.

TVA has been active in recent years in encouraging communities to look at designating greenways as devices to preserve floodplains and create recreational opportunities. Agency personnel helped in producing plans for Chattanooga and Kingsport and are presently providing assistance for several other greenway planning efforts.

## LOCAL REGULATORY AND PROTECTION TECHNIQUES

There are several types of regulatory and protection techniques for river corridors. They can be integrated or juxtaposed with the more widely used flood loss reduction regulatory measures.

Amendments to existing floodplain regulations and sanitary codes and standards for subdivision development can be enacted to protect wetlands and other natural areas. Other techniques may include public and private acquisition, tax incentives, public education, and special efforts and actions by citizen advocates.

## SOME STEPS TOWARD MULTIOBJECTIVE MANAGEMENT

Multiobjective management of the State's river corridors will require involvement of many disciplines, governmental levels, and groups and interests. These include local officials and other decisionmakers; environmental groups, interests, and advocates; floodplain managers, wetlands managers, and other natural resources managers at all governmental levels; and overall political and public interest and support. The State's stream and river corridors are important and valuable natural assets well worth protecting through sound planning and wise management.

# IMPROVING RESERVOIR RELEASES AT TVA DAMS

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## INTRODUCTION

The Tennessee Valley Authority (TVA) was created by an Act of Congress in May 1933. Since then, the agency has built and acquired 48 dams which are operated as a unified system to realize multiple public benefits. The system was designed to provide for navigation, flood control, and the production of hydroelectricity. More recent projects have included purposes such as water quality, water supply, and recreation. Other functions such as fishery and wildlife management, wastewater disposal, and commercial development of various kinds have become important. Releases from TVA dams are low in dissolved oxygen (DO) during much of the summer and early fall. In addition, releases are intermittent so that streams below the tributary dams remain dry for extended periods. These conditions are stressful and have an adverse impact on the number, diversity, and health of the aquatic life in the tail- water areas below the dams. DO is not only essential to fish life, but to the entire aquatic community: the plants, insects, and microorganisms, all of which are vital in a healthy stream. Minimum flows provide an expanded and more stable habitat for aquatic life.

Providing for DO and minimum flows was not a consideration when the dams were built. Consequently, the projects must be retrofited and operations modified to obtain improvements. Conditions are unique at each project and significant adaptation is usually required to achieve satisfactory levels of improvements. Further, retrofiting is expensive and virtually any change in operation adversely impacts power production.

TVA's Reservoir Releases Improvements (RRI) program identifies, evaluates, and implements promising techniques to enhance reservoir releases, particularly in terms of DO and provision of minimum flows. Improvements in DO have been accomplished by schemes that increase air flows in the turbines and ones that operate in the reservoir. Implemented alternatives are considered as introductory until impacts have been observed and the devices are shown to have been adapted satisfactorily into the operation of the dam. The RRI program was initiated in 1981 and nearly $20 million has been spent by TVA on these activities including approximately $7 million in capital costs.

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## CURRENT RRI PROGRAM ACTIVITIES

Norris Dam is located on the Clinch River about 20 miles northwest of Knoxville, Tennessee. The dam is about 265 feet high and controls the runoff from a 2,919-square-mile drainage area. There are two hydropower turbines installed at Norris with a combined discharge capacity of approximately 9,000 cfs. The low temperature of the water released makes the river suitable for a put-and-take trout fishery. During the spring and summer, the reservoir becomes stratified and oxygen in the hypolimnion becomes depleted. Hub baffles are installed each year on both turbines to improve the DO in the releases. The baffles are of a semicircular wedge shape made of sections of 10-inch pipe and bolted over the vacuum breaker ports on the turbine hub. The baffles divert water flow out from the port, thus creating a partial vacuum. The partial vacuum increases air flow into the water over the full range of turbine operation and increases the DO level about 3.5-4 mg/L. A 400-foot- long, 5-foot-high reregulating weir has been built 2 miles below Norris Dam for use in sustaining a minimum flow in the river. One turbine is operated a minimum of 30 minutes every 12 hours whenever generation is not otherwise scheduled to maintain the pool behind the weir. Pipes through the weir release water to provide a minimum flow.

Tims Ford Dam is located on the Elk River about 70 miles southeast of Nashville, Tennessee. The dam is about 175 feet high and controls the runoff from a 529-square-mile drainage area of south central Tennessee. The large hydropower turbine is normally operated with a discharge of 3,900 cfs. The low temperature of the water released makes the river suitable for a put-and- take trout fishery. During the spring and summer, the reservoir becomes stratified and the oxygen in the hypolimnion is gradually depleted. Releases from Tims Ford are typically lower in DO and for longer periods of time than for any other dam release in the TVA system. A 10 psig, 3,000 scfm air com- pressor has been installed to aerate the releases from the large turbine during the period each year when they decline to unsatisfactory levels. The compressed air system has the capability to improve the DO level in the turbine releases by up to 4½mg/L. A 500-kw hydropower unit has been installed at the dam for use in sustaining a minimum flow. The small unit was origi- nally purchased by TVA for using as a cooling tower makeup pump/motor at Hartsville Nuclear Plant. It was purchased by the RRI program after the nuclear plant was canceled and the cooling tower pumps were declared surplus. This small unit operates whenever the large unit is not operating, which is about 75 percent of the time. Two small compressors have been added to the small turbine to aerate its releases during the low DO season.

South Holston Dam is located on the South Fork Holston River about 110½miles northeast of Knoxville, Tennessee. The dam is about 285 feet high and controls the runoff from a 703-square-mile drainage area. The hydropower turbine has a discharge capacity of about 3,000 cfs. The releases remain sufficiently cold to support a viable trout fishery below the dam. Histori- cally, DO in the releases is below satisfactory levels during a 13-week period from September through November. An air compressor was operated for three years on the single turbine at the dam to improve DO levels. During the evalu- ation of the compressor, it was observed that the turbine with the compressor shut off was capable of drawing large quantities of air through the aeration piping system over much of the unit's operating range. The turbine's inherent ability to aspirate is now used and the compressor has been remove. It is planned to begin construction of a labyrinth reregulating weir next spring about a mile below South Holston Dam. This weir combined with a pulsing operation at South Holston will sustain a minimum flow in the river. This weir will also aerate turbine discharges by creating a long and relatively thin sheet waterfall that will plunge about 4 feet to the water surface below. The water will pick up air as it falls into the plunge pool below the weir and will remain long enough to be dissolved.

Tellico Dam is located on the Little Tennessee River about 30 miles southwest of Knoxville, Tennessee. The dam is about 129 feet high and controls the runoff from a 2,627-square-mile drainage area. A canal connects Tellico Reservoir to Fort Loudoun Reservoir. There are no hydropower units in Tellico Dam. Normal flows from the reservoir pass through the canal and then through the Fort Loudoun hydropower turbines. Provisions have been made to release cold water from Tellico Reservoir into the Little Tennessee River to help provide a cool water refuge for stripped bass. During the spring and summer, a siphon discharges cool water onto one of the ogee sections of the spillway. The flow ducks under the surface because of its greater density, travels downstream, and backs up behind an underwater barrier dam constructed near the mouth of the river. The refuge consists of the deeper cool water behind the barrier dam.

Douglas Dam is located on the French Broad River about 26 miles east of Knoxville, Tennessee. The dam is about 202 feet high and controls the runoff from a 4,541-square-mile drainage area. The four hydropower turbines have a total discharge capacity of about 17,000 cfs. During the spring and summer, the reservoir becomes stratified and oxygen in they hypolimnion is gradually depleted. A warm water fishery exists below the dam. Multi-year tests are being conducted in an effort to sustain a minimum flow and to evaluate various techniques for improving the DO in the releases. A minimum flow of about 585½cfs has been sustained since 1987 by pulsing one turbine for about 30 minutes once every 4 hours when Douglas generation is not otherwise scheduled. Surface water pumps and a high purity oxygen diffuser system are being tested to evaluate their effectiveness in increasing DO levels in the releases. Surface water pumps have been installed in front of units 2 and 4. These low-speed axial pumps are used to push highly oxygenated surface water into the turbine intake flow when the reservoir is thermally stratified. Diffuser racks of the oxygen diffuser system have been place near the bottom of the reservoir in front of unit 4, well below the turbine intake area. High purity oxygen is bubbled through flexible diffuser membranes and stays in contact with the water long enough to become dissolved. Operation of the diffusers results in an upwelling current which is pulled into the turbine intake flow. Tests conducted last summer revealed the surface water pumps and the diffuser system were less effective than anticipated. It is hoped that each of these techniques will increase DO by 2 mg/L.

Cherokee Dam is located on the Holston River about 28 miles northeast of Knoxville, Tennessee. The dam is about 175 feet high and regulates the flow from a 3,428-square-mile drainage area. The four hydropower turbines have a total discharge capacity of about 18,000 cfs. During the spring and summer, the reservoir becomes stratified and oxygen in the hypolimnion is gradually depleted. A warm water fishery exists below the dam. Multi-year tests are being conducted in an effort to sustain a minimum flow and to evaluate various techniques for improving the DO in the releases. A minimum flow of about 325 cfs has been sustained since 1988 by pulsing one turbine for about 30 minutes once every six hours when Cherokee generation is not otherwise scheduled. It is planned to test two small surface water pumps this summer to evaluate their effectiveness in increasing the DO in the releases from unit 4.

## RESULTS

Tables 1 and 2 provide data showing the performance of the modifications described above. The performance of the DO devices at the Douglas and Cherokee projects are under assessment.

Table 1

Dissolved Oxygen Improvements

| Project | Average Number of Days per Year Dissolved Oxygen Concentration in Releases are Less Than the Selected Levels (mg/L) | | | | |
|---|---|---|---|---|---|
| | 6 | 5 | 4 | 3 | 1 |
| Norris | | | | | |
| Prior to improvements | 131 | 100 | 84 | 55 | 9 |
| After improvements | 55 | 29 | 4 | 0 | 0 |
| Tims Ford | | | | | |
| Prior to improvements | 183 | 160 | 124 | 95 | 0 |
| After improvements | 183 | 130 | 35 | 0 | 0 |
| South Holston | | | | | |
| Prior to improvements | 72 | 53 | 35 | 26 | 2 |
| After improvements | 47 | 36 | 8 | 1 | 0 |

Table 2

Minimum Flows

| Project | Minimum Flow (cfs) |
|---|---|
| Norris | |
| Prior to improvements | 50 |
| After improvements | 200 |
| Tims Ford | |
| Prior to improvements | 25 |
| After improvements | 75 |
| Tellico | |
| After improvements | 8 |
| Douglas | |
| Prior to improvements | 50 |
| After improvements | 585 |
| Cherokee | |
| Prior to improvements | 50 |
| After improvements | 325 |

Note: The minimum flows prior to improvements consisted of leakage through and around the dams. The minimum flow at Tellico is provided only during the warmer months.

# A STUDY OF THE TREATMENT AND RE-USE OF COOLING TOWER BLOWDOWN WATER AT GENERAL MOTOR'S SATURN PLANT

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The cooling tower at General Motor's Saturn Plant in Spring Hill, Tennessee will produce blowdown wastewater at rates as high as 500,000 gallons per day. This wastewater is currently discharged to the Columbia POTW, resulting in significant wastewater disposal costs, as much as $90,000 per year, and reduction of sewer capacity. The University of Tennessee and Saturn Corporation performed a feasibility study for several alternatives proposed for managing the blowdown water. The recommended alternative was treatment of the blowdown water on site in a constructed wetlands, and with plant stormwater runoff, spray irrigation of crops farmed by Saturn. The advantages of such a system include: water conservation, improved crop yields, reduction of wastewater disposal charges, reduction of stormwater impact on the environment and conservation of sewer capacity.

The cooling tower blowdown water quality is a function of city water quality used in the cooling system, the evaporative losses during operations of the cooling tower and the addition of a corrosion inhibitor and biocide. The concentrations of pollutants contributed by the city water are simply the water quality characteristics of the city water concentrated by the number of cycles of concentrations, i.e. evaporation in the tower. Both a three and five cycle scenario was being considered by Saturn, but for this study the five cycle scenario was used because the contaminants were more concentrated, i.e. worst case. The projected cooling tower blowdown water quality was supplied by Saturn and is shown in Table 1. The constituents listed as less than a value, are listed as five times the minimum detectable concentration in city water. The actual concentrations may be significantly less than the stated values. The organic compounds contributed by the corrosion inhibitor and biocide are not shown in Table 1.

The impact of the organic chemicals contributed by the biocide and corrosion inhibitor was determined by reviewing relevant literature and toxicity information from the chemical supplier. For those compounds with little or no information available, the octanol/water partition coefficient, a predictor of

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a chemical's behavior in the environment, was calculated from a correlation with the chemical's structure (Hansch and Leo, 1979). Because the biocide will be used in shock concentrations, the study recommended directing the blowdown water to the sewer during use of the biocide. The compounds in the corrosion inhibitor were not expected to adversely effect the environment because of one or more of the following reasons for each compound: low toxicity, low bioaccumulation potential, or effective removal in constructed wetlands by adsorption or degradation.

To assess the impact of common pollutants on crops, irrigation water quality standards have been developed for a number of common constituents (Wescot and Ayers, 1985). These irrigation water quality standards are based on the most likely damaging effect of a given pollutant and the most sensitive plant species. In some cases, soil characteristics are considered in establishing limits for pollutant buildup in soils. In most instances, the limiting pollutant concentration is based on introduction of pollutants into the food chain.

The irrigation water quality standards are considered safe for long-term irrigation of all plant species under normal conditions. Exceeding these limits for short periods or increasing the limit for certain situations is not necessarily unfavorable, and exceptions may be warranted for individual cases. As a result of the comparison of blowdown water quality with irrigation water quality limits, the pollutants in the blowdown water were divided into three categories: those with concentrations below irrigation water quality standards, those with concentrations above irrigation water quality standards, and those with unknown irrigation limits. The majority of the pollutants in the blowdown water were below irrigation water quality standards. Several pollutants did not have limits listed, but were not considered to pose significant environmental problems. The pollutants with concentrations in excess of identified limits are shown in Table 2.

A constructed wetlands should remove the majority of metals (Gersberg, et al, 1984). The constructed wetlands recommended for Saturn is effective for removal of several metals, so the copper, mercury and cadmium which are above irrigation limits should be effectively removed. As noted in Table 1, the concentrations of cadmium and mercury are five times the minimum detectable concentration in the city water. The actual concentrations are probably below irrigation limits. Of the other pollutants above irrigation limits, TDS and bicarbonate may not be removed efficiently in a constructed wetlands. Bicarbonate may actually increase from biological activity or contact with limestone media in the wetlands. However, because of the definitions of the limits (on a scale of no restrictions, slight restrictions, severe restrictions both pollutants are in the middle classification), neither of these pollutants should pose a problem for crop irrigation, especially, if as recommended, quarry or stormwater is used to supplement the irrigation water which will reduce the concentrations of pollutants.

The study recommended use of stormwater at a rate sufficient to reduce the fluoride level from the blowdown water to less than the recommended irrigation water quality. No published data was identified documenting fluoride removal in a constructed wetlands. However, fluoride is very soluble and other monovalent species such as sodium are not removed efficiently. Fluoride then is thought to be the controlling parameter, requiring reduction by mixing with stormwater on the order of 3:1 to 5:1 (ratio of volume of stormwater and blowdown water to the volume of blowdown water). The required volume of stormwater is based on the stormwater not containing a significant concentration of fluoride.

The type of constructed wetlands recommended for Saturn is a Subsurface Flow System (SFS). However, a small portion of the wetlands should be devoted to a Free Water Surface System (FWS) to enhance photodegradation of the organic compounds. An SFS is a shallow, typically 2 to 3 feet deep, wetlands with a porous media, either gravel, soil or sand, to support emergent plant growth. Actual water flow remains below the surface so there is no free standing water. An SFS is recommended for Saturn primarily because of the metal concentrations in the blowdown water. Adsorption/ion exchange is an important removal mechanism for metals, as well as other pollutants, and is facilitated in an SFS because of the high surface area per unit volume of wetlands. An SFS is initially more expensive than a free water surface system (FWS), the other predominant type of wetland, because of the bed media cost. An SFS may be operated at a higher loading rate and thus require less land area for a given wastewater flow. SFS also exhibit less problems with mosquitos.

Although loading rates for constructed wetlands reported in the literature are widely variable, approximately 10 acres is recommended for treating blowdown water at the summer average rate of approximately 500,000 gallons per day, which is based on a cooling tower operating with three cycles. If the cooling tower is operated with five cycles, the average summer blowdown rate will be approximately 250,000 gallons per day, and the resulting wetland size would be 5 acres. Sizing the wetlands for the peak summer blowdown rate is not necessary because the loading rate is not rigorously derived, and a higher loading rate may work as effectively. In addition, the supplemental stormwater/make-up water should reduce pollutants below irrigation limits if increasing effluent concentrations are encountered during peak blowdown periods. During off-peak periods, the remaining wetlands area may be used to treat stormwater runoff or other Saturn wastewaters.

The costs for constructed wetlands reported in the literature are also widely variable but $20,000 per acre seems to be a fairly typical value, though higher values have been reported. TVA reported wide variances in cost per acre for constructed wetlands for treating mine and ash pond drainage, but felt $40,000 per wetland was a typical value (Brodie, et al, 1988). Almost all of the cost of a constructed wetlands is encumbered in the initial construction. Maintenance costs are typically only a few thousand dollars per wetlands per year.

The study concluded that the treatment of the blowdown water in a constructed wetlands followed by spray irrigation of crops is a viable alternative with many advantages. Elimination of the constructed wetlands may be possible with careful monitoring to allow irrigation with wastewater in excess of established limits. Several unknowns regarding constructed wetlands exist, including the long-term metal removal capacity. Construction of a pilot scale constructed wetlands was recommended to establish loading rates and verify the pollutant removal efficiencies reported in the literature. The project provided an excellent opportunity for UT faculty and students to work with Saturn to identify innovative, economic and environmental alternatives in managing wastewater.

## REFERENCES


Brodie, G. A., et al., 1988. Constructed Wetlands for Acid Drainage Control in the Tennessee Valley. In: Mine Drainage and Reclamation, Bureau of Mines Information Circular IC 9183.

Gersberg, R. M., et al., 1984. The Removal of Heavy Metals by Artificial Wetlands. In: Future of Water Reuse Symposium III. pp. 639-648.

Hansch, C. and A. Leo, 1979. Substituent Constants for Correlation Analysis in Chemistry and Biology. John Wiley and Sons. New York.

Wescot D. W. and R. S. Ayers, 1985. Irrigation Water Quality Criteria". In: Irrigation with Reclaimed Municipal Wastewaters. Lewis Publishers, Chelsea, MI.

Table 1

Projected Blowdown Water Quality

| Component (mg/l) | Concentration |
|---|---|
| Silica | 2 |
| Total Organic Carbon | 77 |
| Total Dissolved Solids | 1116 |
| Sulfate | 290 |
| Chloride | 50 |
| Carbonate | 5 |
| Bicarbonate | 712 |
| Nitrogen-Nitrate | 4 |
| Fluoride | 5 |
| Turbidity | 1 NTU |
| Phosphates-Total | 2 |
| Hardness (as $CaCO_3$) | 590 |
| pH | 7.8 |
| Alkalinity (as $CaCO_3$) | 720 |
| Calcium | 320 |
| Magnesium | 28 |
| Iron | 0.03 |
| Manganese | <0.025 |
| Beryllium | <0.005 |
| Copper | 0.7 |
| Nickel | <0.025 |
| Aluminum | 1.6 |
| Sodium | 64 |
| Potassium | 6.5 |
| Antimony | <0.025 |
| Cadmium | <0.025 |
| Chromium-Total | <0.025 |
| Lead | 0.04 |
| Mercury | <0.001 |
| Selenium | <0.005 |
| Silver | <0.025 |
| Thallium | <0.025 |
| Zinc | <0.025 |
| Arsenic | <0.005 |

Table 2

Pollutants with Concentrations above Irrigation Water Quality Limits

| Component | Concentration (mg/l) | Limit[2] (mg/l) | Limit[3] (mg/l) |
|---|---|---|---|
| TDS | 1166 | 500[4] | |
| Bicarbonate | 712 | 90[5] | 150[6] |
| Fluoride | 5 | 1 | 1.8 |
| Copper | 0.7 | 0.2 | 0.4 |
| Mercury | <0.0010 | .0009 | |
| Cadmium | <0.025 | 0.01 | 0.02 |

[2] Wescot, D. W. and R. S. Ayers. "Irrigation Water Quality Criteria" In: Irrigation with Reclaimed Municipal Wastewater. Lewis Publishers. Chelsea, MI. 1985.

[3] Tennessee Department of Health and Environment, Chapter 16, Slow Rate Land Treatment in Design Criteria of Sewage Treatment Works.

[4] Limit is defined as <500 no restrictions, 500-2000 slight to moderate restrictions, >2000 severe restrictions.

[5] Limit is defined as <90 no restrictions, 90-500 slight restrictions, >500 severe restrictions.

[6]Limit is defined as <150 no problem, 150-850 increasing problems, >850 severe problems.

# MITIGATING ENVIRONMENTAL IMPACTS OF A GRAVITY SEWERLINE INSTALLLATION IN THE CARTERS CREEK WATERSHED, MAURY COUNTY, TENNESSEE: A CASE STUDY

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## INTRODUCTION

The City of Columbia, Tennessee, constructed a gravity sewerline, 3.6 miles in length, over the period of July to September, 1988, providing sanitary sewer service to the Saturn Corporation site and accomodating anticipated growth in the Carters Creek watershed.

The major environmental concern was the necessity to construct sewer crossings under Carters Creek at twelve locations and under two tributary channels; the construction had the potential to damage the stream's natural habitat, via erosion and sedimentation, and to impact the stream's designated uses for fish and aquatic life, recreation, irrigation, livestock watering, and wildlife.

Carters Creek, a tributary to the Duck River drainage basin, is a clear, shallow, fairly narrow, low velocity stream, with a drainage area of approximately 35 square miles and a 3Q20 of zero. The predominant substrate is cobble or bedrock overlain by cobble. The area surrounding the creek is sparsely populated and primarily used for agriculture.

Saturn Corporation, a greenfield automobile manufacturing facility located in Spring Hill, Tennessee, examined the potential environmental impacts of construction, and designed an erosion and sedimentation control plan to prevent and minimize impacts.

## METHODS AND PROCEDURES

Mitigation was accomplished through:

- Minimizing equipment in Carters Creek and limiting the work area to that specified on design drawings;
- Stockpiling excavated soil in areas upstream of crossings and sedimentation control devices;
- Installing a device for allowing equipment crossings of creek;
- Using silt fence along all downgrade edges of construction to prevent silt laden storm runoff from entering the creek;

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- Placing sandbag cofferdams upstream and downstream of an active creek crossing, and channeling the stream through a pipe;
- Placing a sediment boom, having fabric similar to a silt fence, downstream of the crossing area;
- Pumping water from the crossing trench area to a dewatering pit, located on the downgrade of the creek crossing trench, where silt settled out before the water returned to the creek; typical size was 500 cubic feet;
- Placing silt fence downstream of the dewatering pit;
- Conducting crossings during the low flow season;
- Establishing spill prevention procedures for fuel trucks;
- Restoring the stream bed and banks following creek crossings; bank stabilization with riprap and geotextiles;
- Restoring adjacent areas with landscaping; and
- Sodding, seeding, and mulching of work areas

To oversee these measures, the flow rate of Carters Creek was monitored daily from a U.S. Geological Survey Gage Station, consisting of a concrete weir, crest stage gage, and water stage recorder. Weather data, collected daily from an on-site weather monitoring station, included wind speed, wind direction, temperature, precipitation, and relative humidity. Saturn personnel inspected construction work daily. Water samples, used to monitor the overall water quality of Carters Creek, as well as the water quality above and below each creek crossing, were collected periodically and analyzed for TSS and turbidity. A visual record of the work was maintained through pre-construction, active, and post-construction phases.

## RESULTS

Figure 1 summarizes stream flow rates over the construction period. The low flow rates of Carters Creek reflect the extremely dry weather experienced. Precipitation totaled 7.2 inches recorded over the construction period of July 5 to September 19, 1988, approximately half of typical seasonal values.

Results indicate that the overall sediment load to Carters Creek due to the gravity sewerline project was negligible compared to the natural load from tributaries to Carters Creek. There was no significant difference between TSS and turbidity above and below an active creek crossing. The discharge from the dewatering pit was initially found to be high in TSS and turbidity, however, changes such as additional placement of silt fencing and routing the dewatering pit discharge upstream of the sediment boom further improved control efforts.

Water quality conformed with Tennessee State Water Quality Criteria for all stream designated uses: "There shall be no turbidity or color in amounts or characteristics that cannot be reduced to acceptable concentrations by conventional water treatment processes...that will materially affect fish and aquatic life...that will result in any objectionable appearance in the water." No value or range is specified for Total Suspended Solids.

TABLE 1

BASELINE WATER QUALITY DATA
CARTERS CREEK

| sampling date | TSS (mg/L) N | TSS (mg/L) X | TSS (mg/L) SD | Turbidity (NTU) N | Turbidity (NTU) X | Turbidity (NTU) SD |
|---|---|---|---|---|---|---|
| September 1985 | 4 | 59.0 | 55 | 4 | 4.3 | 2.1 |
| June 1986 | 5 | 3.0 | 2 | 3 | 8.7 | 1.2 |
| May 1987 | 2 | 7.5 | 1.5 | - | - | - |

N - number of samples
X - arithmetic mean
SD- standard deviation

TABLE 2

SUMMARY OF WATER QUALITY RESULTS

Total Suspended Solids (mg/L)

| Date | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| 7-13-88 | 20 | 196 | 10 | 30 | 374 | 200 |
| 7-21-88 | 18 | 6 | 4 | 7 | 12 | 9 |
| 7-28-88 | 9 | 5 | 13 | 7 | 7600 | 7 |
| 9-17-88 | 5 | 10 | 10 | 7 | 16 | 7 |

Turbidity (NTU)

| Date | 1 | 2 | 3 | 4 | 5 | 6 |
|---|---|---|---|---|---|---|
| 7-13-88 | 8.6 | 145 | 6.1 | 22 | 300 | 280 |
| 7-21-88 | 9.6 | 6.4 | 3.6 | 3.2 | 9.3 | 7.7 |
| 7-28-88 | 5.3 | 3.1 | 4.3 | 4.4 | >1000 | 5.5 |
| 9-17-88 | 3.6 | 2.2 | 3.5 | 2.6 | 11 | 2.3 |

SAMPLING LOCATIONS: (Fig. 2):

1. Carters Creek 200' above Titan Creek
2. Carters Creek 200' below Titan Creek
3. Carters Creek at USGS Gage Station
4. Carters Creek at Harlan Road
5. Unnamed tributary to Carters, 500'downstream of Harlan Rd.
6. Bridge at Carters Creek Pike

TABLE 3
WATER QUALITY AT CREEK CROSSINGS

Total Suspended Solids (mg/L)

| Date/Station Number | Upstream of Crossing Area | Downstream of Crossing Area | Silt Pit Discharge |
|---|---|---|---|
| 7-22-88 sta.28 | 17 | 7 | 181 |
| 7-22-88 sta.44 | 6 | 14 | 32 |
| 7-25-88 sta.28 | 11 | 37 | 184 |
| 7-25-88 sta.44 | 14 | 35 | 122 |
| 8-02-88 sta.55 | 4 | 8 | 91 |
| 8-10-88 sta.80 | 15 | 22 | 140 |
| 8-18-88 sta.111 | 14 | 36 | - |
| 8-22-88 sta.109 | 12 | 40 | 16 |
| 8-29-88 sta.155 | 3 | 10 | 11 |
| 9-02-88 sta.176 | 7 | 88 | 16 |
| 9-08-88 sta.205 | 6 | 45 | 28 |
| 9-13-88 sta.15 | 14 | 45 | 12 |

Turbidity (NTU)

| Date/Station Number | Upstream of Crossing Area | Downstream of Crossing Area | Silt Pit Discharge |
|---|---|---|---|
| 7-22-88 sta.28 | 12 | 6.5 | 118 |
| 7-22-88 sta.44 | 3.9 | 9.3 | 21 |
| 7-25-88 sta.28 | 6.0 | 28 | 96 |
| 7-25-88 sta.44 | 7.1 | 22 | 99 |
| 8-02-88 sta.55 | 1.6 | 7.4 | 54 |
| 8-10-88 sta.80 | 3.8 | 12 | 78 |
| 8-18-88 sta.111 | 3.5 | 7.3 | - |
| 8-22-88 sta.109 | 5.2 | 18 | 5.6 |
| 8-29-88 sta.155 | 5.0 | 11 | 9.9 |
| 9-02-88 sta.176 | 2.1 | 43 | 7.5 |
| 9-08-88 sta.205 | 2.3 | 18 | 15 |
| 9-13-88 sta.15 | 2.6 | 11 | 3.0 |

## SUMMARY AND CONCLUSIONS

On September 13, 1989, representatives from the City of Columbia, the Tennessee Wildlife Resources Agency, Saturn Corporation, and Memphis Construction performed an on-site inspection of the completed sewerline. Stream crossings, streambank stabilization, vegetation restoration and landscaping exceeded expectations. These mitigation measures are now required by the Corps of Engineers for similar creek crossings.

## REFERENCES


Saturn Corporation. 1986. Assessment of Water Quality in Carters, Rutherford, Titan, and McCutcheon Creeks in the Vicinity of the Saturn Corporation Site. Roy F. Weston, Inc., West Chester, Pennsylvania.

Saturn Corporation. 1987. Biological and Water Quality Investigation of Carters, Titan, and McCutcheon Creeks and the Duck River, Maury County, Tennessee. Roy F. Weston, Inc., West Chester, Pennsylvania.

Tennessee's Water Quality Criteria and Stream Use Classifications for Interstate and Intrastate Streams, Tennessee Water Quality Control Board, Rule No. 1200-4-3 (August, 1983).


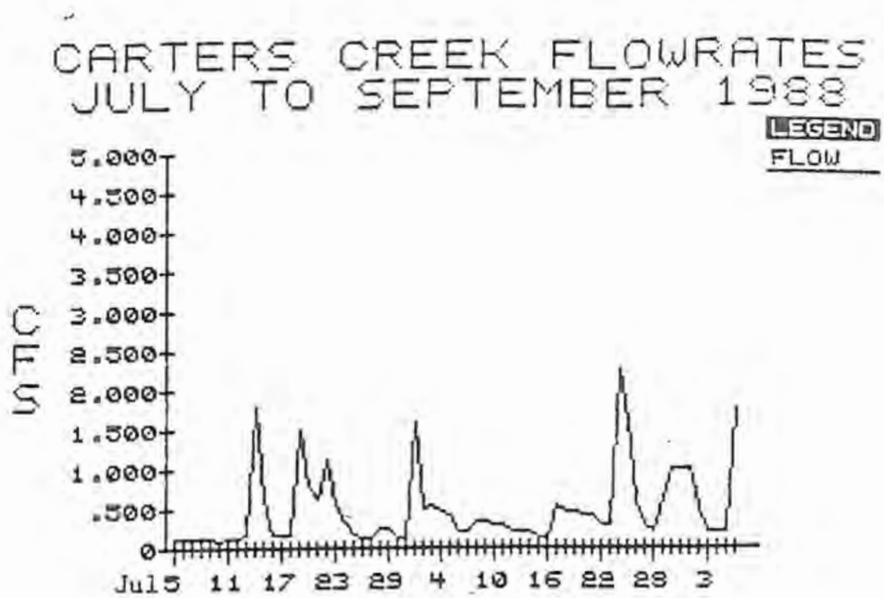


Figure 1. Carters Creek Flowrates July - Sept. '88

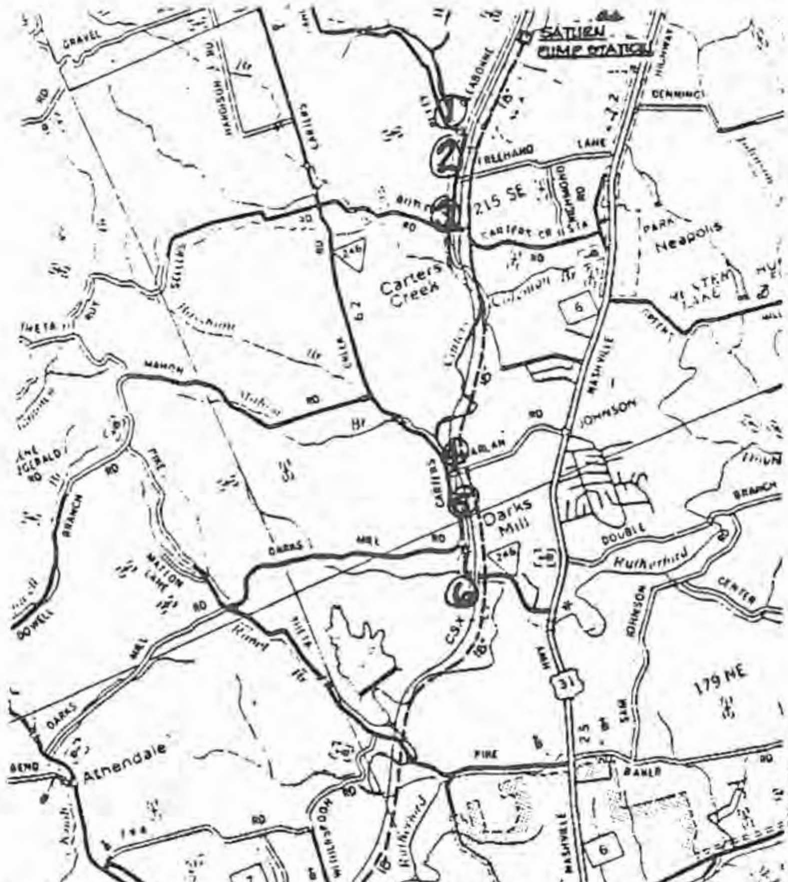


Figure 2. Water Quality Sampling Locations

# IS THERE LIFE IN OIL CREEK ? AN INTENSIVE STREAM SURVEY OF BROWNS CREEK DAVIDSON COUNTY - MAY, 1989

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## INTRODUCTION

Browns Creek system is composed of three major forks; the East, Middle and West. The middle and west forks converge near the junction of I-440 and I-65 in Nashville and flow for approximately a half a mile before reaching the confluence of the East fork. The middle and west forks are primarily surrounded by residential areas. In contrast, the east fork flows through a commercial and industrial watershed. Together, the three forks of Browns Creek flow northeast through a largely commercial and industrial area until reaching the Cumberland River.

During the past four years, the Division of Water Pollution Control has investigated over fifty complaints, and worked on as many as ten oil and/or chemical spills in the Browns Creek system. For approximately thirty years, the East Fork of Browns Creek (EFBC) has received discharges laden with oil, diesel fuel, and other associated contaminants from the Radnor railyard, now owned and maintained by CSX. The spring fed headwaters of EFBC are located just before the newly formed creek flows under Radnor Yard. This small creek sustains aquatic benthic organisms and therefore provides an opportunity to compare its conditions with those below the CSX discharge point. As the East Fork continues flowing towards the main creek, it receives permitted discharges of cooling water from the Baird-Ward plant located on Armory Drive, runoff from the parking lots at 100 Oaks Mall, I-65 and the other major roadways around the area.

In an attempt to quantify the extent of degradation in EFBC, an intensive stream survey was conducted by TDHE Division of Water Pollution Control personnel in May, 1989. The survey involved collecting macroinvertebrate samples, taking physical/chemical water quality measurements and performing habitat assessments at six designated sites in the Browns Creek system (see figure 1).

## METHODS

Water quality field measurements of pH, conductivity, temperature and dissolved oxygen were collected using properly calibrated meters (Orion, YSI & YSI respectively) at each station on May 24, 1989. An oil & grease sample was

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collected from the site directly below the railyard in a properly prepared bottle and transported to the state lab for analyses.

Macroinvertebrate benthic samples were collected at the six stations using quantitative and qualitative sampling techniques. The quantitative samples included three replicates of one square foot Surber samples from each station. Multihabitat pickings, were done to sample all available habitats. Kick screens, kick nets and forceps were employed for these collections. The collected samples were placed in jars, preserved with 80% ethanol and transported back to the lab where the organisms collected were identified to the lowest taxa practical.

A habitat assessment was made at each station using a standardized format. The assessment included: evaluating general water and substrate conditions, recording types and abundances of substrate, noting the surrounding land uses, estimating canopy cover and identifying the quantity and availability of habitat for aquatic macroinvertebrates.

## RESULTS

There were only slight variations in pH, dissolved oxygen, temperature and conductivity, among the six stations sampled. Average values were 8.0 units, 8.9 mg/l, 18.0 deg. C and 550 uhmos, respectively. All of the measured values were within the tolerance range of most aquatic organisms.

The results of the oil & grease and total hydrocarbon sample collected at site 003 show the extent of contamination in the East Fork. Thirtyfour percent (34%) of the sample was determined to be oil & grease and ten percent (10%) was hydrocarbons. The oils and petroleum products were degraded and mixed to the point that no one specific hydrocarbon could be identified. This represents the conglomeration of material getting into EFBC.

In the EFBC, station 002 (at the headwaters), had the greatest number of taxa and had the only intolerant and/or facultative organisms in the reaches of study. The invertebrate population at the station directly below the railyard discharge, (003), was entirely comprised of tolerant oligochaetes and Chironomids. There was no evidence of recovery at the station located near the confluence with Browns creek proper, where again, only tolerant organisms were present.

The benthic macroinvertebrate data collected at the two stations located in Browns creek proper showed the improvement in water quality over that in EFBC. The data also showed the impact that the EFBC has on the main stream. At the site above the confluence, 44 taxa were collected, - 84% of which were classified as either intolerant or faculative. In contrast, at the station below the conluence, only 21 taxa were found with just 48% of the organisms being classified as intolerant or facultative. The water quality data did not indicate any major changes in pH, dissolved oxygen, conductivity or temperature.

A control station (007) was located in the upper reaches of the West Fork of Browns Creek. Thirty (30) taxa were collected with 62% of them being intolerant or facultative.

Figure 2 summarizes the data from EFBC and the control site (007) by showing total number of identified taxa at each station. Figure 3 shows total number of taxa identified for Browns Creek sites 005 & 006 while comparing them to EFBC site 004. In figure 4 one can see a summary of all of the stations sampled in the survey. The graph shows the tolerance classifications of the taxa found and the wide variation between the control sites and the impacted sites.

The most observable difference among the stations was the presence of, at times, large amounts of oil and petroleum products (sites 003, 004 and 006). Station 003 was heavily contaminated with oil and other petroleum products in both the water and sediments. Oilsheens and petroleum odors were also evident at stations 004 and 006. Small, localized oil sheens were observed in backwater areas near site 005, apparently from runoff of the interstate and railroad bridge.

Toxicity tests performed on the subnate of an oil/water sample collected below the railyard discharge resulted in LC 50 values of 1% for Ceriodaphnia and 10% for fathead minnows. Fish were observed at sites 005, 006 and 007, but were clearly absent from all other sites. Behavioral avoidance of the EFBC by the fish was noted at the confluence with Brown's Creek proper.

## DISCUSSION

The results from this study confirm past studies (completed by Metro, 1987 and TNDHE,WPC, 1988) that the EFBC is a severely impacted system. Further, EFBC is so contaminated that it degrades the water quality at and below its confluence with Browns Creek proper.

These conclusions are supported by the benthic data. At the headwaters site (002), the stream is so small that it is very limited in all habitats, including water. Despite this, the stream supported almost three times as many different taxa as site 003, and had organisms in all three tolerance classifications. Station 003 had only tolerant organisms. The conditions remain about the same throughout the EFBC as evidenced by the results at site 004.

Site 005, in the stream proper, is located in roughly the same geographic area as 003 and has similiar physical characteristics yet, at 005 there were eleven times as many taxa as at 003. It becomes obvious that anthropogenic stresses are dictating the environmental conditions in the creek. This scenario is repeated when one looks at the differences in the number of taxa and number of organisms between sites 005 and 006. Site 006 is only about 200 yards downstream from 005, but the number of taxa is cut in half and the number of intolerant taxa drops from six (6) taxa to one (1) taxa. It appears that the introduction of EFBC waters to the system stresses and degrades the water quality of Browns Creek (site 006).

## CONCLUSIONS

EFBC has experienced severe degradation resulting in 1) loss of natural habitat, at site 003 and below, from continuous discharges of oil, petroleum and other products from Radnor yards, 2) loss of fish and aquatic life from the point of CSX's discharge to the confluence with Brown's Creek proper, 3) degradation of water quality to the extent that EFBC no longer supports any of its stream use classifications, and 4) EFBC acting as a source of contamination, as a whole, to Brown's Creek. This area needs continuing monitoring and requires immediate, effective and concentrated efforts to eliminate further contamination from Radnor Yards.

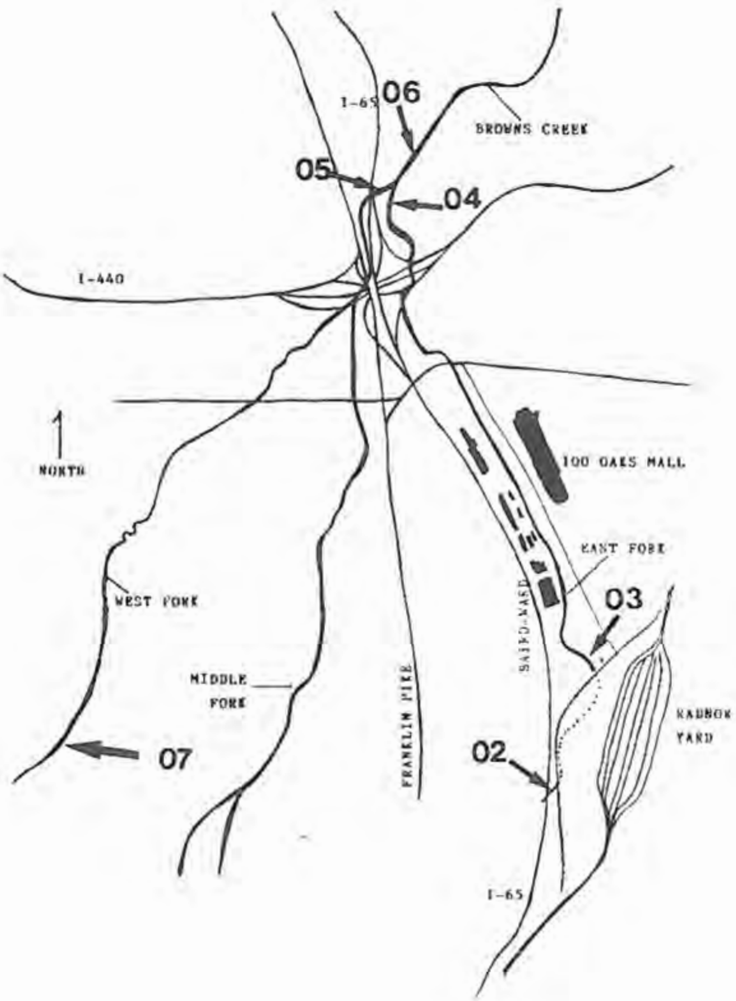


Figure 1. Station Locations for Browns Creek Instream Survey. May, 1989,

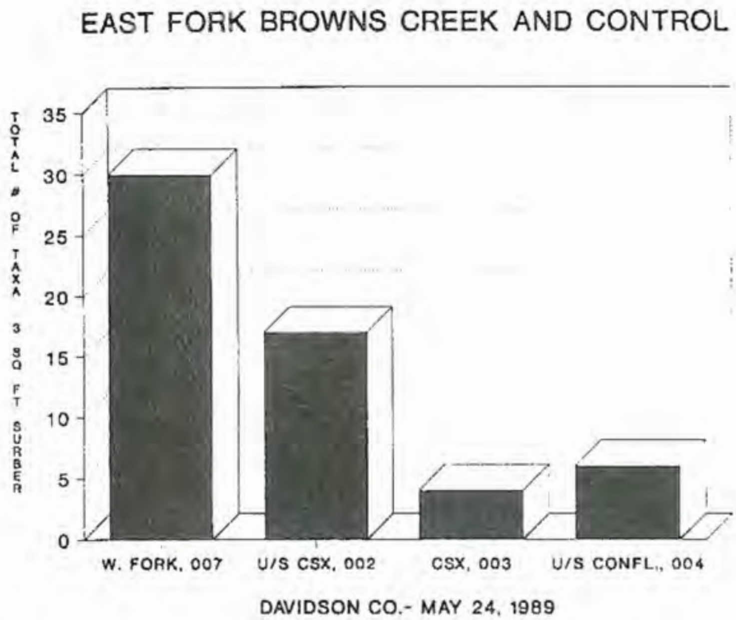


Figure 2. East Fork Browns Creek and Control.

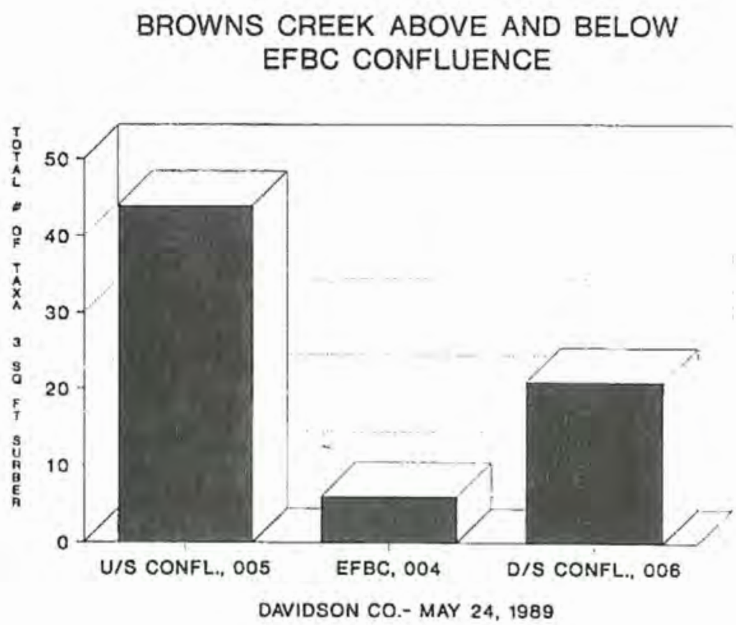


Figure 3. Browns Creek Above and Below EFBC Confluence.

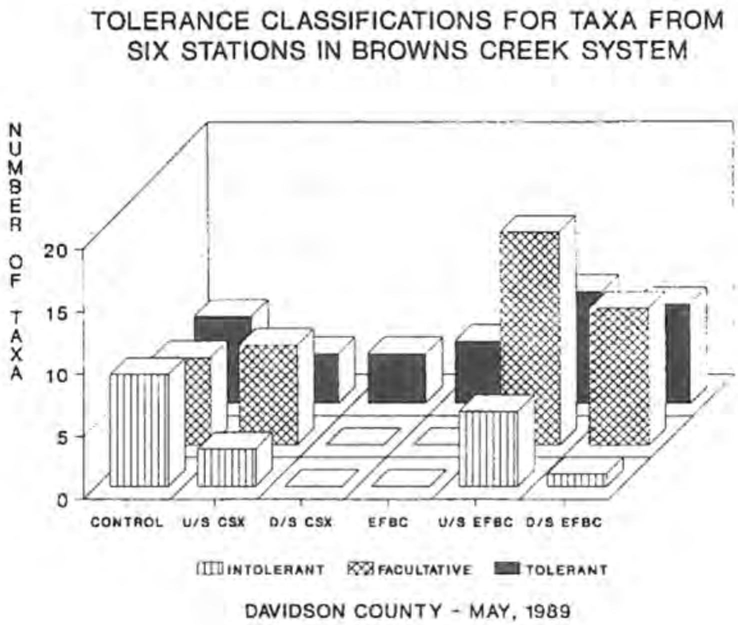


Figure 4. Tolerance Classifications for Taxa From Six Stations in Browns Creek System.

# THE IMPACT OF MANGANESE MOBILITY IN SEDIMENTS IN FRESHWATER MUSSELS

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## INTRODUCTION

For over 30 years, declines in the populations of freshwater mussels have been noted throughout the Tennessee River/Kentucky Lake system (Isom, 1969; TVA, 1987). During the drought years 1984-1988, thousands of mussels were found "gapped open" or found empty. These die-offs occurred in commercial mussel beds as well as in recreational and agricultural areas. Speculations about the decline and the die-offs have raised questions about sediment quality, water flow, oxygen depletion, microbial infection, industrial wastes and nonpoint source pollution. In a limited study at Memphis State University, Scholla et al. (1986) found only one consistent difference between mussels judged to be sick vs healthy: sick mussels contained 10 times more "yellow" gram-negative bacilli, dubbed "Yellow Colony Formers". The Yellow Colony data do not distinguish whether these bacteria are pathogens or are symptoms of other problems. In 1987 the Kentucky Reservoir Water Resources Task Force concluded that there was no obvious cause for the toxicity.

The data we present here are part of an interagency attempt to determine the status of the mussel population in this system and to determine what factors are most important. A major consideration was to control all specimens and samples from the point of collection through the analysis. Sampling sites were chosen both upstream (near Pickwick Dam) and downstream (near Paris Landing and in the Big Sandy region). Sampling started in March and continued into October, 1989. The protocols included the following measures: distribution of mussel species, sediment and water quality, [Mn] in sediments and pore water, mussel condition index (the ratio of soft tissue mass to shell volume), total bacteria and "Yellow Colonies" in mussel extracts, [Mn] in mussel tissue, and screening of sediments and mussels for organochlorine pesticides.

## RESULTS

The mussels appeared healthy in general. Only 16 mussels were found gapped open (with soft tissue still present). These represent less than 1% of all mussels surveyed at the sampling sites. 436 mussels (including those 16 gapped ones)

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were brought to MSU for analyses. Total bacterial counts averaged $10^6$ per gram of tissue; Yellow Colonies were 2% or less of the total counts and were often absent. There is no apparent relationship of these bacterial data to "gapping" or sites or species or condition index (ANOVA).

There were, however, at least three distinct differences among the sampling sites: 1) the species distribution, 2) the [Mn] in mussel tissue and 3) the [Mn] profile of sediment pore water. Sites in the Big Sandy region, in particular, had very few (only 4 or 5) mussel species, but those mussels had the highest [Mn] (mean=1395 ppm, N=40). Also, there were species differences, e.g., Quadrula quadrula (Maple Leaf, mean=1116ppm, N=89) versus Fusconaia ebena (Ebony Shell, mean=310ppm, N=45).

The pore water [Mn] ranged from <0.1 to 19ppm. The upstream (Pickwick) sites had the lowest values and showed depth profiles with low [Mn] near the sediment surface. Downstream sites in the main body of the river had comparable profiles. Sites in the Big Sandy, though, had high pore water [Mn] and had depth profiles with higher [Mn] near the surface. For most of 1989, the dissolved oxygen (D.O.) at the mussel beds was close to saturation (6-8 mg/l), as might be expected from the heavy rainfall and water flows. Even with the relatively high D.O., [Mn] mobilization toward the sediment surface was evident throughout the Big Sandy. In an isolated case of low D.O., recorded 100ft away from a Big Sandy mussel bed, the pore water [Mn] was the highest and [Mn] mobility greatly increased.

## SUMMARY

1) The sampling sites in the Tennessee River/Kentucky Lake system varied in sediment quality and mussel species diversity.

2) There were differences in [Mn] among the sites and among the mussel species.

3) [Mn] mobilization was evident at some sites even though the system showed relatively high D.O. during most of 1989.

4) Under low D.O. conditions, [Mn] mobility was greatly increased.

5) The sites with [Mn] mobilization support few mussel species. The combination of sediment quality, [Mn] mobility and D.O. fluctuations may limit both species diversity and viability.

6) The 1989 data establish a basis for evaluating changes in the Tennessee River/Kentucky Lake system. This may be particularly important during years of low rainfall and flow.

## REFERENCES

Isom, B. G., 1969. The Mussel Resources of the Tennessee River. Malacologia, 7:397-425.

Scholla, M. H., M. L. Hinman, S. J. Klaine, and J. Conder, 1986. Evaluation of a Mussel Die-Off in the Tennessee River in 1985. Proceedings, Conference on Mussel Die-offs in the Mississippi Basin, Davenport, Iowa.

Tennessee Valley Authority, 1987. Report for the Period September 1986 - September 1987, Kentucky Reservoir Water Resources Task Force.

# EFFECTS OF TWO ELECTROPLATING PLANTS ON THE MACROINVERTEBRATE COMMUNITIES OF CANE CREEK AND ITS TRIBUTARIES, LAUDERDALE COUNTY, TENNESSEE

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## BIOMONITORING AND BIOASSAYS

During the summer and fall of 1989, quantitative (Surber and Hester-Dendy samples) and qualitative (standard effort bank water interface samples) collections at thirteen stations showed impacts caused by two electroplating plants. Instream conductivities used as effluent concentration indicators were highly correlated (R2) with biometrics calculated from benthic data. Toxicity as determined in chronic tests using Pimephales promelas and Ceriodaphnia dubia was indicative of instream effects on benthos. When using qualitative bank samples, the tolerant Oligochaete, Air Breather, Chironomid percent of total sample number (OAC%) proved to be a simple but highly effective measure of biological impact when intolerant taxa were generally absent even at control sites. The OAC% was felt to be more precise than the more complicated and costly quantitative measures of diversity and evenness.

Cane Creek originates in the West Tennessee Uplands region east of Nelson's Chapel in eastern Lauderdale County. Mostly channelized, it flows generally southwest approximately 20 miles to its mouth on the Hatchie River at river mile 27.5. The principle tributaries to Cane Creek in the Ripley area are Nelson Creek (mile 12.2, has been relocated from old channel) and Hyde Creek (mile 9.3). Cane Creek, Nelson Creek, and Hyde Creek are classified by the Tennessee Water Quality Control Board for fish and aquatic life, recreation (except for 1 mile below Ripley STP), irrigation, and livestock watering and wildlife uses.

Cane Creek has historically experienced water quality problems and has been assessed in Tennessee's biennnial water quality survey on multiple occasions as not supporting designated uses (TDHE, 1986,1989). Within a three mile stretch of Cane Creek, direct and indirect discharges from three major facilities enter the creek near Ripley. These discharges are: the City of Ripley Sewage Treatment Plant (Ripley STP), Seigel-Roberts (S-R) of Tennessee, and Tennessee Electroplating (TEP). Ripley STP effluent enters Cane Creek at mile 11.1. S-R effluent flows approximately 1 mile down the Old Nelson Creek Channel and enters Cane Creek at mile 12.1. TEP effluent flows approximately 0.7 mile down an unnamed tributary and enters Hyde Creek 2.5 miles above its confluence with Cane Creek. The receiving streams have a 3Q20 of 0 cfs at the points of discharge. This paper emphasizes study parts concerning S-R and TEP.

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Whole effluent 7-day chronic toxicity tests using the Pimephales promelas (fathead minnow) survival and growth test and/or the Ceriodaphnia dubia survival and reproduction test performed during July of 1989 on S-R and TEP indicated toxicity at both facilities. In July 1989, S-R effluent was toxic to Ceriodaphnia dubia as determined by TDHE and EPA testing on split samples. Both agencies agreed on an acute and chronic No Observed Effect Concentration (NOEC) of 30% and a Lowest Observed Effect Concentration of 100%. Both agencies agreed that S-R's effluent was not toxic to fathead minnows during the sampling period. TEP effluent was toxic to fatheads in July, 1989 with a chronic NOEC of 30% and chronic LOEC of 100% but no lethality. A Ceriodaphnia test in October 1989 likewise showed a chronic NOEC of 30% and chronic LOEC of 100% but was also lethal to the small crustaceans at the 100% concentration. These and past toxicity tests performed on these effluents indicated a high potential for instream biological impacts especially when considering the lack of dry weather receiving stream dilution. A definitive benthological survey was therefore commenced in July 1989 and finalized in January of 1990.

Benthic macroinvertebrate collection stations were located in order to best facilitate determination of upstream downstream differences. Reconnaissance determined 2 control sites and 11 test sites (Figure 1.). Overall, habitat was poor and depth inconsistent from station to station. Field methods used negated the affects of habitat differences between stations. A common habitat at all stations was bank-water interface (bank) habitat characterized by similar mud and vegetation. The bank habitat was sampled using a standard effort kick net technique. Quantitative efforts (3 reps) included Hester-Dendy sampling in pools at all 13 stations and Surber sampling at 3 stations with suitable substrate (NN1,CC2,CC3). Bank samples were collected and Dendys set August 9,10,11. Bank samples were again collected, Dendys retrieved, and Surbers collected September 9 and 10. D.O., pH, cond, and temperature were recorded at each station visit.

Control stations NN1 and CC2 had stable benthic communities. Shannon Weaver diversity of Surber samples at control station NN1 was 3.9 and was 3.4 at CC2 while equitability was 0.8 and 0.7 respectively. Dendy diversity was 3.6 at CC2 and a perplexing 1.8 at NN1. Dendy equitability was a bewildering 0.4 at NN1 and 0.7 at CC2. Bank samples were diverse with 61 taxa found at NN1 and 35 taxa present at CC2. The mean percentage of tolerant Oligochaetes, Air-breathers, and Chironomids (OAC) to total taxa was 43% at NN1 and 65% at CC2 for bank samples.

The S-R sampling block stations (UTN1,ON1,CC3,CC4) downstream showed depressed benthic communities as compared to the control stations (NN1,CC2). ON1 showed an extremely depressed community with a diversity of 0.3. The OAC was 96% being composed almost entirely of tolerant Chironomus (1236/ft2). Metrics improved to a mediocre diversity of 1.7 and OAC of 86% at CC4. Instream conductivities in the vicinity of S-R ranged from 3580 umhos/cm3 to 295 umhos/cm3 at CC2. Data from NN1, ON1, CC2, CC3, and CC4 showed conductivity (receiving stream waste concentration indicator- X) to be correlated (R2) with biometrics (Y).
R2 was -0.7 for Dendy diversity, -0.8 for Dendy equitability, 0.9 for August's bank OAC%, and 0.7 for September's OAC%. S-R was the only source of high conductivity waters within this stream sampling section.

The TEP sampling block included stations HC1, UTH1, HC2, NN1, and CC2. HC1 (upstream) was initially considered as a control station but was found to be of

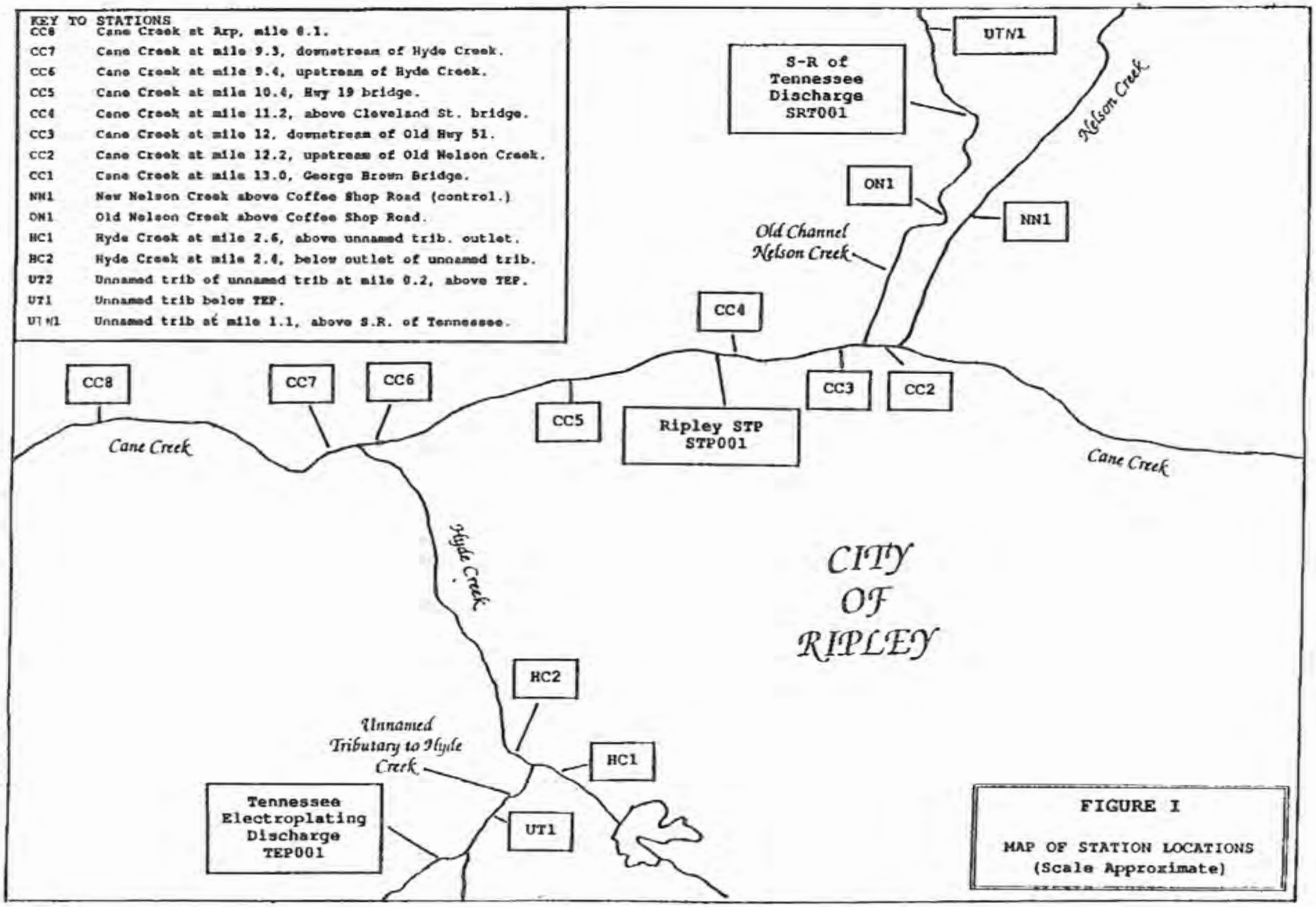


**FIGURE I**

MAP OF STATION LOCATIONS (Scale Approximate) marginal biological quality. The tolerant OAC was 63% while the diversity was 1.9 and the evenness was 0.5. Sampling block conductivity ranged from 188 at umhos/cm3 to 2030 umhos/cm3 at UTH1. Diversity at UTH1 was 2.1 with an evenness of 0.6. The appearance of the station and the assemblage of organisms present at UTH1 contradicted the optimistic diversity, as did the OAC of 99%. The diversity at UTH1 was reflective of equally high numbers of a few tolerant taxa. The OAC% was better at HC2 with 78% in August and 95% in September, but was still inferior to NN1, CC2, and HC1. R2 values comparing conductivity to OAC% were notable. August sampling resulted in an R2 of 0.8 and in September collections R2 equaled 0.9. Diversity values were not reflective of macroinvertebrate assemblage quality at these sites.

This study documented significant biological impacts to Cane Creek and its tributaries. The OAC percent of total sample by numbers proved to be highly correlated with instream waste as determined by instream conductivities.

These data along with chemical data and assimilative capacity modeling data have resulted in S-R and TEP agreeing with the State on lower NPDES permit limits.

# IN-SITU SEDIMENT OXYGEN DEMAND (SOD) MEASUREMENTS CONDUCTED IN THE BIG SANDY RIVER AND WEST SANDY EMBAYMENT OF KENTUCKY LAKE, HENRY COUNTY, TENNESSEE

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## INTRODUCTION

On September 19-21, 1989, State Division of Water Pollution Control (WPC) personnel conducted in-situ sediment oxygen demand (SOD) measurements in the Big Sandy River and the West Sandy Embayment of Kentucky Lake. The purpose of conducting these measurements was to 1) engage in an initial reconnaissance of SOD rates in portions of Kentucky Lake where low dissolved oxygen levels had previously been reported, 2) perform a trial run with WPC-SOD chambers and equipment in a deep water reservoir, and 3) evaluate the potential for using SOD measurements to assess and model the impacts of nonpoint source (NPS) runoff in lake embayments.

## DESCRIPTION OF SAMPLING STATIONS

The first series of SOD measurements was recorded September 19, 1989, on the Big Sandy River, mile 11 (BSRM 11) approximately 9.1 meters southwest of a TWRA fish habitat buoy (Figure 1). The sonar calibrated depth at this station averaged 4.1 meters. Approximate distance from the west bank of BSRM 11 was 914 meters.

The remaining SOD measurements were recorded in the West Sandy Embayment of Kentucky Lake. West Sandy Station 1 (WS STA 1) SOD measurements were recorded September 20, 1989, at the center of the mouth of West Sandy Creek, mile 0, at its confluence with the Big Sandy River, approximately 183 meters north of a navigational day marker (range finder distance, Figure 1). Range finder distance to a conspicuous tree 320o north of this station was 85 meters. The depth at this station averaged 4.7 meters. West Sandy Station 2 (WS STA 2) SOD measurements were also recorded September 20, 1989, near the north shore of West Sandy Creek, mile 1.5, approximately 250o, 228 meters northwest of West Sandy Landing (range finder distance, Figure 1). The depth at this station was 4.6 meters. West Sandy Station 3 (WS STA 3) SOD measurements were recorded September 21, 1989, at the center of West Sandy Creek, mile 2.5, 180o northeast of the Pleasant View Dock. The depth at this station was 3.1 meters.

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[1]Tennessee Department of Health and Environment, Division of Water Pollution Control, Nashville, Tennessee 37247.

## MATERIALS AND METHODS

Temperature and dissolved oxygen (DO) were measured at each station using YSI Model 50 meters. Meters were calibrated according to the manufacturer's instructions. An in-situ method for measuring SOD was employed as suggested by Murphy and Hicks [1]:

> In general terms, an in-situ SOD measurement involves isolating a known volume of water and area of sediment under an opaque (stainless steel) chamber placed on the bottom. The dissolved oxygen (DO) concentration in the chamber water is monitored until sufficient time has elapsed to establish a measurable rate of change in DO concentration. The SOD rate is then calculated using the following equation:
>
> $$SOD = 1.44 \frac{V}{A} (b_1 - b_2)$$
>
> where SOD is the sediment oxygen demand rate, in $g/m^2 * day$; $b_1$ is the rate of change of DO concentration inside the SOD chamber, in $mg/L * min.$; $b_2$ is the rate of change of DO concentration inside the "blank" chamber, in $mg/L * min.$ ($b_2$ is, in effect, the water column respiration rate in the chamber); V is the volume of the chamber, in liters (excluding the chamber core) = 64.86 L; A is the area of the chamber, in square meters = 0.27 $m^2$; and 1.44 is the constant converting $mg/L * min.$ to $g/m^2 * day$. Values for $b_1$ and $b_2$ may be determined graphically or from a linear regression analysis, where $b_1$ and $b_2$ are the slopes of the curves obtained by plotting SOD chamber and blank chamber DO concentrations versus time.

SOD chambers were deployed by TWRA and TDHE personnel using SCUBA. DO readings were recorded at each station at 15 minute intervals for at least 2 hours or until an oxygen depletion of 1.0 mg/L was observed. A control SOD chamber was set at mid- depth in the water column at each station, and two replicate SOD chambers were set at the sediment-water interface at each station. Standard size (500 mL) light and dark biological oxygen demand (BOD) bottles were also deployed with the SOD control chamber at each station as a means of comparing water column respiration rates. Light and dark BOD bottle DO concentrations (mg/L) were determined according to the Winkler Method as suggested by Wetzel and Likens [2].

## RESULTS AND DISCUSSION

Table 1 is a listing of the average rates of oxygen depletion ($\bar{b_1}$ as mg/L/min), adjusted averages ($b_1$ -$b_2$), replicate SOD rates, and other information calculated from data collected in the field from the four sampling stations. Figure 2 shows linear plots of the DO (mg/L) versus time (minutes) for the control SOD chamber plus the two replicate chambers at each of the four sampling stations. During this study, none of the sampling stations were thermally stratified.

Several equipment problems were encountered in the field during this study. The dark BOD bottle DO value of 12.61 mg/L versus the light BOD bottle DO value of

10.18 mg/L at the BSRM 11 station was due either to pipetting errors encountered during titration procedures or to unintentional air entrapment in the dark BOD bottle. Improved quality control procedures were developed for subsequent stations and more reasonable values were measured. The control chamber (b2) at BSRM 11 was unsealed due to diver unfamiliarity with equipment, and may account for the higher water column oxygen depletion rate observed at this station (Table 1).

The higher rate of oxygen depletion observed may also be the result of immigration of respiring fauna into the unsealed chamber. A pump failure was detected in the replicate 2 SOD chamber from the BSRM 11 station, and probably accounts for the disparity among the replicate DO depletion rates at this station.

The DO ranges in the b1 SOD chambers at WS STA 1 and WS STA 2 were substantially lower than the DO ranges at the other stations (ranges: 4.9-3.0, and 5.1-1.9 mg/L respectively; Table 1). These DO differences may be partly due to organic inflows from the Big Sandy River. The differences in DO measurements between the replicate SOD chambers from stations WS STA 1 and WS STA 2 may also be partly due to an air bubble that was discovered in the membrane of the replicate 1 DO meter during the WS STA 1 measurements. This problem was corrected for the WS STA 3 sampling.

## CONCLUSIONS

The high oxygen depletion rates of West Sandy warrant future investigation on a larger scale within the Big Sandy River Drainage (i.e., more stations, more replicates). A study with EPA and WPC is tentatively planned for the fall of 1990. These high SOD rates must be tempered with the understanding that one of the primary objectives of this study was to test the equipment and methods involved in an in-situ SOD experiment. As such, the measurements in this report should be viewed only as a first step in the type of SOD analyses that will be required to model the impacts of NPS runoff on the SOD in this system.

Some general observations from the reported replicate SOD rates in Table 1 are 1) SOD rates increased slightly from the mouth of West Sandy Creek southwest towards the TVA pumping station at West Sandy Dike, 2) SOD rates reported were within the range of other organically enriched systems within the state where in-situ SOD rates have been reported (see Appendix A & B; in June 1987 the mean SOD rate for the Duck River Embayment = 0.154 g $O_2/m^2/hr$ [EPA]; in May 1989 the mean SOD rates for three replicate Reelfoot Lake stations were 0.057, 0.099, and 0.104 g $O_2/m^2/hr$, respectively [EPA]; whereas the mean SOD rate for West Sandy Embayment, September 1989 = 0.097 g $O_2/m^2/hr$ [WPC]), and 3) The mean SOD rates reported for West Sandy (0.097 g $O_2/m^2/hr$,) are higher than mean SOD rates reported in June 1987 for Kentucky Lake (TN River Mile 102, near New Johnsonville) and the Harpeth River (see Appendix, 0.081 and 0.089 g $O_2/m^2/hr$, respectively).

## REFERENCES

Murphy, P. J., and D. B. Hicks, 1986. In-Situ Method for Measuring Sediment Oxygen Demand. In: K. J. Hatcher, (Editor), Sediment Oxygen Demand: Processes, Modeling and Measurement, Institute of Natural Resources, University of Georgia, Athens, GA.

Murphy, P. J., 1988. Memorandum Listing the Results of the Sediment Oxygen Demand Measurements Conducted in Kentucky Lake, Duck River Embayment of Kentucky Lake, and the Harpeth River During June 1987 in Tennessee. U.S. EPA, Region IV, Environmental Services Division, Athens, GA.

Howard, H. S., 1989. Memorandum Listing the Results of the Sediment Oxygen Demand Measurements Conducted in Reelfoot Lake on May 22-26, 1989 in Tennessee. U.S. EPA, Region IV, Environmental Services Division, Athens, GA.

Wetzel, R. G., and G. E. Likens, 1979. Limnological Analyses. W.B. Saunders Company, Philadelphia, PA, pp. 71-81.

Table 1. Big Sandy River and West Sandy Embayment, Kentucky Lake SOD Rates, September 1989.

| Station | $\bar{b}_1$-Avg. Rate of $O_2$ Depletion (mg/L/min) | $\bar{b}_1-b_2$ Adjusted Average (mg/L/min) | Replicate SOD ($g\ O_2/m^2/hr$) | Mean SOD West Sandy ($g\ O_2/m^2/hr$) | $b_2$-Water Column Depletion Rate (mg/L/min) | Dark BOD Bottle (mg/L/min) | DO Range (mg/L) |
|---|---|---|---|---|---|---|---|
| BSRM 11 | 0.01129 | 0.00770 | 0.11341 | | 0.00359* | ** | 8.5-5.4 |
| WS STA 1 | 0.00641 | 0.00641 | 0.09016 | 0.09716 | 0.00000 | 0.00581 | 4.9-3.0 |
| WS STA 2 | 0.00756 | 0.00672 | 0.09645 | | 0.00084 | 0.00185 | 5.1-1.9 |
| WS STA 3 | 0.00972 | 0.00711 | 0.10487 | | *0.00261 | 0.00340 | 5.5-3.8 |

*The control SOD chamber was not properly sealed at this station, and may account for the higher water column depletion rate at this station.

**At BSRM 11, the dark BOD bottle value was higher than the light BOD bottle, due to the improper technique discussed in the text of this report.

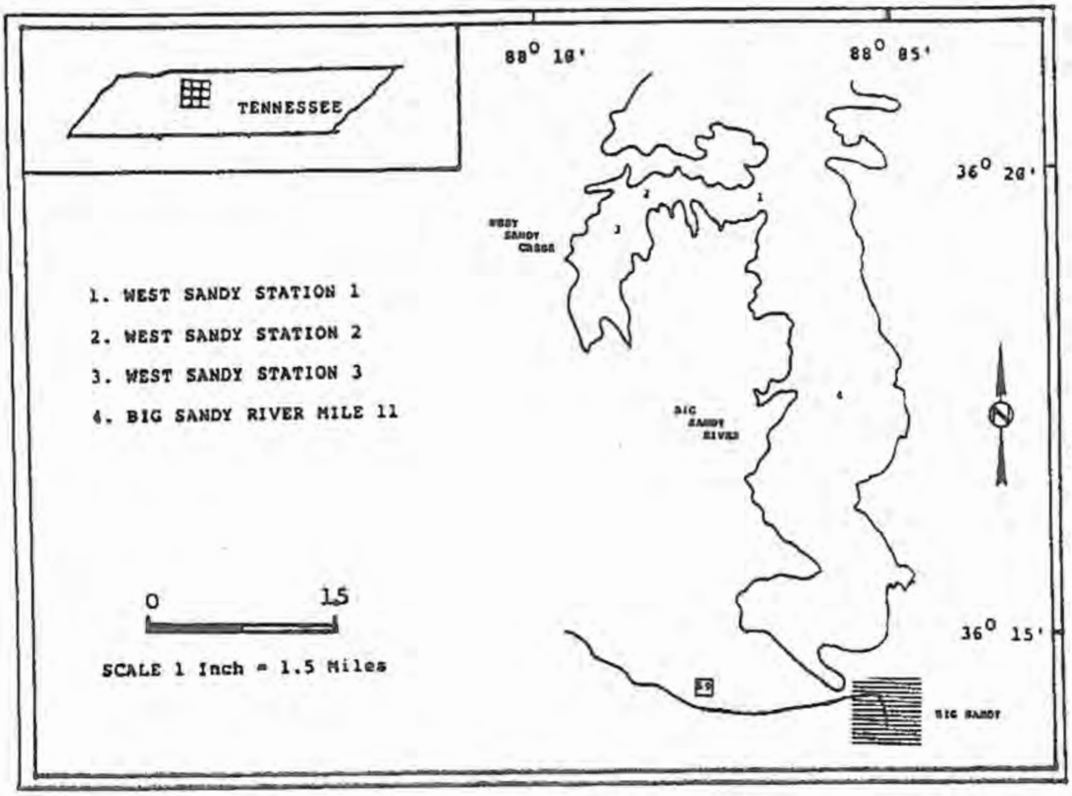


Figure 1. Map of Kentucky Lake, Big Sandy River and West Sandy Embayment, showing sampling station locations for sediment oxygen demand (SOD) measurements, 19-21 September, 1989.

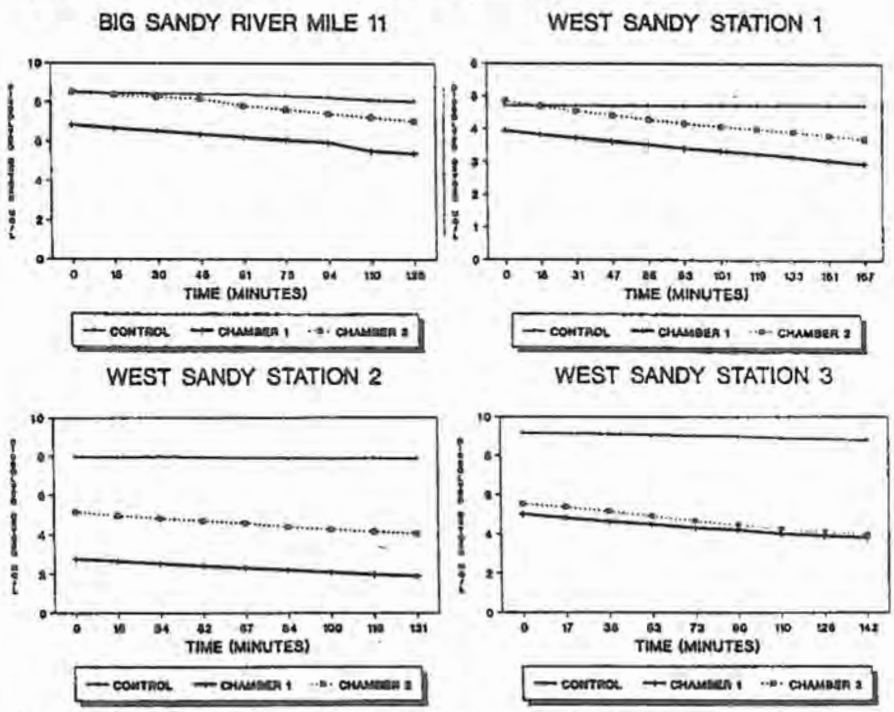


Figure 2. Linear plots of the dissolved oxygen (DO) [mg/L] versus time [minutes] for the control sediment oxygen demand (SOD) chamber plus the two replicate SOD chambers at each of the four sampling stations, 19-21 September, 1989.

## APPENDIX A

Kentucky Lake, Duck River Embayment, and Harpeth River SOD Rates (Replicates and Means) June 1987

| Station | Rep | Avg. Rate of (mg/L/min) | Adjusted Average (mg/L/min) | Replicate SOD (gO /m /hr) | Mean SOD (g O /m /hr) | Water Column R (mg/L/min) | DO Range (mg/L) Initial/Final | Average Water Temp. C |
|---|---|---|---|---|---|---|---|---|
| Kentucky Lake Sta. 1 | 1 | .00529 | .00453 | .065 | .081 | .00076 | 8.5-6.8 | 28.0 |
| | 2 | .00928 | .00852 | .123 | | | | |
| | 4 | .00539 | .00463 | .067 | | | | |
| | 5 | .00547 | .00471 | .068 | | | | |
| Duck River Sta. 1 | 1 | .01215 | .01056 | .152 | .154 | .00159 | 8.8-6.0 | 26.0 |
| | 2 | .01182 | .01023 | .147 | | | | |
| | 3 | .01282 | .01123 | .162 | | | | |
| | 4 | .01074 | .00915 | .131 | | | | |
| | 5 | .01382 | .01223 | .176 | | | | |
| Harpeth River Sta. 1 Downstream of STP | 1 | .00563 | .00434 | .063 | .089 | .00129 | 6.3-5.4 | 24.0 |
| | 4 | .00798 | .00669 | .096 | | | | |
| | 5 | .00871 | .00742 | .107 | | | | |
| Harpeth River Sta. 2 Upstream of STP | 3 | .00778 | .00668 | .096 | .089 | .00110 | 7.3-6.8 | 23.5 |
| | 4 | .00717 | .00607 | .087 | | | | |
| | 5 | .00625 | .00518 | .075 | | | | |

## APPENDIX B

Average Rate of Change, SOD Rates ($g/O_2/m^2/hr$) and Water Column Respiration
Reelfoot Lake, TN, May 1989

| Station | Rep | Avg. Rate of[1] Change (mg/L/min) | Adjusted Average* (mg/L/min) | Replicate SOD[2] ($g\ O_2/m^2/hr$)) | Mean SOD[3] ($g\ O_2/m^2/hr$) | Water Column[4] R (mg/L/min) | DO Range (mg/L) Initial/Final | Average Water Temp $^oC$ |
|---|---|---|---|---|---|---|---|---|
| 1 | 3 | .00944 | .00449 | .06466 | .05659 | .00495 | 8.2/6.4 | 24.5 |
| 1 | 4 | .00832 | .00337 | .04853 | | | | |
| Note: 2 replicates aborted because of either pump failure or probe problems (Reps 1 and 5) | | | | | | | | |
| 4 | 1 | .01048 | .00693 | .09979 | .09907 | .00355 | 7.75/5.45 | 25.0 |
| 4 | 4 | .00935 | .00580 | .08352 | | | | |
| 4 | 5 | .01146 | .00791 | .11390 | | | | |
| Note: Replicate 3 not used due to erratic readings (possible leak) | | | | | | | | |
| 5 | 1 | .01020 | .00757 | .10901 | .10386 | .00263 | 9.0/7.6 | 22.0 |
| 5 | 3 | .00876 | .00613 | .08827 | | | | |
| 5 | 4 | .01178 | .00915 | .13176 | | | | |
| 5 | 5 | .00863 | .00600 | .08640 | | | | |

* = Slope for Rep. minus water column R
1 = Slope for Rep.
2 = Adjusted avg. x 14.4
3 = Avg. of Reps. SOD
4 = Avg. of blank chambers (slope)

# BOONE RESERVOIR NON-POINT SOURCE WATER POLLUTION CITIZEN DEMONSTRATION PROJECT

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## INTRODUCTION

The Tennessee Environmental Council, U.S. Department of Agriculture Soil Conservation Service in Sullivan County and the Boone Lake Property Owners Association and Friends are employing volunteer water quality monitors in the Boone Reservoir watershed to help citizens understand and, ultimately, control non-point source water pollution (NPS). Boone Reservoir lies in the northeastern corner of Tennessee within Carter, Sullivan and Washington counties.

The primary purpose of this project is to educate citizens in the Tri-Cities area about NPS water pollution. There are five components to this project. These include: 1) monitoring water quality of streams that flow into Boone Reservoir; 2) identifying possible NPS of water quality degradation; 3) educating landowners and other Tri-Cities citizens about NPS; 4) helping landowners initiate cost-effective methods for controlling NPS and 5) helping landowners interface with government agencies which provide both cost share and technical assistance.

## NEED/PROBLEM

Of Tennessee's 19,124 stream miles only an estimated 42 percent have been monitored or assessed. Of the state's 90 official water quality monitoring stations, many are located around industrial discharges. Once established, volunteer monitoring teams will be able to generate continuous benthic macroinvertebrate data on streams that, until now, have been ignored.

Boone Reservoir was created for flood control and power generation, however, additional uses have developed as the surrounding area's population has grown. Three cities now draw their water supplies from the reservoir, and it also supports a large sport fishery. Because of its proximity to the Tri-Cities area, Boone Reservoir has become one of the U.S. Tennessee Valley Authority (TVA) system's most popular recreational lakes (2).

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Boone Reservoir is one of the most eutrophic in the TVA system. It has a history of bacteriological contamination associated with runoff. According to the State Department of Health and Environment's 1988 305b Water Quality Status report 4400 acres of the lake does not fully meet designated uses (Denton 1988). This is partially, if not largely, due to NPS pollution.

Livestock operations are a major source of NPS pollution within the 669 square mile watershed. TVA analysis of high altitude aerial photography identified 401 cattle operations, 39 dairies, 23 poultry and 2 swine operations in the watershed (Lewis et.al. 1985).

Many of these operations are contributing pollutants to the streams that feed the reservoir, however, field personnel must visit each site to determine the extent of the NPS pollution. There are three Soil Conservation Service (SCS) District Conservationist within the watershed that are candidates for initiating the field work. This would appear to be inadequate given the size of the watershed. Therefore, we have enlisted citizens to use simplified biological (benthic macroinvertebrates) monitoring techniques to identify those streams that are degraded by NPS water pollution.

## METHODS AND MATERIALS

Upon initiation in September of 1989, there were no known standard methods for the collection by volunteers of benthic macroinvertebrates. The volunteers were using a variety of different monitoring methods to collect samples. These methods ranged from homemade surber samplers to large eight by three foot kick-nets. In late spring of 1990 the Izaak Walton League (IWL) methods were adopted and introduced to part of the group. By fall of 1990 all volunteers should be using the IWL method.

The materials required for stream monitors to begin sampling is quite simple. These include: 1) A kick-net or an old window screen three by three foot (without holes and with supporting poles); 2) tweezers; 3) a six by ten inch white or clear plastic container; 4) several jars for sorting and/or collecting; and 5) a magnifying glass or hand lens. Additionally, an IWL "bug card" and stream survey forms are essential.

The collection of the insects is also very simple and by design very thorough and reliable. Once on site volunteers select a riffle area that is approximately two to twelve inches in depth and has cobble size (two to ten inches) rocks.

Entering downstream of the collection area, the volunteer places the kick-net at the downstream edge of the site making certain that the weighted bottom of the net comes in contact with the stream bottom. Next, the stream bed is disturbed with foot action three feet upstream and the width of the three foot net. The volunteer then rubs their hands over all rocks in the three by three foot square area. Then foot action and hands are used to dislodge the more stubborn of insects. Finally, for not more than 60 seconds foot action is used to kick toward the net. The net is then removed using a forward scooping motion. The volunteer is careful not to let the sample go over the top of the net. Once this is done, it is a matter of identifying the sample to genus, counting the numbers of each organism and recording the data on the stream survey form. When this is done it is simple to determine the water quality in the stream. Additionally, to ensure accuracy, it is recommended that three samples be taken at the site.

## RESULTS AND DISCUSSION

Most of the data that has been collected to date has little, if any scientific value given the lack of consistent collection methods, however, it does have educational value. The educational value is realized by the volunteers, landowners, and some citizens.

By far, the greatest educational value this project has is to the volunteers. It teaches stream ecology and about NPS pollution that degrades stream ecology. With 80 percent of Tennessee's water pollution coming from NPS, yet with very few lay people even knowing what NPS is, this educational value is worthwhile. This type of hands on education may serve to prevent NPS or other pollution problems from occurring in the future.

Once polluted streams are identified and a NPS is located, education of the landowner is initiated. This is accomplished using data collected from the stream and by trying to relate that data to the landowners actions such as allowing livestock in the stream. Asking the landowner to participate in the sampling or having them present on stream side during collection makes landowners feel more a part of the process and also gives volunteers a chance to better understand the landowner and his motivations for using certain methods. Most farmers realize that their actions affect water quality. However, there ability to change methods is more times than not, tied to financial considerations. In some situations, volunteers are willing to help landowners overcome labor costs by assisting in fencing creeks off from livestock. However, other larger NPS control projects are very limited without sufficient cost-share money. At present $3500 per year is the maximum any one landowner can receive from the US Department of Agriculture SCS, while some dairy or feedlot operations require as much as $15,000 to $20,000 for animal waste management systems. Because of this we have initiated an Agriculture Conservation Program (ACP) grant request.

It is also important to point out to farmers that reducing NPS not only helps keep streams clean but, also benefits the farmer. Reducing soil loss helps keep valuable nutrients on the land. Using chemical application best management practices (BMP) can reduce the amount of chemical used and make those chemicals used more effective, saving the farmer money. By storing animal waste through the year the farmer has his own source of fertilizer, again saving money. Each of these points can be used to help the farmer understand that there are on-site benefits as well as off-site benefits to controling NPS. If the landowner is receptive to our project, we will help him get in contact with the appropriate agency to provide best management practices for his industry and find cost effective ways to deal with his NPS problem.

Seventeen stream monitoring groups have been recruited and a small contingent of citizens have been giving slide presentations to civic, scout and school groups. In total, the NPS message has reached almost 1000 people via slide presentations.

Two articles have been published in local papers and the project has been featured on two television news programs.

The value of the educational aspects of this project may not be realized for some time, however, education, we believe will lead to prevention of future NPS problems. Finally, with the state's non-regulatory approach education would appear to be one of the more powerful tools in controlling NPS.

## REFERENCES

Denton, Greg, 1988. The Status of Water Quality in Tennessee. 305b Rpt. Tennessee Department of Health and Environment, Office of Water Management. Nashville, Tennessee.

Lewis, Alicia R., et al, 1985. Boone Reservoir Water Quality Assessment. pp. 7 and 46. U.S.T.V.A. Office of Natural Resources and Community Development, Water Quality Branch. Chattanooga, Tennessee.

# RESERVOIR INTERPOOL PLANT HABITAT DYNAMICS II

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## INTRODUCTION

Reservoir interpool zones delimit ecologically new, seasonally exposed, wetland habitats between winter and summer pool level shores. Tennessee Valley Authority (TVA) documents list 26 impoundments with summer shorelines of 9200 km. upstream of Chickamauga Dam in the Tennessee River watershed. Drawdown, interpool zones comprise a nominal 42,500 ha. within these impoundments. They commonly exist as definable entities for several consecutive months of every year, often from October or earlier to April. Conspicuous alluvial flats have led to their encompassing designation as mudflat ecosystems. Distinctive phenological biotypes have evolved on these repetitively unstable transported and residual, truncated substrates. The mudflats (and the winter constant reservoir pools) are large when compared to any regional pre-impoundment counterparts, lentic or wetland.

In contrast to the material instability of the substrates is the winter thermal stability of the sediments. A model has been developed to express the brumal mudflat sediment temperatures. The associated winter pool is characterized as a stabilizing, hyperthermal heat sink.

## OBJECTIVES, HYPOTHESES AND OBSERVATIONS

Floral collections, non point source pollutant (NPSP) dynamics and recognition of overt bio-physical ecological interactions have had predominant importance. As the study develops, five objectives have been treated as hypotheses and are being considered:

1. Subaerial, winter or brumal exposure must be of several weeks duration for successful plant maturation and therefore classification as wetland mudflat ecosystem.

2. Biotal behavior and development varies from traditional, stable, terrestrial strategies within previously undescribed amplitudes.

3. The presence or absence of successfully adapted higher plants reflects physical and chemical mudflat dynamics, identifying and modifying NPSP and localized perturbations.

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4. Compensatory life history strategies and appropriate physical and microenvironmental condition adjustments account for the development of successful ecosystem bio-components.

5. Mudflat manifestations and areal extents increase with hydraulically induced summer shore retreat and hydrological NPSP silt accumulations.

Studies of Southern Appalachian mudflats are not extensive; however, TVA and USA Corps of Engineers sponsored analyses are useful. Regional air temperatures are a matter of record, but do not directly lend to plant response phenomena on the mudflat substrates. We have not found shoreline eco-chemistry or temperature studies. Mudflat sediments show no pedogenetic development. Hydraulic alterations of mudflat substrates and plant propagule source banks have been examined empirically.

Floral collections have not revealed any taxa unexpected for the Ridge and Valley Province of the Southern Appalachians, although the mudflat presents a new habitat record for most of the identified species. Macrofauna, including avifauna and venturesome terrestrial vertebrates, have been noted. Microflora and microfauna have not yet been characterized.

This report focuses on Watts Bar Reservoir (WB) which was closed in January,1942 (35° 40′ N, 84° 50′ W). For 30 consecutive months, beginning in January 1988, temperature and pH have been recorded weekly (instantaneous measures), along with plant phenological development. Aerial biomass clip plots have been taken subjectively to determine potential productivity.

A sediment heat transfer model for reservoir mudflats which benefit from winter pool heat has been developed using literature from the disciplines of heat transfer physics and of microclimatology. Benchmark tests were derived from studies of saturated soil temperatures and from the weekly WB reservoir/shore and flats temperatures. Lotus 123 is used as the programming tool.

As a part of this study erosive shore retreat was measured on consecutive editions of topographic maps. Kinetic wave force generated by boat passage was calculated, and relationships can be postulated. Observations of bottom silt agitation by boat propeller and passage are only qualitative, as is the disturbance of the sediment by bottom dwelling macrofauna. Entrapment of alluvium by inundated mudflat plant communities is visually evident but unmeasured in this study.

## QUANTIFICATIONS AND DISCUSSIONS TO DATE

WB has a shoreline of 1240 km at 226m., msl, and a mudflat of 2550 ha. (within a 15780 ha. pool, TVA, 1980) which has now existed for 48 drawdown year periods. The mudflat is exposed from late October to late April most years. The zone has an averaged slope of 09% with a 20m width.

Boat waves, the major cause of WB shore erosion, impact shores normally with a force of 0.96 kw/m, for 0.50 meter waves in one meter water depth. Consequent topographic maps show a localized summer shoreline retreat of two meters a season. An overall one meter summer shoreline retreat would geometrically add 113 ha. a year to the mudflat. TVA projections of sediment accumulations for WB (1980) are 1.64 x $10^6 m^3$ a year. Bank collapse plus sediment accumulations account for the enlargement of the mudflats of WB.

We have no knowledge of the channel pool of WB freezing over. During the two and one half years of this study, the minimum air temperature (shore, one meter above surface) was -18°C, while the at that time water surface (bulb immersed) was 5°C. Ice forms occasionally on the surface of the inshore sediment mudflats, and in shallow, isolated coves, but where the ice remains in contact with continuous sediment, it has not been seen to persist throughout the day.

The sloping, truncated historic soils may remain surface frozen throughout the day. This is often due to less capillary retention and the greater vertical distance to the isothermal stratum in contact with the winter pool heat sink. The alluvial sediments with high moisture retention are usually incorporated within a few (5 to 15) cm above the continuous saturated zone of dramatic heat sink influence.

Channel pH tends to be basic, summer; neutral winter. Mudflat water edge tends to be neutral to basic when channel water exchange is occurring (current, wind, boat wave) and acidic when calm and subject to vegetated shoreline runoff. Local precipitation makes little difference otherwise. In shallow coves, on warm spring days, mudflat plants occasionally may be subjected to pH in standing micropools varying between 6.5 and 10.0 as solar warming and algal respiration raise basicity. Most mudflat community associated pH measures 6.5 to 7.5 depending upon the current weather conditions and the amount of mixing.

Textures of abutting, historic soils are those of the edaphic series common to higher flood plains and first terraces. The exposed profiles (lower B, C) tend to be high in once reduced clays with lenses of alluvial sand and scattered, rounded, cherty gravel. The transported sediments tend to be homogeneous, fine textured silts, high in clay and low in organic matter. They may overlie reduced ochric clays and are usually above a conspicuous saturated gray zone, anoxic and demarked by the horizontal development of otherwise geotrophic plant roots and the incorporation of scarcely decomposed allochthonous plant litter.

The collection of plants considered successful on the WB drawdown are mostly from seed banks and total less than 30 species with some exhibiting nearly monotypic dominance. Mudflat plants have generally shifted their seasonal terrestrial phenology to fall germination - late winter maturation and in this arrested sere behave as periodic initiators in tune to anthropogenic drawdown exposure. Subjective clip plot placement has yielded up to 200g per $m^2$ constant weight aerial plant biomass. Many areas are barren of higher plants for reasons yet to be determined.

An initial autumn depression of the sediment temperatures was not followed by a continuing drop, but showed a temperature plateau in the mudflat plant root zone that usually registered well above freezing. In the absence of exposed microelevations, subsurface sediment temperatures were very close to the main pool temperatures.

The pool-continuous, water saturated, anoxic gray zone is empirically isothermal, and as a stabilizing heat sink provides heat to the mudflat sediments during cold periods. Taking atmospheric radiative and convective heat transfer into account, and including conductive heat flow from the gray zone, a model was developed to allow calculations of effective temperature across the sediment mantled mudflats.

A time dependent finite difference heat conduction approach is used to describe the desired profile. The model is solved by computer for various ambient conditions. The sediment is considered homogeneous and inserting approximations for small, finite changes in time and distance yields an equation solved at each node in the computer model. The heat equation is modified to include net absorbed surface radiation and surface convective energy transfer. For each solution, constant initial temperatures for both average air and the gray zone are assumed. Results are shown at four hour intervals (Figure 1).

As the air temperature is decreased the amount of temperature reduction of the ground surface below the air is reduced. As the surface temperature is reduced there is more heat transferred from the gray zone since heat energy transfer is consequential to temperature difference. During very cold air periods the sediment surface temperature will be much warmer than the air a few centimeters above, and the root zone temperatures will not drop to frost. If the gray zone is closer to the surface, the profile in Figure 1 is shifted to higher temperatures. This is often the instance in sediment flats perched on water retaining gray zones only centimeters above winter pool elevation.

## CONCLUSIONS

A distinct, periodic wetland ecosystem is evolving and enlarging across the ecologically new brumal, interpool zones of substantial duration in upper Tennessee Valley impoundments. These newly characteristic mudflats have assumed a substantial consideration in the hydrological, biological and ecological functioning of the upper Tennessee River Basin.

Acknowledgments: Other contributors to this paper include: R. Kramel, V. Spicer, J. Rodgers, H. Rial- Meador, S. Lawson-South, T. Bailey, J. Duff, B.E. Wofford (UTK), K.W. Wittingham (USACE), D.H. Webb and others (TVA), and R. Meinert of WB.

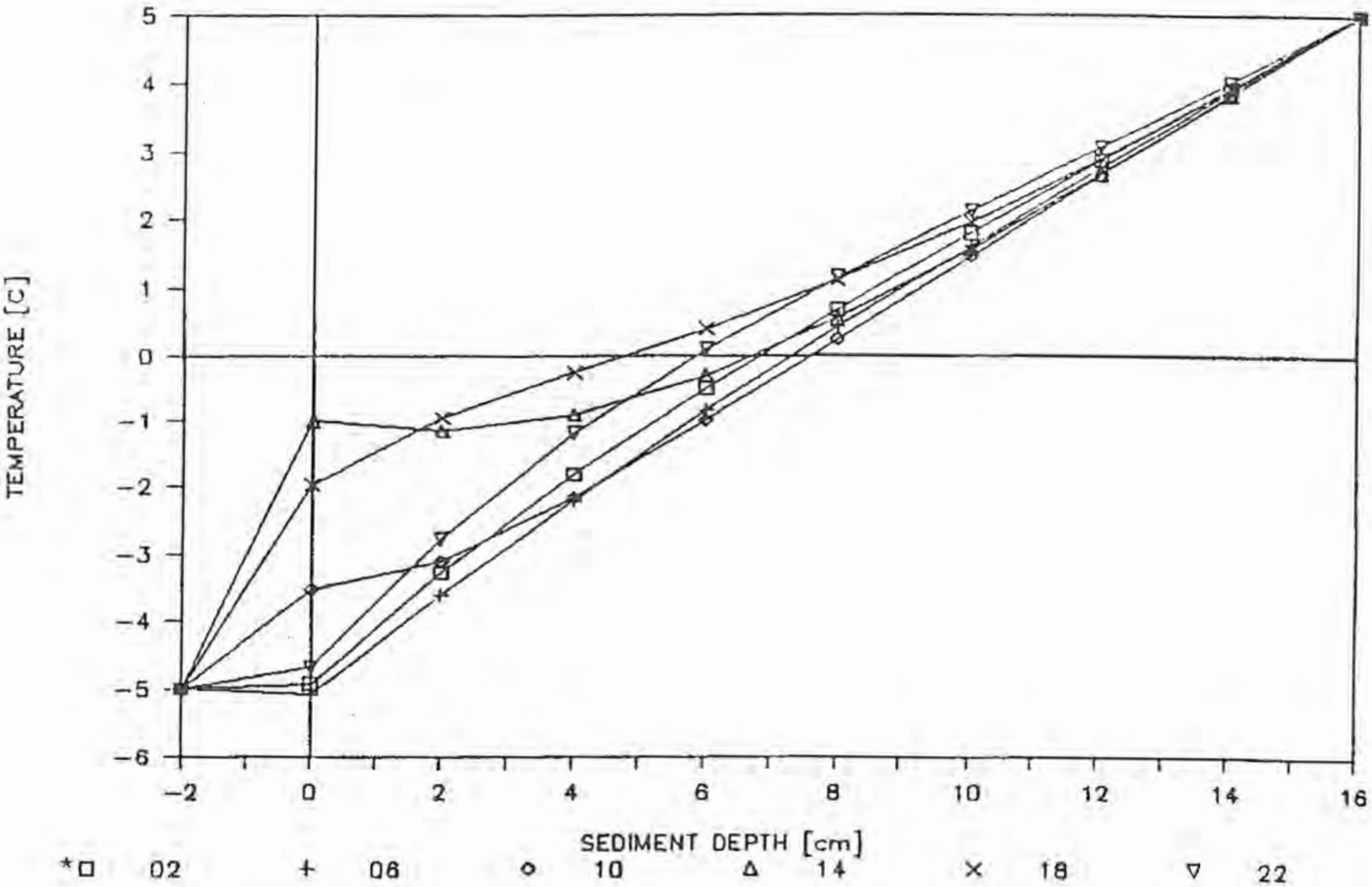


Figure 1. Sediment temperature profiles. Symbols indicate 2400 hour times* for an average air temperature of -5°C and grayzone, heat sink temperature of 5°C for a mudflat sediment thickness of 16 cm.

# THE FEDERAL COOPERATIVE PROGRAM OF THE TENNESSEE DISTRICT OF THE U.S. GEOLOGICAL SURVEY, WATER RESOURCES DIVISION

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The Federal Cooperative Program of the Tennessee District of the U.S. Geological Survey, Water Resources Division, represents a diverse program that addresses a multitude of water-resources problems and issues within Tennessee. During the 1989 and 1990 fiscal years, the Tennessee District conducted surface-water, ground-water, and water-quality monitoring activities, interpretive investigations, and research in cooperation with the following number and levels of agencies:

| | |
|---|---|
| 20 | Municipalities |
| 8 | Utility districts |
| 2 | Counties |
| 1 | River-basin authority |
| 8 | State agencies |
| 2 | Universities |

The following categories, and the number of cooperative activities of the Tennessee District in each category, reflect water-resources issues in Tennessee:

| | |
|---|---|
| 10 | Ground-water protection |
| 8 | Environmental effects of waste disposal |
| 7 | Surface-water supply |
| 7 | Ground-water availability |
| 6 | Floods and associated urban studies |
| 4 | Ground-water quality |
| 4 | Surface-water quality |
| 2 | Water use |
| 2 | Channelization research |
| 2 | Storm-water quality |
| 1 | Wetlands protection |
| 1 | Lake-level management |

[1]U.S. Geological Survey, A413 Federal Building, Nashville, Tennessee 37203 (615/736-5424).

# URANIUM SERIES DATES OF TENNESSEE CAVE DEPOSITS: PALEOCLIMATIC AND PALEOHYDROLOGICAL IMPLICATIONS

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Oak Ridge, Tennessee

## INTRODUCTION

Cedar Ridge Crystal Cave and Spring Cave, Gizzard Cove, are two caves formed in Mississippian limestones of the Cumberland Plateau, Marion County, Tennessee (Fig. 1). Field and laboratory studies were undertaken to both study the relationship between cave morphology, surrounding lithology, and structure, and to obtain ages of speleothems (secondary mineral cave deposits such as stalagmites and stalactites) using the 230-Th/234-U dating technique. The absolute age of the base of a speleothem marks the minimum oldest time that the passage within which the deposit grew was located in a vadose cave. Also, the continuous growth of a speleothem implies that the ground-water supply did not significantly change during that time period. Absolute ages of speleothems can record the chronology of the draining of an aquifer (Ford, 1989). Dates obtained from the growth axis of an 80-cm stalagmite from Cedar Ridge Crystal Cave are presented along with dates obtained from other speleothem fragments from both caves. Data from Tennessee are compared with dates obtained from other caves in the United States, Great Britain, and the Canadian Rocky Mountains. Paleoclimatic and paleohydrological implications are discussed.

## METHODS AND PROCEDURES

In the area around and in the two caves, information was obtained concerning karst features, relationships between lithology and speleogenesis, and structure. Structural information, including joint-type and orientation, and strike and dip of bedding, were obtained inside and outside the caves. The caves were surveyed using a compass, clinometer, and fiberglass tape. Speleothem fragments that had been broken naturally as well as an 80-cm stalagmite which was removed from its growth position were collected for dating. The samples were prepared using a modified method after Cowart (1974). In summary: samples were weighed, dissolved in dilute nitric acid and filtered if insoluble residue was obviously present. Carrier iron and 228-Th/232-U spike solution was added. The uranium and thorium was then co-precipitated with the iron. The flocculate was then isolated by centrifuge and washed with distilled water and dissolved in concentrated hydrochloric acid. The carrier iron was removed and uranium and thorium purified, first by solvent extraction and then by anion exchange columns. The oxide residues were then electroplated onto stainless steel planchets for alpha

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counting. Precision and accuracy were demonstrated, and reagent blanks were analyzed and a correction applied. Additionally, ages were corrected for detrital contamination (when the measured 230-Th/232-Th activity ratio was less than 20). Isotope activities were determined by alpha spectroscopy. Because concentrations and some yields were low, some count-times exceeded 25,000 minutes (17.4 days). Backgrounds were generally low and were monitored every two months. Data were entered into a BASIC computer program that was used to calculate: ages, isotopic data, associated 1 sigma errors, concentrations, chemical yields, 230-Th/234-U, 230-Th/232-Th, 234-U/238-U activity ratios and 230-Th/234-U ages.

## RESULTS

Jointing (sets at N 25 E and N 50 W) seems to have had the dominant influence on the direction of drainage in the area of Cedar Ridge and Gizzard Cove. The location of Cedar Ridge Crystal Cave on an isolated topographic "spur" could mean that the cave was once part of a larger system dissected by Battle Creek valley and Gizzard Cove. The present entrance of Cedar Ridge Crystal Cave is located at 679 ft above sea level. The elevation of Battle Creek adjacent to the cave is 630 ft above sea level. The entrances of both Spring Cave and Fiery Gizzard Creek are at an elevation of 660 ft. It is believed that Cedar Ridge Crystal Cave is an abandoned part of a cave system since isolated, either by entrenchment and subsequent phreatic bypassing of its stream, or downcutting of Gizzard Cove, forming initially as a karst valley.

Twenty-eight 230-Th/234-U ages from eight speleothem fragments from both caves and an 80-cm stalagmite from Cedar Ridge Crystal Cave were obtained that range from 89.8 to 1.3 Ky (Ky = thousand years). The mean of 16 ages along the growth axis of the 80-cm stalagmite is 69 Ky; alternatively, it could have started growing between 90 Ky and 70 Ky, terminating about at 70 Ky B.P. (Fig. 2). There is no significant difference in age within 1 sigma limits, but small-scale variations were observed in uranium isotopic data. These ages correspond to stages 1, 3, 4, and 5 in the marine 18-O/16-O record. In particular, the 80-cm stalagmite grew during stage 4 which corresponds to the next to last cold (glacial) period. These results clearly indicate that climatic conditions in Tennessee were favorable for speleothem growth and thus limit the severity of the climatic effects of stage 4 glaciation. These ages also suggest that the aquifer had drained below the elevation of Cedar Ridge prior to 90 Ky (104 Ky at 1 sigma limits). Growth of a speleothem in the lowest section of Spring Cave, Gizzard Cove, at the same level as the present valley floor, suggests that this section of cave was first drained at sometime prior to the last ice age (stage 2) prior to 38 Ky (51 Ky within 1 sigma limits). These data strongly suggest that ground-water recharge was not disrupted during the growth of the speleothem. The relationship between the initial 234-U/238-U activity ratio and the uranium concentration in the dated samples implies that its chemical and isotopic composition was relatively constant. Only one sample had a 234-U/238-U activity ratio near secular equilibrium.

## SUMMARY AND CONCLUSIONS

The passages directions within both caves are controlled by two dominant joint sets. The caves are parts of an epiphreatic cave dissected by the formation of Gizzard Cove, a karst valley. Absolute dates obtained on speleothems from both caves, including 16 ages from the growth axis of an 80-cm stalagmite range from 1.3 Ky to around 90 Ky. Ages along the growth axis of the 80-cm stalagmite are not significantly different (within 1 sigma limits). It is thought that this speleothem grew rapidly at two possible growth rates; more precise dating by mass spectrometry is being performed to confirm this. The respective absolute ages suggest that the relative date of aquifer lowering (phreatic bypassing) at Cedar Ridge Crystal Cave was at least sometime prior to about 90 Ky (101 Ky at extreme 1 sigma limits) and the relative date of phreatic bypassing to the level of the present valley floor was prior to 38 Ky (51 Ky at 1 sigma limits).

Speleothem growth is recorded in marine oxygen isotopic stages 1, 3, 4, and 5. No growth was recorded in stage 2 (the last Wisconsinian glacial maximum) albeit from a limited sample set. The 80-cm stalagmite grew during stage 4, described as a brief glacial stage where advance of the ice front was thought to be toward the Canadian Arctic instead of toward the south (Eyles and Westgate, 1987). Maximum glaciation occured 20 Ky B.P. (Kominz et al, 1979). Continuous growth of the 80-cm stalagmite suggests that recharge (vadose seepage) was not disrupted and the water budget in the soil and epikarst (the upper part of the bedrock) was probably not compromised.

## REFERENCES

Cowart, J. B., 1974. 234-U and 238-U in the Carrizo Sandstone Aquifer of South Texas. Ph.D. Thesis, Florida State University, Tallahassee, Florida.

Eyles, N., and J. A. Westgate, 1987. Restricted Regional Extent of the Laurentide Ice Sheet in the Great Lakes Basins During Early Wisconsinian Glaciation. Geology, p. 537-540.

Ford, D. C., 1989. Uranium Series Dating of the Draining of an Aquifer. Geological Society of America, Abstracts with Programs, vol. 21, no. 6.

Kominz, M. A., G. R. Heath, T.- L. Ku, and N. G. Pisias, 1979. Brunhes Time Scale and the Interpretation of Climatic Change: Earth and Planetary Science Letters, 45:394-410.

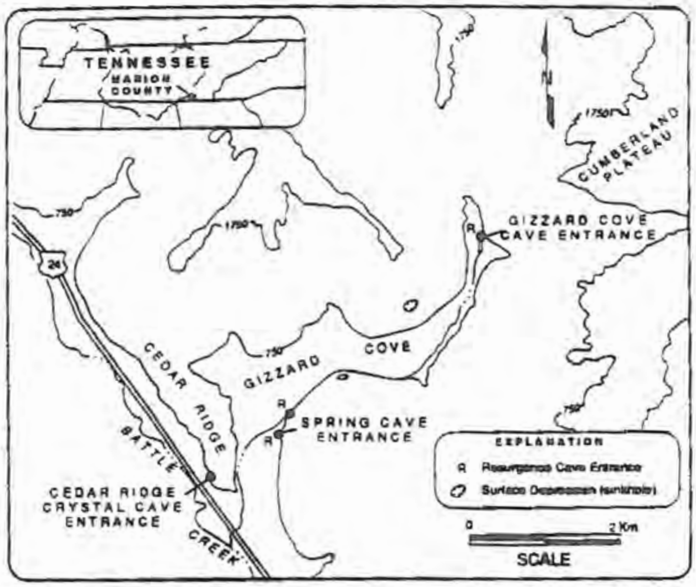


**Figure 1. Location of Cave Entrances and other Karst Features, Gizzard Cove, Marion County, Tennessee**

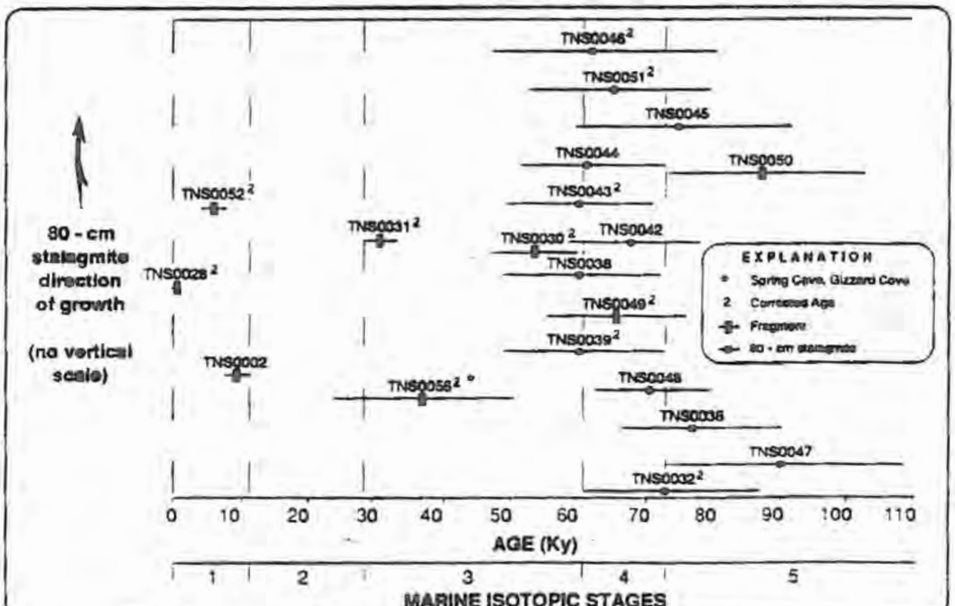


**Figure 2. Ages and 1 Sigma Errors from Individual Samples and 80 - cm stalagmite from Cedar Ridge Crystal Cave, and Spring Cave, Gizzard Cove, and Comparison with Marine Isotopic Record.**

**(Dates of Stage Boundaries from Kominz and et al, 1979)**

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# THE USE OF A GIS DATABASE TO AID IN THE ASSESSMENT OF IMPACTS ON SURFACE WATER QUALITY BY SURFACE COAL MINING IN THE BIG SOUTH FORK DRAINAGE AND TENNESSEE RIVER TRIBUTARIES

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Water resource managers and regulatory agencies have long been faced with the problems of assimilating the many-faceted and large volumes of data necessary to make sound management decisions. The relatively large geographic areas sometimes involved have also posed problems in bringing data together in the required format for recognition of factors such as landuse trends, drainage patterns, and topography. Geographic information systems (GIS), which have the ability to bring together large volumes of data in the combinations desired and for user specified geographic areas, can provide managers with a valuable tool to aid in this decision-making process.

A GIS database has been developed for the Big South Fork River drainage in Tennessee and Kentucky and for the Tennessee River tributaries from Chattanooga north to the Watts Bar area. This database incorporates physical features including surface water, roads, and landuse (derived from current satellite imagery) with water quality and biological data collected from both coal-impacted and non-coal-impacted sites within the two drainages. EPA river reach numbers have been assigned to streams, and watershed boundaries based on individual river reaches have been developed. Wetland areas have also been mapped and coded using the National Wetland Inventory classification system.

The GIS will be used to compare parameters of impacted and non-impacted sites, determine correlations between percentage of mined area within watershed and degree of surface water degradation (both biotically and abiotically), and to assess the effects of upstream mines on surface water in watershed which do not themselves contain mining.

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# AVAILABLE WATER IN FOREST SOILS OF CENTRAL AND EAST TENNESSEE

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Available water capacities of representative forest soils of the Highland Rim, Cumberland Plateau, and Ridge and Valley physiographic provinces were modelled using Thornwaite's method of calculation of the water balance. Rainfall records from average years and dry years were used when comparing soils with different water holding capacities. During an average year only the shallowest soils in areas with high potential evapotranspiration show soil water deficits. In day years most soils exhibit soil water deficits. Even during a series of dry years forest soils return to field capacity during the winter months. The results are compared to site indices for various tree species commonly found in the three physiographic provinces. Correlation with site index varies by species.

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# FIELD EXPERIMENTS TO DEFINE GROUNDWATER TRANSPORT OF CONTAMINANTS

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TVA is conducting large-scale field experiments to better define movement and dispersion of contaminants in groundwater. This will support groundwater protection by making it possible to design more effective waste storage facilities, improving monitoring systems for detecting groundwater contamination, and designing better mitigation programs where contamination has been detected. TVA began these field experiments in 1983 at the TVA Columbus Groundwater Research (CGR) test site located on Columbus Air Force Base in Mississippi. Tracers introduced into the groundwater to simulate contaminants are sampled at two-month intervals from a network of about 300 wells, each designed to collect samples from as many as 30 discrete depths. Tracer concentrations from this three-dimensional grid are used to define how a contaminant moves underground and is diluted with groundwater under realistic conditions. Research conducted or planned by TVA at CGR includes projects to protect groundwater from ash and other waste from fossil-fueled power plants; to analyze the movement of jet fuel and gasoline in groundwater; to improve bioreclamation of contaminated sites for the U.S. Air Force; to develop and demonstrate instrumentation and monitoring devices for groundwater investigations; and to define subsurface geochemistry and transport to pesticides and fertilizers.

This research has been supported by the Electric Power Research Institute, and the USAF Engineering and Services Center.

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# A CASE STUDY OF THE HOLMES ROAD SANITARY LANDFILL AS A NATURAL ATTENUATION SITE

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Attenuation describes the effectiveness of a landfill liner to filter pollutants from a leachate solution. The salient theme of attenuation then is to reduce the pollutant concentration to innocuous levels. The Holmes Road Sanitary Landfill in Memphis, Tennessee has a confining layer in the subsurface that qualifies as a liner with attenuation capabilities. Borings, performed by Hall, Blake and Associates (a geotechnical firm in Memphis), into the coastal plain lithology of this landfill reveal a 50 foot clay bed, laterally continuous across the site. Gamma-ray logs, provided by the U.S.G.S., confirm that this clay bed overlies a sand body, identified as the Memphis Sand aquifer by Parks of Graham and Parks (1985). Graham and Parks (1985) have identified the composition of the clay to be montmorillonite. The geophysical and lithologic data both indicate the clay layer caps the underlying Memphis Sand aquifer in the area of the landfill. Pump test data collected by Hall Blake and Associates (HBA) of Memphis, helped define the hydrogeology of the Memphis Sand aquifer as it relates to the clay cap (liner) in the landfill site. The HBA data in conjunction with U.S.G.S. pump test data indicate no vertical windows in the liner due to no anomalous decrease in pressure head across the site. Baty, as part of the in-house report prepared by HBA, modeled the potentiometric surface of the Memphis Sand aquifer as it relates to the landfill site. This model was constructed using the previously mentioned pump test data.

Fuller and Warrick (1985) state that naturally occurring clays are rarely homogeneous enough to exist as liners without mechanical manipulation. The optimum condition for a liner then is to be semi-porous enough to allow leachate water through at a slow rate so that pollutants, such as metal cations, can be adsorbed by the clay. Fuller and Warrick list in their Table 8 metal cations such as Ca, Mg, Na, K, among others, to be common in leachate water of municipal solid waste landfills. Based upon the limited criteria established by this investigation, the montmorillonite clay liner in the Holmes Road Sanitary Landfill has the potential to be an effective barrier to the underlying Memphis Sand aquifer.

## REFERENCES

Hall, Blake and Assocaites, 1989. Hydrogeologic Report for Redesigh of Part of The Holmes Road Landfill, Unpublished in-house report, pp. 1-22.

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Fuller, W. H., and A. W. Warrick, 1985. Soils in Waste Treatment and Utilization. CRC Press, Inc., Boca Raton, pp. 2-43.

Graham, D. D., and W. S. Parks, 1985. Potential for Leakage Among Principal Aquifers in the Memphis Area, Tennessee. U.S.G.S. Water Resources Investigation Report 85-4295, Memphis, Tennessee, pp. 14-37.