VIRGINIA WATER RESOURCES RESEARCH CENTER

# Virginia Water Research Symposium 2000

## ADVANCES IN LAND AND WATER MONITORING TECHNOLOGIES AND RESEARCH FOR MANAGEMENT OF WATER RESOURCES

PROCEEDINGS

**Editor: Tamim Younos**

**Virginia Polytechnic Institute and State University**
**Blacksburg, Virginia Tech**

**P6-2000**

# FOREWORD

The purpose of the Virginia Water Research Symposium 2000, held on November 7-9, 2000 in Roanoke, Virginia was to facilitate an interdisciplinary forum to present and discuss advanced and innovative water monitoring technologies (physical, chemical, biological), research for natural waters (surface water, ground water, estuarine, wetlands, and precipitation, etc.), and advanced land-use monitoring (i.e., remote sensing and satellite imagery) that impact decision-making processes in the management of water resources. The symposium was intended for individuals involved with the development of monitoring technologies, research scientists (i.e., hydrologists, biologists, ecologists, chemists, water resources specialists, economists, engineers), educators, consultants, watershed managers, and policy/decision makers.

The symposium program was developed using a series of invited and submitted papers. Invited papers will be complied in a separate publication after the symposium. This publication is a compilation of many excellent papers submitted for oral and poster presentations in response to a "Call for Papers." Session topics for submitted papers included: parameter assessment and modeling applications, applications of GIS/GPS and remote monitoring, evaluation and assessment of monitoring programs, bacterial source tracking, and water monitoring/modeling in coastal environments. It is expected that these proceedings will serve as an updated reference for water quality and quantity assessment and set the direction for future research and technology transfer in the areas of water and land monitoring for effective management of water resources.

Finally, this proceeding is the first Virginia Water Resources Research Center's report to be distributed in CD-ROM format, and is therefore, in some ways an experiment. Appreciation is extended to Ms. Annabelle Fusilier for her assistance in producing the CD-ROM proceedings.

Tamim Younos
Symposium Chair &
Associate Director
Virginia Water Resources Research Center
Virginia Polytechnic Institute and State University

TABLE OF CONTENTS

16. Reservoir Mapping and Volume Measurements using a GPS/Depth Sounding System – Harold E. Post, Thomas J. Grizzard, Adil N. Godrej

17. Monitoring the Impacts of Animal Waste BMPs on Nutrient Losses in Runoff from the Owl Run Watershed – K.M. Brannan, S. Mostaghimi, P.W. McClellan

18. Evolution of Monitoring Program Strategies for Water Quality Management in the Occoquan Watershed – Randy Dymond,, Thomas J Grizzard, Adil N. Godrej, and Newland Agbenowosi

19. Lake Water Clarity Assessment of Minnesota's 10,000 Lakes: A Comprehensive View from Space – Leif G Olmanson, Patrick L. Brezonik, Steve M. Kloiber, Marvin E. Bauer

20. A New Tool for Tracing Human Sewage in Waterbodies: Optical Brightner Monitoring – Donald F. Wayne

21. Optrodes for Long-Term Remote Monitoring of Environmental Water Quality – Sean M. Christian, Paul Grems Duncan, and John A. Schroeder

22. Bacterial Source Tracking: A Tool for Total Maximum Daily Load Development – James Kern, Byron Petrauskas, Phillip McClellan, Vernon Shanholtz

23. Estimating Nonpoint Fecal Coliform Sources in Northern Virginia's Four Mile Run Watershed – George M. Simmons, Jr., Donald F. Wayne, Sue Herbein, Sharon Meyers and Ellen Walker

24. Evaluation of Water Quality Benefits From An Urban Stormwater Wetland System Retrofit – Dipmani Kumar, Russell Smith, Fred Rose

25. Lessons Learned from a New Environmental Monitoring Approach in the Mid-Atlantic Region – Tom DeMoss, Tom Pheiffer, Patricia Bradley

26. Pollution Source Identification Strategies for Coastal New Hampshire – Stephen H. Jones, Natalie Landry

# Evaluating Contaminated Metal Mine Drainage in Virginia Using Hyperspectral Techniques

J. E. Anderson, Ph.D.[1], Kimberly Royal [1] and Alan Bishop[2]

[1] Virginia Commonwealth University, Department of Biology
816 Park Avenue, Richmond, Virginia 23284
E-mail: John.Anderson@tec02.usace.army.mil

[2] Virginia Department of Mines and Minerals
Division of Mineral Mining
Charlottesville, Virginia 22903

## ABSTRACT

Hyperspectral imagery is maturing as a data source with the availability of several airborne and soon-to-be launched spaceborne sensor systems. The availability of these data is allowing researchers and analysts to detect and quantify a wide range of materials that possess unique spectral signatures in the reflected optical spectrum. The advantage of hyperspectral data is the availability of a full compliment of narrow spectral bands usually covering the ultraviolet through the short-wave infrared (350 nm – 2500 nm). We have successfully tested the capability of the HYMAP hyperspectral sensor to characterize bacterially-mediated iron oxide precipitates associated with acid and circumneutral discharges at Contrary Creek, Mineral Virginia. Contrary Creek is the site of several abandoned metal mines in Virginia's Gold-Pyrite Belt and exemplifies the problems associated with orphaned mine lands throughout the Commonwealth. Using field spectrometers we obtained in situ signatures for precipitates and associated ground features. Within the imagery, these field data were used to calibrate and spectrally match areas of acid and neutral precipitates occurring at the creek to detect and map the discharge locations. The unique spectral properties of the iron oxide precipitates permitted their separation and classification in image space. Spectral separation of acid versus neutral precipitates was found to be consistent with recent data obtained from contaminated streams in the both the Pennsylvania coal fields and other Virginia metal mine sites as well as measurements obtained by others for similarly impacted watersheds. As hyperspectral data becomes more available, it should soon be possible to inventory rogue discharges associated with orphaned mine lands based upon iron oxide precipitates.

**Keywords:** hyperspectral imagery, metal contamination, mine drainage

## INTRODUCTION

Mining in the Virginia Gold-Pyrite Belt has led to the abandonment of numerous orphaned metal mines. Gold, copper, arsenic, iron, lead and many other minerals have been extracted from this region that extends north to south between the eastern Appalachians and the Fall Line. Within this region many streams are impacted by contaminated mine drainage (CMD), heavy metals from contaminated spoils, and heavy sedimentation due to barren landscapes. Due to the historic concentration of metal mines and processing facilities in the Upper York River drainage, many tributary streams in this area have been adversely impacted.

When the discharge of CMD from abandoned, underground mines contains ferric iron, precipitates form downstream for tens of meters to kilometers from the discharge origin. Different minerals precipitate in stream under varied pH regimes and exhibit the many phases associated with the transformation of iron species from amorphous (i.e., ferrihydrite) to more crystalline forms (i.e., goethite) (Bigham et al. 1996). These iron oxide materials have been shown to have distinct, discrete spectral properties in the visible region of the reflected electromagnetic spectrum (450 nm to 760 nm) making it possible to detect and evaluate contaminated metal mine drainage. This study demonstrates the use of the relatively new Hyperspectral Mapping System (HYMAP) and the ability to use these data to detect and separate acid from circumneutral discharges associated with transforming iron species. Hyperspectral data differs from conventional aerial photography and multispectral data (i.e., LANDSAT) due to the number of spectral channels available and the narrowness of these channels. For example the Advance Visible and Near Infrared Imaging Spectrometer (AVIRIS) flown by the Jet Propulsion Laboratory offers 220 channels ranging from the visible through the short-wave infrared. The HYMAP system is comparable in capabilities and for the Contrary Creek mission was configured to acquire107 bands at 20 nm bandwidths ranging from 350 nm to 2500 nm (Figure 1). In addition, the spatial resolution of HYMAP was equivalent to 1 m / pixel ground sample distance.

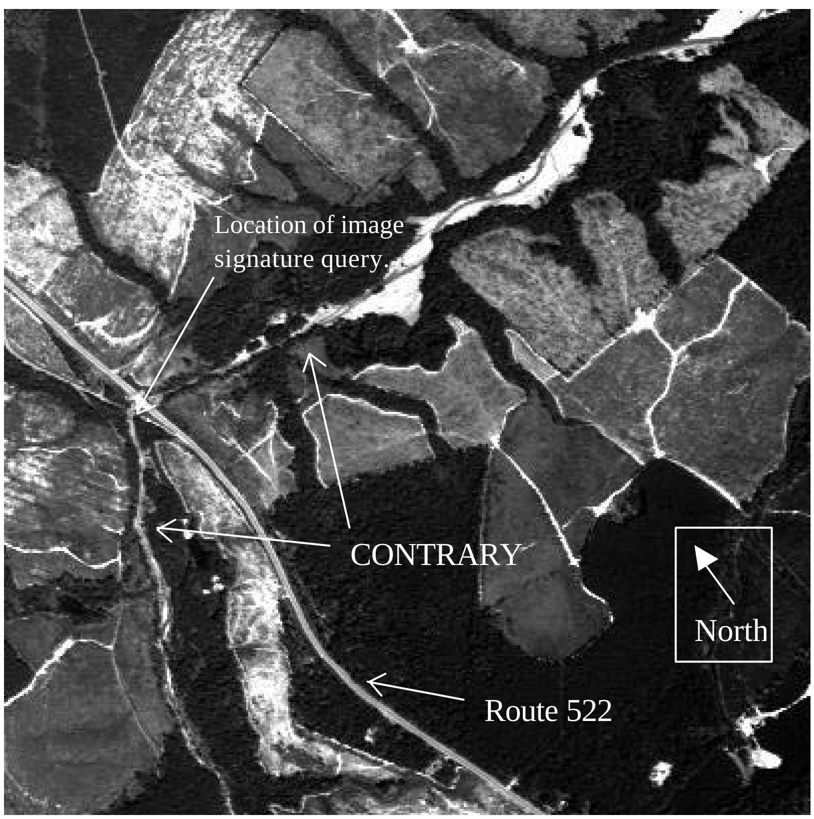



**Figure 1.** HYMAP Band 680 nm (red region) over Contrary Creek

Using the spectral channels offered by HYMAP and a spectral catalogue representing iron oxide precipitates forming at Contrary Creek we were able to perform empirical calibration of the data to the ground spectra. In addition, we were able to clearly distinguish, through spectral matching techniques, the heavily affected acidic main channel of the creek as well as map the minor discharge areas where neutral water enter the stream. The ability to perform this kind of evaluation using remote sensing presents a new and confident way to detect and quantify the effects of contaminated mine drainage originating from orphaned mine lands in the Commonwealth.

## SITE

Contamination of tributary streams within Virginia's York River drainage system is well documented. Most notably is Contrary Creek (38°06 54 N, -77°91 10 W Mineral USGS Quadrangle), the site of five major mines from the early 1900's until the 1960s. Several mining companies were still prospecting in the Contrary Creek area as late as the 1970's. Iron, gold, copper, and zinc were all mined at the creek (Sweet, 1976; Poole, 1974; Katz, 1961). Large spoil piles, pits, shafts, and rubble lie adjacent to the creek, exposing vast amounts of iron-sulfide minerals to chemical weathering and biological conversion. These processes produce highly toxic mining effluent, that enter the creek along a 1 km reach west of the Route 522 bridge.

Many studies exist on the adverse effects CMD has had on the water quality and biota at Contrary Creek and the (downstream) North Anna River in Louisa County. For 50 years, state academic and resource agency scientists have documented high metal loads in fish and invertebrates, heavy sedimentation, and low pH levels resulting from orphaned metal mines in these watersheds (Simmons 1971 and Chance 1971). An attempt at reclamation of the mines at Contrary Creek in the 1970s did not succeed in the recovery of the stream and may have exacerbated the problem (Nordstrom and Dagenhart 1978). In addition, there is no record of follow up monitoring or success criteria used to measure the effectiveness of the reclamation. A recent and more successful attempt at reclamation at the Cabin Branch Mine on Quantico Creek in Prince William County has effectively reduced metal loads and stabilized eroding soils (Bishop 1997, personal communication, Anderson 1997 unpublished data). This site is managed by the National Park Service and was reclaimed by the Virginia Department of Mines, Minerals, and Energy (DMME).

## BIOGEOCHEMISTRY

The abandoned mines lying within the Contrary Creek watershed contain shafts sunk on the eastern side of the creek bed. Mine waste is heaped on either side of the creek in large spoil piles. Two of the abandoned mines occupy the pre-impoundment stream near the confluence of Contrary Creek and Lake Anna. All of the mines contribute to a veneer of toxic, metal-laden alluvium that has killed most of the flood plain vegetation to the creek's mouth located downstream at Lake Anna.

The mine drainage at Contrary Creek fosters the accumulation of a carpet of yellow amorphous iron oxide precipitate during base flow. This "yellow-boy" precipitate coats bottom substrates to a depth of two to three centimeters. Precipitate does not appear to accumulate uniformly over the streambed, but is distributed throughout the reach extending from the pyrite and sulfur mine above the Route 522 bridge downstream to Lake Anna. This is possibly a result of the normal scouring action of the stream during high flows. The nature of the iron oxide makes it difficult to identify its exact mineral composition using laboratory spectroscopic techniques. Moran and Wentz (1974) describe the amorphous nature of the iron oxide as consistent with precipitates forming in acid mine drainage receiving heavy iron and sulfur mineral inputs. Spectroscopic analysis by Baker (1972) demonstrated that the formation of ferric hydroxide occurs during the oxidation of pyrite. This finding indicated that the acid precipitate forming at Contrary Creek, the waters of which typically range from 2.5 to 4.0 pH, is ferric hydroxide and its associated species. Recent analysis by Nord (USGS unpublished data, 1998) shows the precipitate has the characteristics of the more crystalline goethite and hematite, but not ferrihydrite (Bishop and Murad 1996).

Past analyses by Nordstrom and Dagenhart (1978) found that the Contrary Creek precipitate was dominated (74.7 percent) by goethite minerals. In addition to yellow, other colors ranging from dark brown to red are also associated with the Contrary Creek watershed. Many of these colors are indicative of the mineral transformations that occur as ferric hydroxide ages to form ferrihydrite, goethite, or hematite (Chukhrov *et al.*, 1974; Landa and Gast, 1973). In fact, the red-orange circumneutral discharges along the creek margins exhibit spectral responses associated with amorphous ferrihydrite (see figure) as it precipitates out at the redox boundary. As this material enters the creek it is immediately available for biological conversion and oxidation by members of the acid microbial consortium including *T. ferrooxidans* (Robbins et al. 2000).

The nature of the base flow and drainage system of Contrary Creek establishes a dynamic biogeochemical cycle that helps to maintain the acid conditions that accelerate the weathering of minerals and contribute to the toxicity of the stream water. The base flow of Contrary Creek originates from a very limited aquifer above the country rock in weathered residual soils and fractured bedrock (Leech, 1973). As a result, the creek's base flow is sensitive to seasonal shifts in water table height. These seasonal shifts in groundwater regulate the flow of water through the tailing piles. Where tailings are exposed to groundwater seeps, lateral interflow is encouraged by impermeable clay hardpans that underlie the tailings (Mortimer, 1976). The hardpans may also inhibit the percolation of rainwater into the ground. This kind of movement is consistent where mines are located near headwater streams. Hill (1977) found that groundwater velocities can approach 17 to 128 meters a day through mine tailings in West Virginia. Water table fluctuations that govern the residence time of the water in the mine spoils controls the leaching rate. Movement of water through the tailings also causes the mechanical erosion of waste minerals and sets up both the chemical and biological reactions responsible for the Contrary Creek acid mine drainage effluent.

## METHODS

### Spectral Reflectance Measurements

Field spectral reflectance measurements were acquired with an Analytical Spectral Devices Full Range Spectrometer (FR 120). The FR120 acquires a continuous spectrum from 350 nm to 2500 nm in 3 nm increments. Measurements were made during the HYMAP mission on September 26, 1999 from 1100 to 1200 local solar noon to create a catalogue of signatures representing both iron precipitates and bright and dark spectral endmembers for imagery reflectance calibration. All procedures for spectral data collection followed Satterwhite and Henley (1990). In stream precipitates were measured through the water column at nadir (90°) by leveling the spectrometers foreoptics with a bubble level. An 8° collimator was used at a distance of 1 m to sample. Calibration of the spectrometer was performed using a (NIST-certified) Spectralon standard. All measurements were made under clear sky conditions. Spectral measurements for additional features included road surfaces, vegetation, bare soil areas, shadowed areas and rock outcrops. All spectral data were interpolated from 3 nm to 5 nm using Matrix Laboratory (MATLAB) after post-processing using ASD software. These spectra helped develop the empirical calibration of the imagery.

### Hyperspectral Imagery Data

Collection of the HYMAP data occurred on September 26, 1999 at 1145 local solar time under cloudless skies. These data were acquired as part of a series of missions sponsored by Science Applications International Corporation (SAIC) for both civilian and military targets of interest. The data were of good quality and were virtually free of noise. Although the HYMAP sensor acquired a continuous VIS-NIR-SWIR imaging spectrum (400 nm – 2500 nm), the critical imagery spectral bands for this study were within the visible region (450 nm to 760 nm). The HYMAP data offered 25 bands in this region in which to analyze the mineralogy associated with CMD at Contrary Creek. An image cube representing the entire 107 band data set is presented in Figure 2. This figure illustrates multichannel nature of hyperspectral data as a "sagittal section" through the electromagnetic spectrum recorded as a series of images. The reflectance (bright areas) and absorptions (dark areas) of features change from one wavelength to another and can be queried during image processing by obtaining a trace for each pixel at a given x,y position for a given feature within the cube.

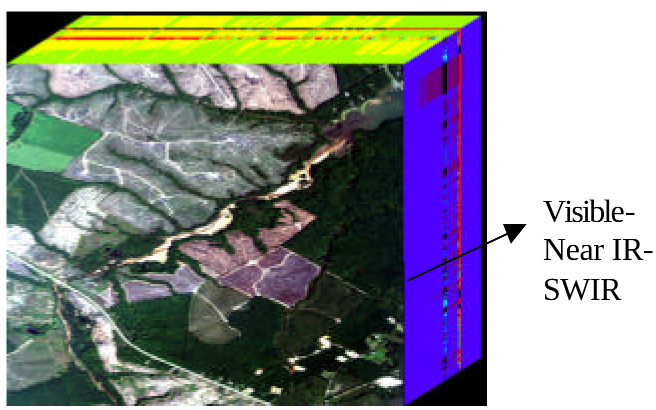



**Figure 2.** HYMAP data cube representing 107 bands of visible-near infrared-shortwave infrared data.

## HYMAP Data Processing

The HYMAP data were processed to calculate the apparent reflectance for target surface within the scene. Since field spectral measurements for targets of interest and associated spectral endmembers was collected at the time of the mission, empirical line calibration was performed using the ELC program within the Environment for the Visualization of Images (ENVI) Version 3.2 (Research Systems, Inc. 1999). The ELC method is a calibration technique that uses field reference reflectance data to match selected targets of known spectral characteristics within image data. A linear regression is used to define a new image resulting from the equation of the line *(y=ax+b)* that band for band equates imagery digital numbers and reflectance. This essentially removes the solar irradiance and atmospheric path radiance (Ferrier and Wadge 1996). To accomplish ELC on the HYMAP data, all spectra were used as input to the calibration algortihm.

## FIELD and IMAGE SPECTRAL RESULTS

### Field Spectrometry

Previous spectral investigations on iron oxides have shown the critical region for through the water column separation of iron rich acid and circumneutral precipitates occurs within the visible region waste (Robbins et al. 1996; Anderson and Robbins 1998; Bishop and Murad 1996; Ferrier 1999; Swayze et al. 1996). As shown in figure 3 acid precipitates associated with the iron minerals goethite and hematite exhibit a characteristic inflection around 650 nm. The inflection indicated by arrows 1 and 2 are typical of the mineral precipitates forming under acid conditions and are diagnostic for the presence of those conditions. This waveform creates a peak with spectral absorptions on either side.

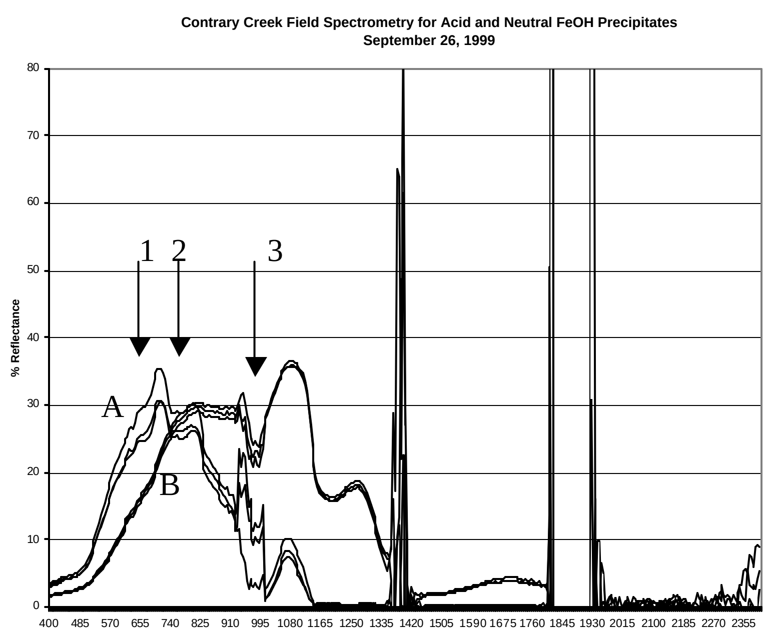



**Figure 3.** Field spectral reflectance for Contrary Creek precipitates (A – acid FeOH and B – neutral FeOH). Arrows indicate critical diagnostic inflections and absorption wavelengths for acid minerals.

The 650 nm and 750 nm inflections (Arrow 1 and 2) are characteristic of the minerals goethite, schwertmannite, and hematite (Bishop and Murad 1996). The inflection at arrow 3 is inherent in all FeOH minerals (acid and neutral) and is also diagnostic. Signature separability for acid versus neutral ASD spectral reflectance data was analyzed using a modified transformed divergence statistical routine (Jensen, 1996; Fischer 1996).

**Equation 1.**

$$TDiverge_{cd} = 2000\left[1 - \exp\left(- Diverge_{cd} / 8 \right)\right]$$

(Jensen 1996)

Transformed divergence is traditionally used in (imaging) remote sensing for training sample anaylsis and scores separability based upon expotentially decreasing weights for increasing distances between classes (Jensen, 1986). For the field spectral data, two “classes” were submitted to the algorithm: acid and neutral. These classes represented the average of all spectra collected at each site. The score for all possible combinations of 3 bands incorporating the 450 nm (blue), 550 (green), 650 nm (red), 680 nm and 700 nm (far red) and the 770 nm (near infrared) wavebands resulted in 1900 to 2000. This score was achieved for all combinations incorporating either the 680 nm or 650 nm bands with the 550 nm and 700 nm band. Scores obtained for data incorporating the 700 nm waveband were slightly less at <1900, but still satisfactory. These combinations effectively covered the absorption points recorded in the field spectra as being diagnostic for acid or circumneutral precipitates. According to Jensen, (1986), computed distances of classes using transformed divergence possess best separation where a score of 1900 or above is computed for all possible combinations. This technique quantitatively demonstrated the separation of acid and circumneutral iron oxides based on their spectral attributes.

**HYMAP Spectral Matching**

The queried image and library spectra for acid-mediated iron oxides and iron oxides forming under neutral conditions are presented in Figure 4. These traces show the spectral signature of the acid precipitates within the creek bed (A) and at a point below the Route 522 bridge where a seep zone discharges circumneutral groundwater into the creek (see Figure 1). Spectral angle mapping (SAM) was used to quantitatively test the fit between the library endmember spectra and the extracted image spectra. The algortithm determines the spectral similarity between two spectra by calculating the angle between the spectra as vectors occurring in a space with dimensionality equal to the number of spectral bands (Kruse et al 1993). The test report shows that additional signatures collected for the acid and neutral precipitates resulted in scores of 0.73 (SAM maximum = 0.78) for each endmember feature type. This results translates into a very high degree of similarity between the endmembers for each feature spectrum queried in the image and thus compared to the spectral library generated for features at Contrary.

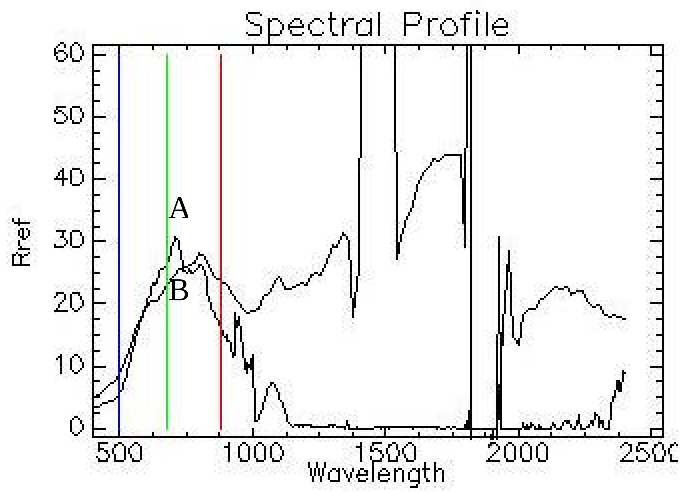



**Figure 4.** Acid (A) and neutral FeOH (B) spectral signatures queried from HYMAP image data cube.

**HYMAP Spectral Classification**

Using the spectral separation of the FeOH precipitates as a starting point, training statistics were developed for imagery pixels having signatures that were positively identified as acid FeOH (more goethite-like) or neutral FeOH (more amorphous, ferrihydrite-like). Training signatures were input to a Minimum Distance classification algorithm. Figure 5 presents the result of the spectral classification showing areas of abundant acid FeOH (arrow) and areas of circumneutral discharge points (arrows). The circumneutral discharges are associated with anoxic groundwater seeps precipitating neutral iron at the redox boundary. This boundary is the site of biological conversion by the neutral consortium of bacteria oxidizing Fe (II) to Fe (III). The overwhelming area classified in the HYMAP image scene is the main channel of Contrary Creek. This area is the site of abundant precipitates that have been spectrally matched to the catalogue signatures as acid-mediated FeOH. Also, some saturated areas that extend beyond the main channel have been classified as having acid-mediated iron minerals. These are seen as broad point bar deposits that are regularly flooded by higher flows.

## CONCLUSOINS

The use of the HYMAP imaging spectrometer to detect, separate, and map iron minerals associated with contaminated mine drainage at Contrary Creek proved very effective. The field optical spectroscopy and the spectral data offered by the imagery permitted the validation of spectral characteristics for each mineral phase of interest, acid or circumneutral.  In the case of Contrary Creek, past spectroscopy showed that the creek contains goethite and hematite iron minerals.  These minerals phases exhibited a distinct spectral signature (inflection at 650 nm) when measured with a field spectrometer or extracted as a signature from the HYMAP image cube.  In contrast, the minerals forming in circumneutral waters were identified as ferrihydrate and also possesed specific spectral characteristics (i.e., lack of the 650 nm inflection).  Based upon the spectral disparity of these signatures, a classification map was developed showing the minor neutral discharges into the creek.  This map also shows the overwhelming dominance of the acid waters presented by the creek.

With results such as these, it should be possible to leverage this technology against the need to inventory and prioritize orphaned mine sites in the Commonwealth. In addition, it should also be possible to check the success of mitigation and reclamation of sites by the Commonwealth's Division of Mineral Mining.

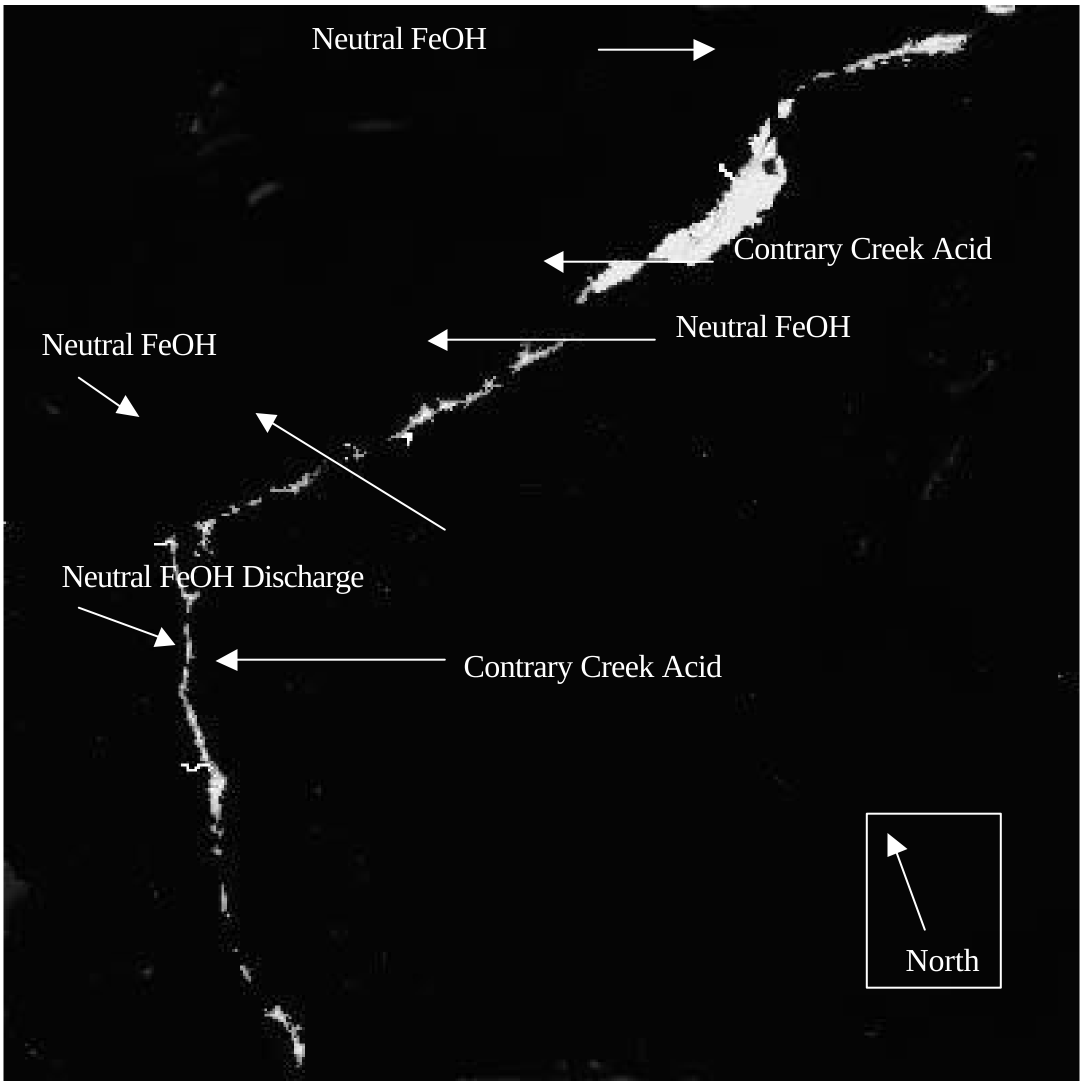



**Figure 5.** Spectral classification of Contrary Creek using acid and neutral FeOH signatures.

## REFERENCES


Anderson, J.E. and E.I. Robbins. 1998. Spectral reflectance and detection of iron-oxide precipitates associated with acidic mine drainage. *Photogrammetric Engineering and Remote Sensing.* Vol. LXIV, No. 12 1133-1236.

Baker, R.A. 1972. Evaluation of puritic oxidation of Mossbauer spectrometry. *Water Resources.* 6:9-17.

Bigham, J. , U. Schwertmann, S. Traina, R. Winland, and M. Wolf. 1996. Schwertmannite and the chemical modeling of iron in acid sulfate waters. *Geochimica Cosmochimica Acta*. 60:2111-2121.

Chance, D. 1971. Fish Kill 70-076. Va. State Water Control Board Memorandum, March 24, 1971.

Chukhrov, F.V., B.B. Zvyagin, A.I. Gorshkov, L.P. Yermilova, and V.V. Balashova. 1974. Ferrihydrite. *International Geology Review.* 16:1131-1143.

Ferrier, G. 1999. Application of imaging spectrometer data in identifying environmental pollution caused by mining at Rodaquilar, Spain. *Remote Sensing Environment.* 68:125-137.

Ferrier G. and G. Wadge. 1996. The application of imaging spectrometry data to mapping alteration zones associated with gold mineralization in southern Spain. *International Journal of Remote Sensing.* 17:331-350.

Fischer, R. 1996. A Program for Determining Transformed Divergence Using Field Spectral Data. USACE ERDC Topographic Engineering Center, Alexandria, VA

Hill, R.D. 1977. Elkins mine drainage pollution control demonstration project. Cincinnati, Ohio: U.S. Environmental Protection Agency (EPA).

Jensen, J.R. 1986. *Inroductory Digital Image Processing: A Remote Sensing Perspective*. Prentice Hall Publishers, New Jersey.

Kruse, F., A. Lefkoff, J. Boardmann. 1993. The spectral image processing system (SIPS) – interactive visualization and analysis of imaging spectrometer data. *Remote Sensing Environment.* 44:145-163.

Katz, A.S. 1961. The Mineralogy of the Sulfur Mine, Mineral, Virginia; Including a History of the Property. M.S. Thesis, Univ. of Va.

Landa, E.R. and R.G. Gast. 1973. Evaluation of crystallinity in ferric oxides. *Clays and Clay Minerals* 21:121-130.

Leech, P.H. 1973. Final Environmental Statement, North Anna Power Station— Virginia Electric and Power Company, U.S. Atomic Energy Commission.

Moran, R.E. and D.A.Wentz. 1974. Thermodynamic constraints on metal solubilities in a stream affected by mine drainage, Bonanza, Colorado. In *Water Resources Problems Related to Mining*, American Water Resources Association Proceeding 18, 54-64.

Mortimer, A.C. 1976. Mine Drainage Abatement; Techniques and Methods for Arminius Sites, Louisa County, Virginia. Richmond, Va.: Va. State Water Control Board.

Nordstrom, D.K. and T.V. Dagenhart, Jr. 1978. Hydrated iron sulfate minerals associated with pyrite oxidation: Field relations and thermodynamic properties. *Geological Society of America Abstracts*. 10:464.

Poole, J.L. 1974. Notes on some abandoned copper, lead and zinc mines in the Piedmont of Virginia. *Virginia Mineralogy*. 20:9-14.

Robbins, E.I., J.E. Anderson, M.H. Podwysocki, H.M. Edenborn, J.R. Eggleston, D. Growitz, R.L. Kleinmann, A.W. Norden, M.E. Passmore, B. Prugh, M. Stanton, P.C. Sweet, and T.M. Van Dyke. 1996. Red slime, oily films, and black rocks—Sensing the iron bacteria that make neutral streams look acid. In *Tenth V.E. McKelvey Forum on Minerals and Energy Resources,* L.M.H. Carter, editor. USGS Circular 1108. Reston, Va: USGS.

Robbins, E.I., J.E. Anderson, M.H. Podwysocki, and G. Nord. 2000. "Seasonal Variations in Spectral Reflectance of Microbial Flocculates, Precipitates, and Oil-like films Associated with Neutral and Acid Mine Drainage" In Environmental Monitoring and Biodiagnostics of Hazardous Contaminants. M. Healy et al. (eds.) Kluwer Academic Publishers. pp.243-266.

Satterwhite, M.B. and J.P. Henley. 1990. *Hyperspectral Signatures (400 to 2500 nm) of Vegetation, Minerals, Soils, Rocks, and Cultural Features: Laboratory and Field Measurements.* U.S. Army Corps of Engineers Engineer Topographic Laboratories Report ETL-0573. Alexandria, Va.: U.S. Army Corps of Engineers.

Simmons, G.M. 1971. A pre-impoundment ecological study of the benthic fauna and water quality in the North Anna River. In *Water Resources Research in Virginia—Annual Report for Fiscal Year 1971*. Bulletin 46. Blacksburg, Va.: Water Resource Research Center, VPI and State Univ., 13-52.

Swayze, G., R. Clark, R. Pearson, and K. Livo. 1996. Mapping acid-generating minerals at the California Gulch Superfund Site in Leadville, Colorado using imagingspectroscopy. In *Summaries of the 16th Annual JPL Airborne Earth Science Workshop*, JPL Publication 96-4, Jet Propulsion Laboratory, Pasadena, CA, Vol. 1 pp. 231-234.

Sweet, P.C. 1976. Silver in Virginia. *Virginia Mineralogy*. 22:9-11.

**Radium in Low-pH, High-Dissolved-Solids Ground Water in the Maryland Coastal Plain**

David W. Bolton, Maryland Geological Survey (corresponding author)
2300 St. Paul Street, Baltimore, MD 21218 (dbolton@mgs.md.gov)
John W. Grace, Maryland Department of the Environment
Harry J. Hansen, Maryland Geological Survey

KEYWORDS: ground water, radium, pH, chloride, coastal plain

## ABSTRACT

Ground water in parts of the Potomac Group and Magothy Formation in the Maryland Coastal Plain commonly contains radium-226 plus radium-228 concentrations that exceed the U.S. Environmental Protection Agency's Maximum Contaminant Level (MCL) of 5 picocuries per liter (pCi/L) (equivalent to 185 becquerels per cubic meter, or bq/m$^3$). Studies conducted in 1997 and 1998 determined that concentrations of naturally occurring radium, gross alpha-particle activity (GAPA), and gross beta-particle activity (GBPA) were positively correlated with specific conductance (a surrogate measure of total dissolved solids), and were negatively correlated with pH. Most of the high radionuclide concentrations were from wells located in the outcrop and updip areas of the Magothy Formation and Potomac Group in Anne Arundel County, where the aquifers are generally unconfined and, therefore, are more susceptible to contamination resulting from anthropogenic activities. Sodium and chloride were the ions most closely correlated with high radium concentrations. Radium-226 plus radium-228 exceeded the MCL in all samples having sodium concentrations greater than about 10 milligrams per liter (mg/L) and chloride concentrations greater than about 15 mg/L. High radium concentrations appear to be related to mobilization processes such as ion exchange and sorption/desorption that are affected by both natural and anthropogenic factors. Radium-224, an alpha-emitting isotope with a half-life of 3.64 days, was associated with large decreases in GAPA measured within three days of sampling (short-term GAPA) and measured again in the same sample after 30 days (long-term GAPA). As a result of these findings, all new wells withdrawing water from aquifers in the Magothy Formation or the Potomac Group in northern Anne Arundel County must be tested for short-term gross alpha-particle activity prior to the issuance of a certificate of potability.

## INTRODUCTION

In 1997, a pilot study of carcinogens in well water in Anne Arundel County, Maryland revealed that concentrations of radium-226 plus radium-228 in the Magothy Formation and Potomac Group aquifers commonly exceed the U.S. Environmental Protection Agency's (USEPA) drinking-water standard of 5 pCi/L (185 bq/m$^3$) (Bolton and Hayes, 1999). The study had been conducted at the recommendation of the Anne Arundel County Advisory Task Force on Cancer Control (1996), which was convened to investigate the county's disproportionately high cancer mortality rate relative to both Maryland and the United States. While no statistically significant higher rates of radium-associated cancers have been observed in the high-radium areas of Anne Arundel County, the pilot study was followed by a regional study in 1998. The objectives of the regional study were to: (1) further define the vertical and lateral extent of the high radium concentrations in Anne Arundel County, (2) determine whether high ground-water radium concentrations existed in the Magothy and Potomac Group aquifers elsewhere in the upper Chesapeake Bay area of Maryland, and (3) examine the relationship between ground-water chemistry and radionuclide concentrations.

This paper presents information from the pilot study and the regional study on the occurrence of radium in ground water in the Magothy Formation and Potomac Group aquifers of the upper Chesapeake Bay area in Maryland, and on the relations between ground-water radium concentrations and ground-water quality. Comprehensive presentations of the radionuclide data from the pilot study and the regional study are given in Bolton and Hayes (1999) and Bolton (in press), respectively.

## Radium: Principles and Health Effects

Radium is a naturally-occurring element that is produced from the spontaneous radioactive decay of uranium and thorium, which are present to some degree in all aquifer materials. Radium is an alkaline-earth metal whose chemical behavior is similar to calcium. Three radium isotopes (radium-226, radium-228, and radium-224) have been identified in ground water in the Magothy and Patapsco Formations in Anne Arundel County (Bolton and Hayes, 1999). Radium-226, an intermediate decay product in the uranium-238 series, emits alpha particles upon decay and has a half-life of 1,622 years. Radium-228 and radium-224 are intermediate decay products in the thorium-232 decay series; radium-228 (a beta-particle emitter) has a half-life of 5.75 years, while radium-224, an alpha-particle emitter, has a half-life of 3.64 days.

Radium-226 and radium-228 are human carcinogens (Cancer Group A) (U.S. Environmental Protection Agency, 1996). Most radium ingested through drinking water is eliminated in urine or feces. However, radium that is not eliminated can accumulate in the bone tissue, where the alpha and beta particles produced from the decay of radium can damage cell tissue and may induce cancer (National Research Council, 1988). Unlike many carcinogens, the human health effects of radium are well documented. Much of the knowledge of the health effects of radium is derived from studies of watch-dial painters in the early part of the twentieth century. These workers painted watch dials with radium, and would routinely shape the tips of their brushes with their tongues and lips. In the 1920s and 1930s, many workers developed head sarcomas and bone carcinomas (Milvy and Cothern, 1990). The USEPA established a MCL of 5 pCi/L (185 bq/$m^3$) for combined radium-226 plus radium-228 in public drinking-water supplies (Federal Register, 1976). There is no explicit MCL for radium-224; however, an MCL of 15 pCi/L (555 bq/$m^3$) has been established for GAPA to screen samples for testing for radium-226 and radium-228. Sample holding times are not specified in the Federal drinking-water regulations, however, and GAPA analysis would be unlikely to indicate the presence of radium-224 unless conducted within several days of sample collection.

## Geologic Setting

The study area is located in the upper Chesapeake Bay area of the Maryland Coastal Plain (fig. 1). The aquifers sampled during this project are the (Cretaceous) Patuxent, Patapsco, and Magothy Formations (fig. 2), which are major aquifers in the study area. The Patuxent and Patapsco Formations are the water-bearing units within the Potomac Group, which outcrops in a 15- to 25-km-wide, northeast-trending band extending from Prince George's County northeast to Cecil County. These formations are an important source of water both in the outcrop areas and in southern Maryland and the Eastern Shore, where they are confined. The Patuxent Formation outcrops in an irregular belt immediately adjacent to the crystalline Piedmont rocks and is composed of discontinuous and interconnected lenses of quartz and feldspathic sands and tough, variegated clays (Mack and Andreasen, 1991). The Patuxent Formation was deposited in a fluvial environment, and consists of a complex series of channel and point-bar sands and gravels interstratified with floodplain (or swamp) silts and clays. It is a multiple-layer aquifer system with individual sand beds, which are difficult to correlate between wells. Kaolinized feldspar, pyrite, and lignite are common; iron stains and siderite granules are also found. The Patuxent Formation is a significant aquifer in northwestern Anne Arundel and northern Prince George's Counties. The Patapsco Formation is a fluvio-deltaic deposit consisting of interbedded gray, brown, and red silt and clay and argillaceous quartzose sand and contains thin beds of gravel and siderite-cemented sandstone (Hansen, 1972; Fleck and others, 1996). Some of the sand beds are reported to be feldspathic and may contain siderite, hematite, or limonite; lignite and pyrite are commonly found in medium to dark gray clays (Glaser, 1969; Otton and Mandle, 1984). In Anne Arundel County, the thickness of individual sand, silt, and clay layers is generally less than about 6 meters thick, but can vary greatly.

The Magothy Formation outcrops in a 5- to 10-km-wide northeast-trending band across northern Anne Arundel County and in isolated exposures in northern Kent and eastern Cecil Counties on the Eastern Shore (Cleaves and others, 1968). The Magothy Formation is a fluviomarine deposit consisting of light gray to white, "sugary," medium-to coarse-grained quartzose sand and fine gravel interbedded with dark gray silts and clays and containing lignite and pyrite (Hansen, 1972). Although the Magothy and Patapsco Formations are separated by a regional unconformity, it is often difficult to identify the contact between the two formations using well logs and routinely collected drill cuttings. In some areas the two formations are hydraulically connected across a sand-on-sand contact and locally may function as a single aquifer.

## METHODS

Untreated water samples were collected from 219 wells completed in the Magothy Formation and the Potomac Group aquifers between September 1997 and November 1998. One hundred twenty-three wells were in Anne Arundel County; 96 wells were in other counties. Most wells were either private or public water-supply wells (80 percent and 15 percent, respectively); the rest were either institutional, industrial, commercial or recreational water-supply wells, or observation wells. Water samples were collected from wells after the water had run until field measurements (pH, specific conductance, dissolved oxygen, and water temperature, measured at 5-minute intervals) were stable and the water was clear.

Samples were analyzed for short-term (measured within 72 hours of sample collection) and long-term (measured 30 days after sample collection) GAPA and GBPA. Differences in the long-term and short-term values were used to indicate the presence of short-lived radionuclides such as radium-224. Samples from Anne Arundel County were analyzed for radium-226 and radium-228; samples collected outside of Anne Arundel County were analyzed only for GAPA and GBPA. Major ions were analyzed in selected wells in Anne Arundel County. Analysis for radium-226 (USEPA Method 903.0) and radium-228 analysis (USEPA Method 904.0) was performed by Quanterra Environmental Services under contract to the U.S. Geological Survey (USGS) National Water Quality Laboratory (NWQL) in Denver, Colorado. GAPA and GBPA analysis (USEPA Method 900) and major-ion analysis (Fishman, 1993; Fishman and Friedman, 1989) was performed by the NWQL.

Statistical analysis of the data was performed using non-parametric statistics, including the Spearman rank correlation test and the Kruskal-Wallis rank test, since none of the data were normally distributed. In this paper, radium-226, radium-228, GAPA, and GBPA are collectively referred to as "radionuclides" as a matter of convenience; this is not meant to imply that there are no other radionuclides present. Analysis for the presence of other radionuclides was not conducted in the study.

# RESULTS

## Distribution of Radionuclides in the Study Area

Radium-226 plus radium-228 concentrations from the Magothy and Potomac Group aquifers in Anne Arundel County ranged from less than 40 to 2,440 bq/m$^3$ (less than 1.1 to 66 pCi/L). Short-term GAPA ranged from less than 111 to 34,000 bq/m$^3$ (less than 3 to 919 pCi/L); short-term GBPA ranged from less than 148 to 6,960 bq/m$^3$ (less than 4 to 188 pCi/L). The highest radionuclide concentrations in the Magothy Formation were from wells located within an approximately 5-kilometer-wide zone in Anne Arundel County extending from the north shore of the Magothy River southwest to the Prince George's County line (fig. 3). All short-term GAPA from samples within this zone exceeded the MCL of 555 bq/m$^3$ (15 pCi/L). The boundary between high and low radionuclide concentrations (relative to drinking-water standards) was quite distinct. South and east of this zone, short-term GAPA from the Magothy Formation was less than the MCL, although some radionuclide concentrations were above the minimum reporting limits (MRLs). Radionuclide concentrations in the Potomac Group samples did not show as strong a geographical trend as samples from the Magothy Formation. Most samples exceeding 555 bq/m$^3$ (15 pCi/L) for short-term GAPA were from wells located in or near the outcrop area of the Potomac Group in Anne Arundel County (fig. 3).

Outside of Anne Arundel County, short-term GAPA in samples from the Magothy Formation and Potomac Group aquifers ranged from less than 111 to 5,180 bq/m$^3$ (less than 3 to 140 pCi/L); short-term GBPA ranged from less than 148 to 2,590 bq/m$^3$ (less than 4 to 70 pCi/L). Short-term GAPA exceeded 555 bq/m$^3$ (15 pCi/L) in only 8 percent (five of 65 samples) from the Potomac Group wells outside of Anne Arundel County (two wells in Prince Georges County and one well each in Baltimore, Harford, Cecil, and Kent Counties). None of the wells completed in the Magothy Formation outside of Anne Arundel County exceeded the MCL for GAPA.

Radium-224 was the dominant radium isotope in four samples that were tested (fig. 4). The presence of radium-224 is reflected in the observed decrease between the short-term and long-term GAPA measurements (fig. 5). Of the 82 samples having both short-term and long-term GAPA greater than the MRL of 111 bq/m$^3$ (3 pCi/L), 62 samples had long-term GAPA that was less than 40 percent of the short-term value. Long-term GAPA exceeded short-term GAPA in only one of 209 samples, and the difference between the values for this sample was within the total analytical error. There was also a decrease between long-term and short-term GBPA, although it was generally smaller than that observed for GAPA. Of the 115 samples for which both short-term and long-term GBPA were greater than the MRL of 148 bq/m$^3$ (4 pCi/L), 70 samples had long-term GBPAs between 50 and 80 percent of short-term GBPA. The short-lived beta-emitting isotopes whose decay is responsible for the decrease in beta-particle activity were not identified.

## Relation of Radionuclides to Ground-Water Quality

There was a statistically significant ($P$<0.01) negative correlation between pH and all measured radionuclides ($r_s$ range, -0.43 to -0.64; Spearman rank correlation test) (tab. 1; figs. 6a,b). There was also a statistically significant difference in radium-226 plus radium-228 and short-term GAPA when these were grouped according to pH quartiles (Kruskal-Wallis rank sum test; $P$<0.01). Median radium-226 plus radium-228 for samples in the lowest pH quartile was 444 bq/m$^3$ (12 pCi/L), whereas median radium-226 plus radium-228 values for the other three pH quartiles were all less than the MCL of 185 bq/m$^3$ (5 pCi/L). The concentrations of all measured radionuclides also showed a significant positive correlation with specific conductance, which is highly correlated ($r^2$= 0.94) with total dissolved solids (TDS; determined by residue on evaporation at 180 degrees Celsius) (tab. 1; figs. 7a, b). When samples were grouped by specific conductance quartiles, radium-226 plus radium-228 concentrations and short-term GAPA for the highest specific conductance quartile (median concentrations, 777 and 925 bq/m$^3$ [21 and 25 pCi/L], respectively) were significantly higher than the values from the other three specific conductance quartiles (Kruskal-Wallis rank sum test, $P<0.01$). Median specific conductance for samples having radium concentrations greater than the MCL was 178 microsiemens per centimeter (μS/cm), compared to 76 μS/cm for the samples with radium concentrations less than the MCL.

In Anne Arundel County, major-ion concentrations tended to be higher in samples with radium-226 plus radium-228 concentrations exceeding 185 bq/m$^3$ (5 pCi/L) than in samples containing less than this amount (fig. 8). Sodium, magnesium, potassium, chloride, nitrate, and ammonium were positively correlated with the radiochemical constituents, with sodium and chloride having the strongest correlations. Spearman rank correlation coefficients between sodium and short-term GAPA, short-term GBPA, and radium-226 plus radium-228 ranged from +0.77 to +0.88; for chloride, the range was +0.75 to +0.83. Sodium and chloride were also highly correlated with each other (fig. 9). All samples having sodium concentrations greater than about 10 mg/L and chloride concentrations greater than about 15 mg/L had radium-226 plus radium-228 concentrations greater than 185 bq/m$^3$ (5 pCi/L). For samples with chloride concentrations above about 10 mg/L, the molar ratio of sodium to chloride is almost exactly 1:1, indicating that the source or sources are likely those that contain sodium and chloride in a 1:1 ratio.

To investigate the relationship between the distribution of radium, chloride, and well depth in a local area, 18 wells were sampled along a north-south transect across a peninsula in northeastern Anne Arundel County, where the Magothy and Patapsco aquifers are unconfined (fig. 10). Well depths ranged from 17 to 58 meters (m) deep (median depth: 26 m). The transect location was selected because of the known high levels of radium and chloride in the area. Land use in the area is mostly residential, with the highest density residential development on the south side of the peninsula. Radium-226 plus radium-228 concentrations along the transect range from 174 to 2,440 bq/m$^3$ (4.7 to 66 pCi/L); chloride concentrations range from 3 to 380 mg/L. Concentrations of radium-226 plus radium-228 exceeding 740 bq/m$^3$ (20 pCi/L) were found in ground water encountered approximately 15 to 50 m below land surface beneath the central part of the peninsula, with the highest radium concentrations found in water samples from the central and southern parts of the transect at depths of less than 30 m. The distribution of radium is very similar to the chloride distribution, but does not correspond as closely to pH distribution (although the pH from most wells along the transect was less than 4). A zone of higher radium and chloride concentrations penetrates deeper into the aquifer near the center part of the peninsula. This may correspond to an area beneath the ground-water divide, where the recharge water has followed a deeper flow path. There is, however, insufficient water-level data to fully explain this relation.

## DISCUSSION

The high concentrations of radionuclides observed in low-pH and high-TDS ground water, in conjunction with the occurrence of ferric hydroxides, oxyhydroxides, and other sorbing minerals in the Potomac Group and Magothy Formation, suggest that sorption and desorption play a major role in radium mobilization and demobilization in the study area. Radium mobility in ground water has been shown to be controlled largely by adsorption or cation exchange (Herczeg and others, 1988; Szabo and Zapecza, 1991). Radium is strongly sorbed by quartz, kaolinite and other clay minerals, and especially by ferric oxides and hydroxides (Langmuir and Riese, 1982; Ames, McGarrah, and Walker, 1983; Ames, McGarrah, Walker, and Salter, 1983; Beneš and others, 1984). Radium desorption from sediments has been shown to be a function of salinity (Li and others, 1977; Kraemer and Reid, 1984; Miller and Sutcliffe, 1985; Webster and others, 1995). If the ground water has low TDS, radium may sorb onto the medium and radium concentrations in ground water will be low, since there are few other cations available to compete for sites on the exchanging medium. In ground water with high TDS, high concentrations of calcium, sodium, and other cations will compete effectively for exchange sites on the sorbing medium, smaller amounts of radium will be sorbed, and the concentration of radium will be higher than in low-TDS ground water. Radium concentrations were positively correlated with TDS in ground water in the Chickies Quartzite in southeastern Pennsylvania (Senior and Vogel, 1995) and in the Kirkwood-Cohansey aquifer system in New Jersey (Kozinski and others, 1995). In a study of ground water in Utah, radium was shown to be most mobile in chloride-rich reducing ground water with high TDS content (Tanner, 1964). Adsorption of radium onto kaolinite, ferric hydroxide and quartz particles is highly dependent on pH, with adsorption increasing at higher pH levels (Langmuir and Riese, 1982; Beneš and others, 1984). Sorption of radium is a reversible process that can be explained by ion exchange of radium for hydrogen (Beneš and others, 1984). Desorption is favored at low pH (high concentrations of hydrogen ions in solution). Radium concentrations were negatively correlated with pH in ground water in the Chickies Quartzite in southeastern Pennsylvania (Senior and Vogel, 1995) and in the Kirkwood-Cohansey aquifer system in New Jersey (Kozinski and others, 1995).

Water samples from wells in the updip areas of the Potomac Group and Magothy Formation aquifers tended to be more acidic than ground water observed in other aquifers of the Maryland Coastal Plain (Hansen, 1972; Shedlock and others, 1999). The low pH in some well-water samples (38 wells had pH less than 4) suggests that additional processes likely occur that further acidify infiltrating precipitation, which in central and eastern Maryland has a pH of 4.2 to 4.4 (National Atmospheric Deposition Program/National Trends Network, 2000). Processes that may act to lower ground-water pH in the study area include decomposition of organic matter in the soil zone, oxidation of lignite, pyrite oxidation, precipitation of ferric hydroxide, and oxidation of ammonium to nitrate (nitrification). The relative importance of these processes has not been investigated.

High ground-water radium concentrations associated with high specific conductance were most closely linked to high concentrations of sodium chloride dissolved in the ground water. The chloride concentrations associated with elevated radium concentrations were higher than would be expected from geological sources or recharge from precipitation. Potential sources of sodium chloride include brackish-water intrusion in areas adjacent to the Chesapeake Bay, septic-system leachate, de-icing salts applied to roads, and brine water used to backflush water-softening systems. The relative importance of these potential sources has not been determined. Brackish-water intrusion does not appear to be related to the high chloride concentrations in the Anne Arundel County wells because most of the affected wells are not located adjacent to shorelines, whereas most wells affected by brackish-water intrusion are located adjacent to brackish-water bodies (Drummond, 1988; Fleck and others, 1996). A sample from a well in Kent County had a chloride concentration of 560 mg/L and a short-term GAPA and GBPA of 1,330 and 1,040 bq/m$^3$ (36 and 28 pCi/L), respectively. This well is completed in a deep (more than 150 m) brackish-water zone at the base of the Potomac Group that likely originated from salt-water intrusion during a previous high stand of sea level rather than from current-day brackish-water intrusion from the Chesapeake Bay (Drummond, 1998). The high GAPA and GBPA from this well indicates that water associated with brackish-water intrusion can mobilize radium in these aquifers.

Water-softening systems are widely used in Anne Arundel County to remove iron, radium, and other cations. Many of these systems utilize an ion-exchange mechanism, in which well water is passed through a column containing resin beads coated with sodium ions. Iron and other undesirable cations (including radium) in the well water replace the sodium ions on the resin beads, increasing the sodium content of the water. Ion-exchange systems have been shown to be effective in removing radium from ground water (Clifford, 1990), and have been demonstrated to work in Anne Arundel County (Bolton and Hayes, 1999). Periodically, the sodium is replenished on the resin beads by passing a sodium-chloride brine solution through the exchange column. This rejuvenation process strips the resin beads of the sorbed iron and other cations and replaces them with sodium. This type of system has the potential to introduce both high-TDS wastewater and sorbed radium into the ground-water system. Brine wastewater from water-treatment plant ion-exchange has been shown to increase sodium and chloride concentrations in ground water (Walker, 1969).

Furthermore, radium concentrations in brine wastewater have been shown to be highly concentrated relative to concentrations in untreated ground-water (Hahn, 1990).

The higher radionuclide concentrations in the Magothy Formation and the Potomac Group aquifers in Anne Arundel County wells relative to the other counties in the upper Chesapeake Bay area may be related to several factors. The outcrop areas of the Magothy Formation and Potomac Group are wider in Anne Arundel County than in other counties, where they subcrop beneath younger formations. The aquifers in Anne Arundel County are usually unconfined in the updip areas and are more susceptible to contamination from anthropogenic sources, which can increase TDS and possibly increase radionuclide concentrations. Sampled wells in Anne Arundel County tended to be shallower (median well depth: 43 m) and tended to have lower pH (median pH: 4.1) than wells in the other counties (median depth, 61 m; median pH, 5.8). Specific conductance tended to be higher in samples from the regional wells than in samples from Anne Arundel County (median: 146 μS/cm and 119 μS/cm, respectively). The higher median specific conductance in the regional wells may reflect the natural geochemical evolution of the ground water along the flowpath (such as increased sodium, calcium, bicarbonate, and sulfate), rather than the high levels of anthropogenic sodium and chloride seen in Anne Arundel County. However, the differences in major-ion chemistry between the two groups of wells are not known since the regional wells were not sampled for major ions. The fact that several widely-spaced wells outside of Anne Arundel County had short-term GAPA exceeding 555 bq/$m^3$ (15 pCi/L) indicates that the potential exists to have elevated radium concentrations in these aquifers throughout the upper Chesapeake Bay area under suitable geochemical conditions such as low pH and high TDS.

The identification of radium-224 as a major component of total radium in ground water in the study area underscores the importance of analyzing GAPA shortly after sample collection. Sixty-two percent (38 of 61) of the samples in this study had short-term GAPA *greater than* the MCL and long-term GAPA *less than* the MCL; that is, 62 percent of the samples probably would not have been identified as MCL exceedances had they not been analyzed within 72 hours of sample collection.

As a result of this investigation, a short-term GAPA test is required for all new water-supply wells screened in the Magothy Formation and Potomac Group aquifers in central and northern Anne Arundel County. Ground water in other geologically similar areas of the Atlantic Coastal Plain may also be at risk for high radium concentrations, particularly in areas having low pH or relatively high TDS.

## SUMMARY AND CONCLUSIONS

This study was conducted to provide information on the occurrence and distribution of radium-226, radium-228, and short-term and long-term gross alpha-particle and gross beta-particle activity in the Magothy and Patapsco Formations in Anne Arundel County, and to identify other areas in the Coastal Plain of the upper Chesapeake Bay area of Maryland where high radium concentrations may exist in these aquifers. The conclusions of this work are:

Radium-226 plus radium-228 and gross alpha-particle activity commonly exceed the USEPA Drinking-Water Standards of 5 and 15 pCi/L (185 and 555 bq/$m^3$), respectively, in ground water from the Magothy Formation and Potomac Group aquifers of central and northern Anne Arundel County, Maryland.

2. Radium, gross alpha-particle activity, and gross beta-particle activity are negatively correlated with pH, and positively correlated with dissolved solids. Sodium and chloride are the ions that are most closely correlated with elevated radium concentrations. Sorption/desorption processes appear to play a major role in mobilization of radium to ground water in the study area.

3. Comparison of short-term (measured within three days) and long-term (measured after 30 days) gross alpha-particle activity, in conjunction with analysis for radium-224, suggests that radium-224 is a major component of total radium in these aquifers in the upper Chesapeake Bay area of Maryland.

4. Radium may be mobilized to ground water anywhere in the study area under conditions of low pH, high dissolved solids, or both.

## ACKNOWLEDGMENTS

This study was a cooperatively funded project by the Maryland Department of the Environment, the Anne Arundel County Health Department, the U.S. Geological Survey, the U.S. Environmental Protection Agency, and the Maryland Geological Survey, which is part of the Resource Assessment Service of the Maryland Department of Natural Resources. The Health Departments of Anne Arundel, Cecil, Harford, Kent, Prince George's, and Queen Anne's Counties, and the Baltimore County Department of Environmental Protection and Resource Management provided invaluable assistance in locating wells and securing permission to sample. The authors are grateful to the residents of Maryland and the operators of the public-supply wells who granted permission to collect samples from their wells.

## REFERENCES

Ames, L.L., J.E. McGarrah, and B.A. Walker, 1983, Sorption of trace constituents from aqueous solutions onto secondary minerals. II. Radium: Clays and Clay Minerals, vol. 31, no. 5, p. 335-342.

Ames, L.L., J.E. McGarrah, B.A. Walker, and P.F. Salter, 1983, Uranium and radium sorption on amorphous ferric oxyhydroxide: Chemical Geology, vol. 40, p. 135-148.

Anne Arundel County Advisory Task Force on Cancer Control, 1996. Final Report, vol. 79 p., plus appendixes.

Beneš, P., P. Strejc, and Z. Lukavek, 1984, Interaction of radium with freshwater sediments and their mineral components: I. Ferric hydroxide and quartz: Journal of Radioanalytical and Nuclear Chemistry, vol. 82, no. 2, p. 275-285.

Bolton, D.W., in press, Occurrence and distribution of radium, gross alpha-particle activity, and gross beta-particle activity in ground water in the Magothy Formation and Potomac Group aquifers, upper Chesapeake Bay area, Maryland: Maryland Geological Survey Report of Investigations No. 70.

Bolton, D.W., and M.A. Hayes, 1999, Pilot study of carcinogens in well water in Anne Arundel County, Maryland: Maryland Geological Survey Open-File Report 99-02-10, 56 p.

Cleaves, E.T., J. Edwards, Jr., and J.D. Glaser, compilers, 1968, Geologic map of Maryland: Maryland Geological Survey, 1 sheet, scale 1:250,000.

Clifford, D.A., 1990, Removal of radium from drinking water, *in* Cothern, C. R., and P.A. Rebers, eds., Radon, radium, and uranium in drinking water: Lewis Publishers, Chelsea, Michigan, p. 225-247.

Drummond, D.D., 1988, Hydrogeology, brackish-water occurrence, and simulation of flow and brackish-water movement in the Aquia aquifer in the Kent Island area, Maryland: Maryland Geological Survey Report of Investigations No. 51, 131 p.

————, 1998, Hydrogeology, simulation of ground-water flow, and ground-water quality of the upper coastal plain aquifers in Kent County, Maryland: Maryland Geological Survey Report of Investigations No. 68, 76 p.

Federal Register, 1976, vol. 41, no. 133, July 9, 1976: U.S. Government Printing Office, Washington, D.C., p. 28402-28409.

Fishman, M.J., 1993, Methods of analysis by the U.S. Geological Survey National Water Quality Laboratory--Determination of inorganic and organic constituents in water and fluvial sediments: U.S. Geological Survey Open-File Report 93-125, 217 p.

Fishman, M.J., and L.C. Friedman, 1989, Methods for determination of inorganic substances in water and fluvial sediments: U.S. Geological Survey Techniques of Water-Resources Investigations, book 5, chap. A1, 545 p.

Fleck, W.B., D.C. Andreasen, and B.S. Smith, 1996, Geohydrologic framework, ground-water quality and flow, and brackish-water intrusion in east-central Anne Arundel County, Maryland, with a section on simulation of brackish-water intrusion in the Aquia aquifer in the Annapolis area using a solute-transport model: Maryland Geological Survey Report of Investigations No. 62, 136 p.

Glaser, J.D., 1969, Petrology and origin of Potomac and Magothy (Cretaceous) sediments, Middle Atlantic Coastal Plain: Maryland Geological Survey Report of Investigations No. 11, 101 p.

Hahn, N., 1990, Disposal of radium from drinking water treatment, *in* Cothern, C.R., and P.A. Rebers, eds. Radon, radium and uranium in drinking water: Lewis Publishers, Chelsea, Michigan, p. 249-267.

Hansen, H.J., 1972, A user's guide for the artesian aquifers of the Maryland Coastal Plain, Part two: Aquifer characteristics: Maryland Geological Survey, 123 p.

Herczeg, A.L., H.J. Simpson, R.F. Anderson, R.M. Trier, G.G. Mathieu, and B.L. Deck, 1988, Uranium and radium mobility in groundwaters and brines within the Delaware Basin, southeastern New Mexico, U.S.A: Chemical Geology (Isotope Geoscience Section), vol. 72, p. 181-196.

Kozinski, J., Z. Szabo, O.S. Zapecza, and T.H. Barringer, 1995, Natural radioactivity in, and inorganic chemistry of, ground water in the Kirkwood-Cohansey aquifer system, southern New Jersey, 1983-89: U.S. Geological Survey Water-Resources Investigations Report 92-4144, 129 p.

Kraemer, T.F., and D.F. Reid, 1984, The occurrence and behavior of radium in saline formation water of the U.S. Gulf Coast region: Isotope Geoscience, vol. 2, p. 153-174.

Langmuir, D., and A.C. Riese, 1982, The mobility of radium in groundwater: Geological Society of America Abstracts with programs, vol. 14, no. 5, p. 540-541.

Li, Y-H., G. Mathieu, P. Biscaye, and H.J. Simpson, 1977, The flux of $^{226}$Ra from estuarine and continental shelf sediments: Earth and Planetary Science Letters, vol. 37, p. 237-241.

Mack, F.K., and D.C. Andreasen, 1991, Geohydrologic data for the coastal plain sediments underlying Broadneck Peninsula, Anne Arundel County, Maryland: Maryland Geological Survey Open-File Report No. 92-02-6, 70 p.

Miller, R.L., and H. Sutcliffe, Jr., 1985, Occurrence of natural radium-226 radioactivity in ground water of Sarasota County, Florida: U.S. Geological Survey Water-Resources Investigations Report 84-4237, 34 p.

Milvy, P., and C.R. Cothern, 1990, Scientific background for the development of regulations for radionuclides in drinking water, *in* Cothern, C.R., and Rebers, P.A., eds., Radon, radium and uranium in drinking water: Lewis Publishers, Chelsea, Michigan, p. 1-16.

National Atmospheric Deposition Program/National Trends Network, Colorado State University, at URL http://nadp.sws.uiuc.edu/nadpdata, accessed 1/22/00.

National Research Council, 1988, The health risks of radon and other internally deposited alpha-emitters, BEIR IV: National Academy Press, Washington, D.C., 602 p.

Otton, E.G., and R.J. Mandle, 1984, Hydrogeology of the upper Chesapeake Bay area, Maryland, with emphasis on aquifers in the Potomac Group: Maryland Geological Survey Report of Investigations No. 39, 62 p.

Senior, L.A., and K.L. Vogel, 1995, Radium and radon in ground water in the Chickies Quartzite, southeastern Pennsylvania: U.S. Geological Survey Water-Resources Investigations Report 92-4088, 145 p.

Shedlock, R.J., J.M. Denver, M.A. Hayes, P.A. Hamilton, M.T. Koterba, L.J. Bachman, P.J. Phillips, and W.S.L. Banks, 1999, Water-quality assessment of the Delmarva Peninsula, Delaware, Maryland, and Virginia: Results of investigations, 1987-1991: U.S. Geological Survey Water-Supply Paper 2355-A, 41 p.

Szabo, Z., and O.S. Zapecza, 1991, Geologic and geochemical factors controlling uranium, radium-226, and radon-222 in ground water, Newark Basin, New Jersey, *in* Gundersen, L.C.S., and Wanty, R.B., eds., Field studies of radon in rocks, soils, and water: U.S. Geological Survey Bulletin 1971, p. 243-265.

Tanner, A.B., 1964, Physical and chemical controls on distribution of radium-226 and radon-222 in ground water near Great Salt Lake, Utah, *in* Adams, J.A.S., and Lowder, W.M., eds., The Natural Radiation Environment: University of Chicago Press, Chicago, Illinois, p. 253-278.

U.S. Environmental Protection Agency, 1996, Drinking water regulations and health advisories: EPA 822-B-96-002, Office of Water, Washington, D.C., 11 p.

Walker, W.H., 1969, Illinois ground water pollution: Journal of the American Water Works Association, vol. 61, p. 31-40.

Webster, I.T., G.J. Hancock, and A.S. Murray, 1995, Modelling the effect of salinity on radium desorption from sediments: Geochimica et Cosmochimica Acta, vol. 59, p. 2469-2476.

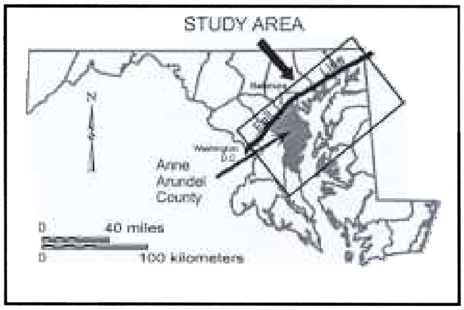



**Figure 1. Map of Maryland, Showing Location of the Study Area.**

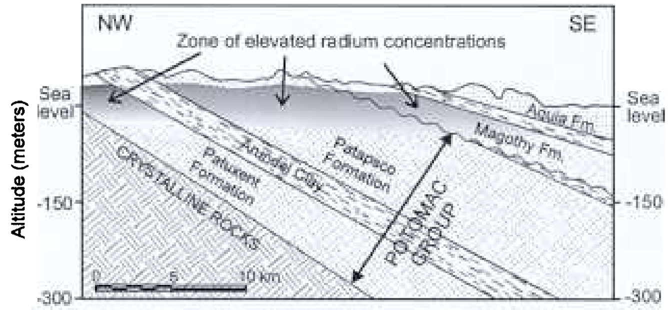



**Figure 2. Generalized Geologic Section across Anne Arundel County, Maryland, showing the Zone of Elevated Ground-water Radium Concentrations.**

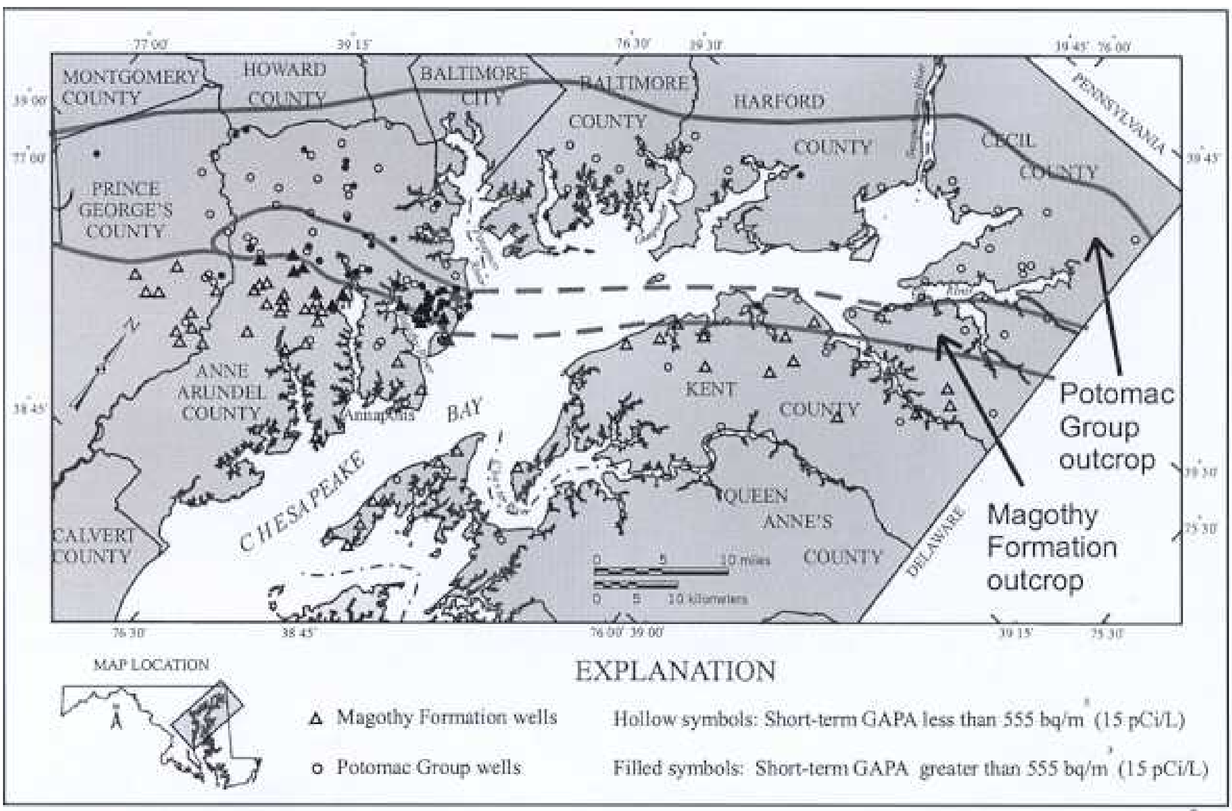



**Figure 3. Distribution of Short-Term Gross Alpha-Particle Activity (GAPA) from Well-Water Samples in the Magothy Formation and the Potomac Group of the Upper Chesapeake Bay Area of Maryland.**

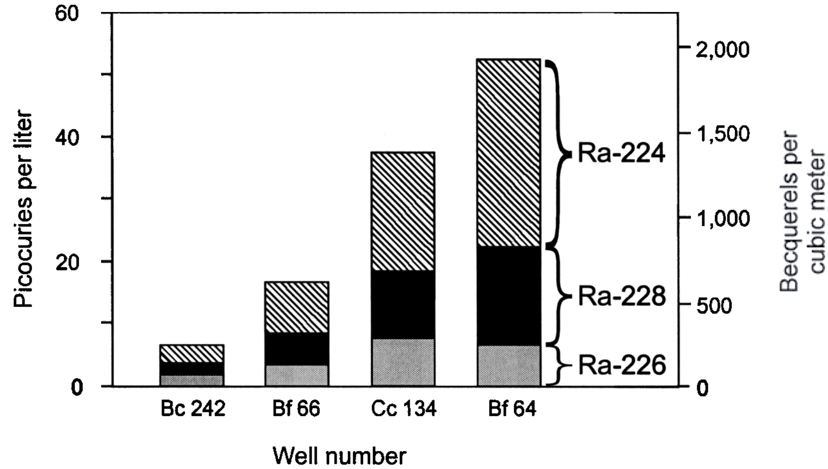



**Figure 4. Concentrations of Radium-224, Radium-226, and Radium-228 in Wells in Anne Arundel County, Maryland. Source: Bolton and Hayes (1999).**

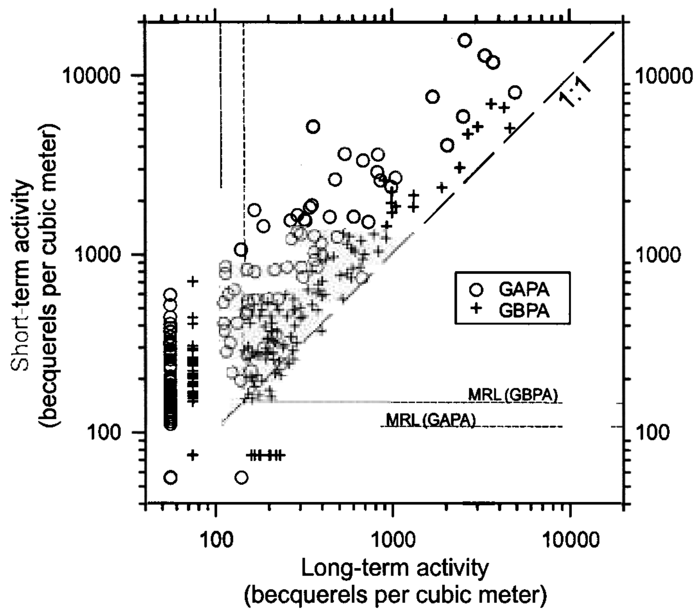



**Figure 5. Relation Between Short-Term and Long-Term Measurements of Gross Alpha-Particle Activity (GAPA) and Gross Beta-Particle Activity (GBPA). Values below Minimum Reporting Levels (MRLs) are plotted at one-half the MRL.**

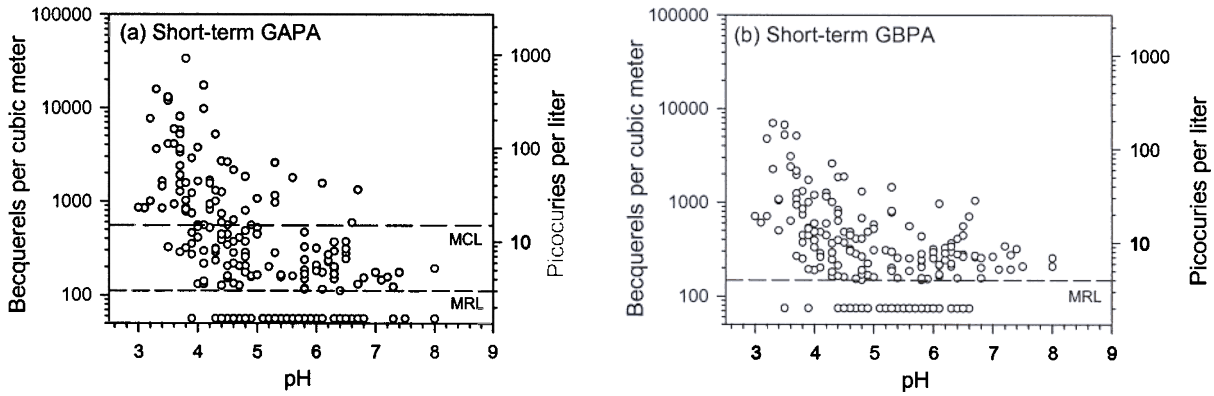



**Figure 6. Relation Between pH and: (a) Short-Term Gross Alpha-Particle Activity (GAPA); (b) Short-Term Gross Beta-Particle Activity (GBPA). MCL, Maximum Contaminant Level; MRL, Minimum Reporting Level. Values less than MRL are plotted at one-half the MRL.**

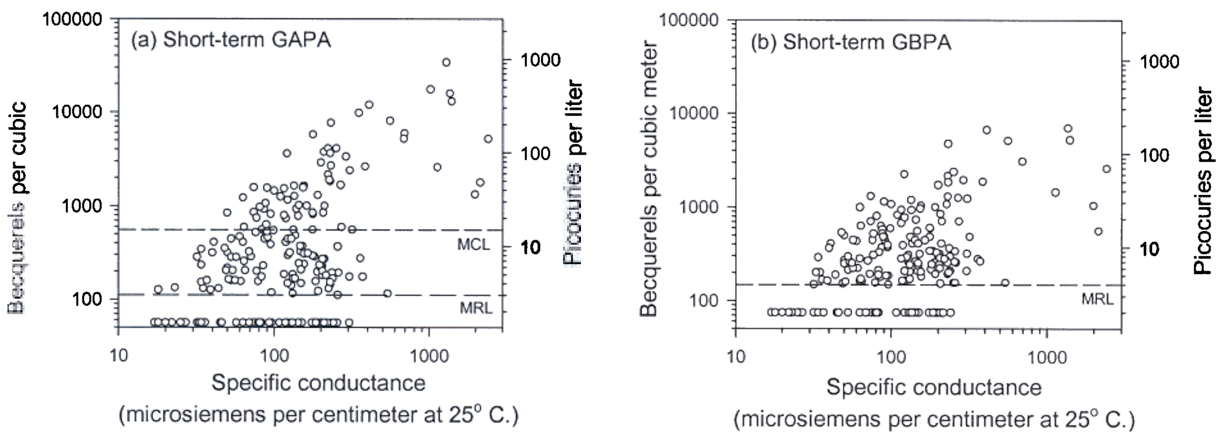



**Figure 7. Relation Between Specific Conductance and: (a) Short-Term Gross Alpha-Particle Activity (GAPA); (b) Short-Term Gross Beta-Particle Activity (GBPA). MCL, Maximum Contaminant Level; MRL, Minimum Reporting Level. Values less than MRL are plotted at one-half the MRL.**

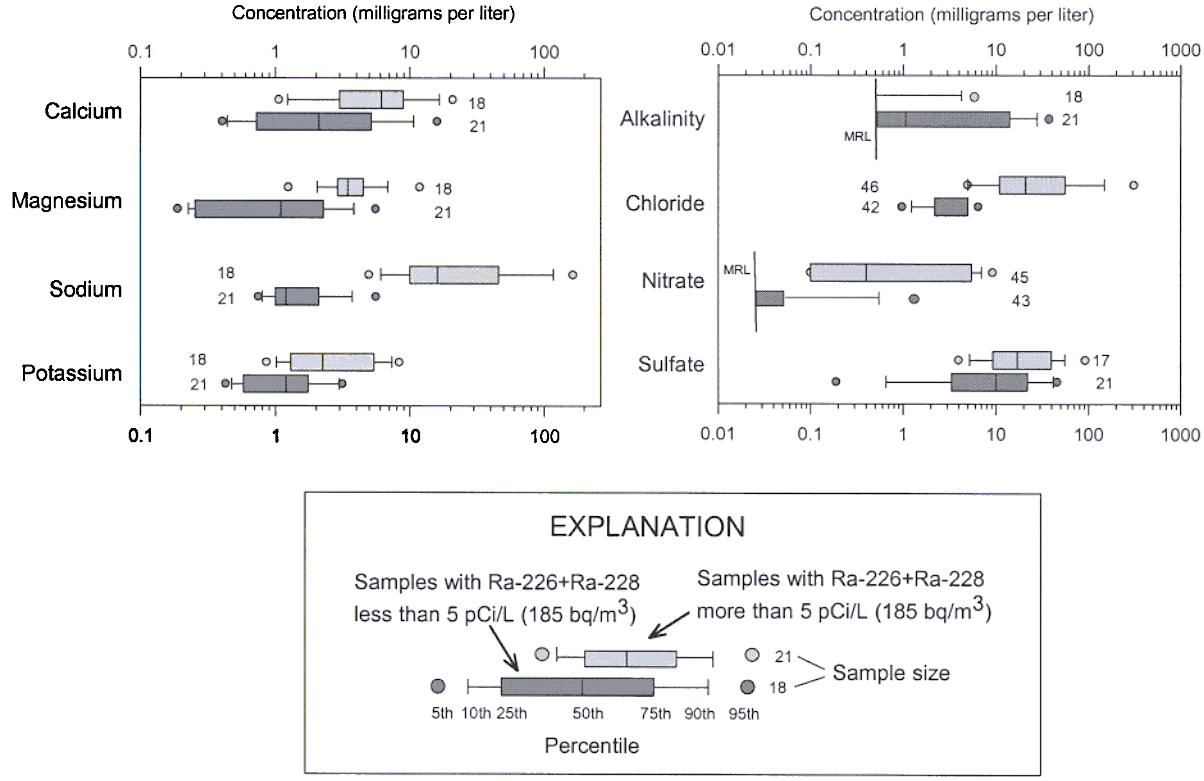



**Figure 8. Comparison of Cation and Anion Concentrations Between Samples Having Radium-226 plus Radium-228 Concentrations Greater or Less than the Maximum Contaminant Level of 5 Picocuries per Liter (185 Becquerels per Cubic Meter). MRL, Minimum Reporting Level.**

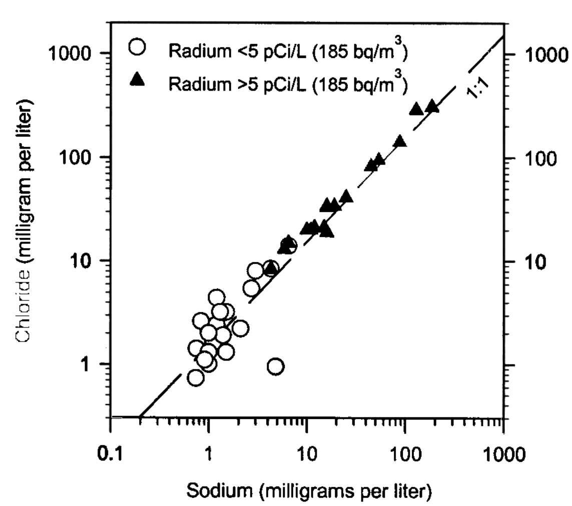



**Figure 9. Relation between Sodium, Chloride, and Radium-226 plus Radium-228 Concentrations.**

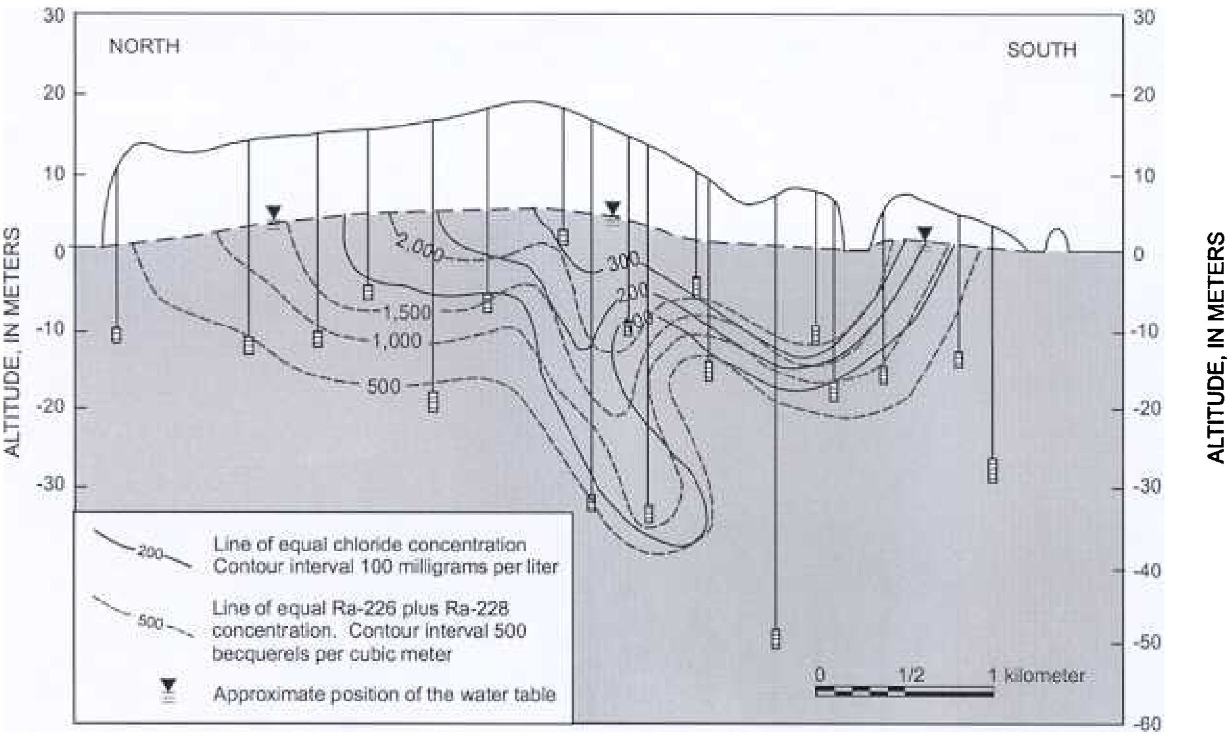



**Figure 10. Relation Between Radium-226 plus Radium-228 and Chloride Concentrations in Northeast Anne Arundel County, Maryland.**

**Table 1. Spearman Correlation Coefficients for Radiochemical Constituents, pH, and Specific Conductance.**

[$r_s$, Spearman correlation coefficient; n, sample size. $P<0.01$ for all cells.]

| Constituent | | Short-term gross alpha-particle activity | Long-term gross alpha-particle activity | Short-term gross beta-particle activity | Long-term gross beta-particle activity | Radium-226 plus radium-228 |
|---|---|---|---|---|---|---|
| pH | $r_s$ | -0.62 | -0.64 | -0.43 | -0.44 | -0.54 |
| | n | 207 | 217 | 197 | 217 | 122 |
| Specific conductance | $r_s$ | 0.37 | 0.37 | 0.43 | 0.51 | 0.69 |
| | n | 209 | 219 | 199 | 219 | 122 |

# Watershed Management Impacts on the South Fork Rivanna Reservoir near Charlottesville, Virginia

Stephen P. Bowler
County of Albemarle
401 McIntire Road, Room 211
Charlottesville, VA 22903

E-mail: sbowler@albemarle.org

## ABSTRACT

The two major issues facing managers of Charlottesville and Albemarle County's major reservoir since its construction in 1966 have been sedimentation and eutrophication. The most significant management action directed at point sources of pollution was the construction of a sewage interceptor to remove a large pollution source from the watershed (1988). The major management actions directed at nonpoint sources included implementation of agricultural best management practices (late 1980s and early 1990s) and installation of a regional stormwater basin (1993). A major change in the reservoir was the installation of a hydropower plant (1988).

Data from (summer) reservoir water quality samples collected during four periods of four years each between 1980 and 1996 were analyzed to consider the impacts of the management efforts. Variables included total suspended solids, total phosphorus, nitrate/nitrite, dissolved oxygen and temperature profiles, and algal community composition. An effort was made to account for variation in weather conditions using discharge prior to sampling at a stream gage on the dominant reservoir tributary and air temperature data.

Phosphorus concentrations declined from 0.045 mg/L in the 1980-83 period to 0.027 mg/L in the 1993-96 period. However, algal counts increased from 6434 cells/mL in the 1980-83 period to 19897 cells/mL in the 1993-96 period. A likely explanation for the increase in algae in spite of the reduction in phosphorus was hydraulic changes in the reservoir due to the hydropower plant operation. Stratification (and surface water residence time) appeared to increase as evidenced by an increase in temperature contrasts in the water column from 2.6 °C in the 1984-87 period to 4.1 °C in the 1993-96 period. (Despite the increased algal counts, the hydropower plant may provide a net benefit for water treatment for other reasons.)

Suspended sediment concentrations did not show a distinct time trend. However, sediment deposition is probably better assessed through bathymetric surveys and tributary load analysis.

**Keywords**: reservoir, eutrophication, watershed, hydropower, algae, phosphorus

## INTRODUCTION

The South Fork Rivanna Reservoir (SFRR) was filled in 1966 to serve as the primary public water supply for the City of Charlottesville and Albemarle County, Virginia. Over the years the SFRR has suffered from the problems typical of impoundments- eutrophication and sedimentation. Eutrophication has resulted in summer algal blooms and low oxygen conditions leading to occasional drinking water taste and odor problems and general concerns regarding the health of the reservoir ecosystem. Sedimentation has led to $49 \times 10^6$ L per year of lost storage capacity amounting to roughly ¾% of original volume (Betz Environmental Engineers 1977, Black & Veatch 1995).

Serious concerns about reservoir water quality and resulting efforts to protect it date back to at least 1974 (Betz Environmental Engineers 1977). Since that time, strategies to reduce nonpoint source pollution loads have included installation of agricultural best management practices, correction of road related erosion problems, and installation of a regional stormwater basin on a reservoir tributary (Table 1). The primary effort to reduce point source pollution was the installation of a sewage interceptor to remove residential and industrial waste generated in Crozet, the largest town in the basin. Presumably, the establishment of the National Pollution Discharge Elimination System had an effect on point source loads as well. Two aspects of reservoir operation could have affected conditions. First, a bubbler system was used in an attempt to cause mixing near the dam during periods of the late 1970s and early 1980s). It was not found to be very effective (F.X. Browne Associates, Inc. 1993). Second, a hydropower plant was installed in the dam in 1988.

Finally, it should be noted that there were also important watershed management actions taken to reduce impacts of new development on the reservoir. These actions included controversial down zoning actions and development of erosion and sediment control and stormwater ordinances. They would not have been expected to reduce the existing pollutant loads, but may have dampened increases from new sources.

**Table1.** Major watershed and reservoir management actions that potentially could have caused changes in the South Fork Rivanna Reservoir near Charlottesville, VA by time period.

| Time Period of Interest | | Major Management Actions Potentially Affecting Time Period of Interest |
|---|---|---|
| 1980-83 | General | • Bubbler mixing system in operation near dam. |
| 1984-87 | | • No direct action. |
| 1989-92 | 1988 | • Crozet interceptor goes on line removing residential and commercial waste from the SFRR watershed.<br>• Hydroplant installed at SFRR dam. |
| 1993-96 | 1993 | • Completion of installation of 67 agricultural best management practices (BMPs) and various highway erosion reduction projects in the 80's and early 90's.*<br>• Lickinghole Basin, a regional stormwater basin serving Crozet, VA completed.* |

*Funded by EPA "Clean Lakes" Grants (CWA Section 314).

Reservoir investigations date back to 1975 (Betz Environmental Engineers 1977). Data were collected in 19 of the 25 years between 1975 and 1999. While consistent data collection methods often were used, these data never have been analyzed in a comprehensive manner. This paper is an attempt to view those data comprehensively. Eutrophication is addressed through the following questions:

- Did nutrient loads change through major eras of reservoir history?
- Did the algal community change through those eras?

- Did algal community changes appear to relate to chemical variables (nutrient loads), physical variables, or both?
- Was there any particular change around the year 1988 when both the sewage interceptor and the hydropower plant went online?

Sedimentation is addressed by asking:

- Did changes in suspended solid concentrations in the reservoir occur?

## METHODS

### *Site and Samples*

The SFRR has a watershed area of 629 $km^2$ dominated by forest and pasture with smaller proportions of residential area and row crops (Figure 1). The western edge of the watershed is the Blue Ridge Mountains (Skyline Drive). The watershed also drains a substantial amount of rolling piedmont. The dam is northwest of the City of Charlottesville 0.75 km west of Route 29. SFRR had an initial surface area of 1.58 $km^2$ and volume of $6.35x10^9$ L. Annual flow into the reservoir is $291x10^9$ L. Mean residence time is roughly eight days. Raw water is drawn at the dam for treatment at the drinking water plant. Water is drawn from any one of three intakes placed at 1.5, 3, and 4.6m of depth. Maximum raw water withdrawal is 0.5 cms. There is a hydro-plant intake in front of the dam capable of drawing between 2.1 and 9.9 cms from the bottom of the reservoir. There are two 0.9 m mud gates at the base of the dam.

Samples used in this analysis were taken roughly 25m in front of the dam at mid reservoir. Two other sample sites existed, one near the longitudinal midpoint of the reservoir pool and one in the Ivy Creek tributary arm of the reservoir. Emphasis was placed on the samples taken at the dam because they integrated the entire reservoir and were nearest the drinking water intakes. Surface samples were grabbed just below the water surface. Bottom samples were collected with a Kemmerer style sampler at a depth of 12m. Dissolved oxygen and temperature measurements were obtained with a YSI 158 (or prior equivalent) meter at 0.5m intervals working up from 12m of depth. Stations were sampled monthly or twice monthly between 1 June and 30 September under all weather conditions except hazardous conditions (such as lightening or flooding).

### *Laboratory Analysis and Environmental Data*

Total phosphorus, total nitrogen, total suspended solids, and chlorophyll a readings and algal counts were obtained using procedures described in the 18$^{th}$ edition of the Standard Method for the Examination of Water and Wastewater or a prior equivalent (Table 2.a., Greenberg et. al. 1992). Analyses were carried out by the Rivanna Water and Sewer Authority (RWSA) laboratory or laboratories contracted by RWSA. Because the data were collected for different studies, not all samples were collected in all years (Table 2.b.). In some years, there was no sampling at all. Sampling methods generally were consistent.

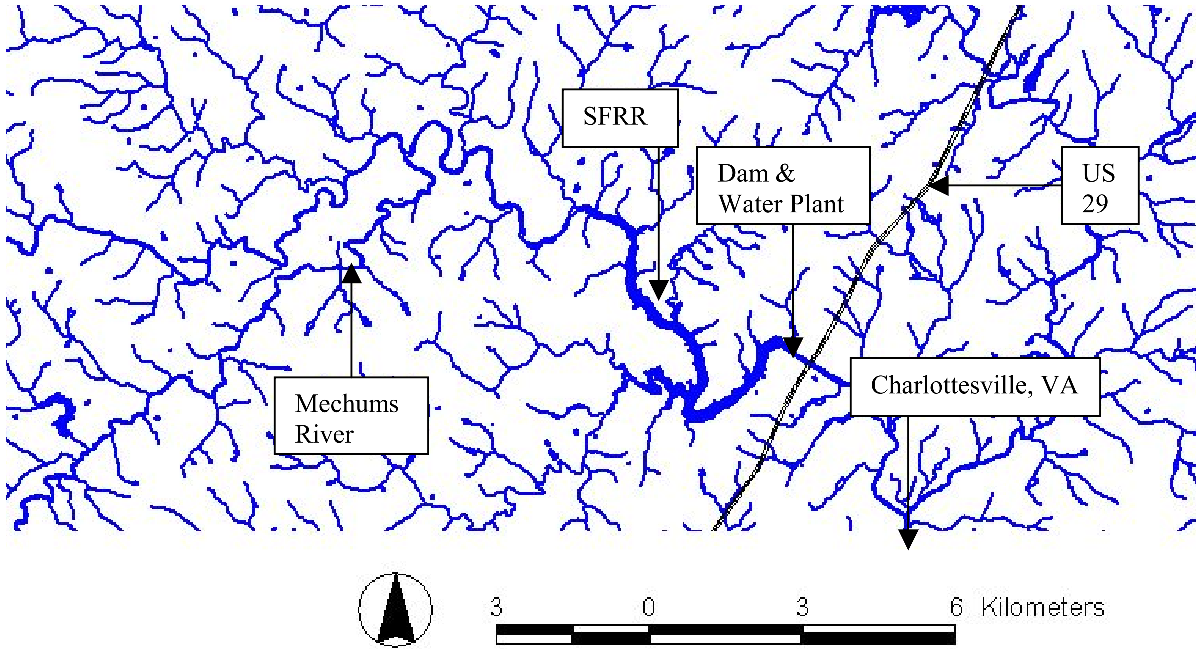



Figure 1. The South Fork Rivanna Reservoir (SFRR) near Charlottesville, VA.

**Table 2.a**. Analysis methods for samples taken in the South Fork Rivanna Reservoir near Charlottesville,

| **Constituent** | **Abbreviation** | **Standard Method*** |
|---|---|---|
| Total phosphorus | TP | 4500-P F (Automated Ascorbic Acid Reduction Method)** |
| Nitrate and Nitrite | $NO_x$ | 4500-$NO_3$-E (“$NO_x$“ Cadmium Reduction Method) |
| Total nitrogen | TN | Sum of 4500-$NO_3$-E (“$NO_x$“ Cadmium Reduction Method) and 4500-$N_{org}$ C (Semi-Micro-Kjeldahl Method) |
| Total Suspended Solids | TSS | 2540 D (Total Suspended Solids Dried at 103-105°C) |
| Chlorophyll-a | Chl-a | 10200 H.2 (Spectrophotometric Determination of Chlorophyll |
| Algal Counts | Algae | 10200 F (Phytoplankton Counting Techniques) |

*( Greenberg et. al. 1992)
** A two reagent method was used before 1992.

**Table 2.b.** Constituents measured in the South Fork Rivanna Reservoir near Charlottesville, VA by year.

| | 76 | **80*** | **81** | **82** | **83** | *84* | *85* | *86* | *87* | 88 | 89 | 90 | 91 | 92 | ***93*** | ***94*** | ***95*** | ***96*** | 99 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| TP^ | | X+ | X | X | X | X | X | X | X | | X | X | X | X | X | X | X | X | X |
| $NO_x$ | X | X | X | X | X | X | X | X | X | | X | X | X | X | X | X | X | X | X |
| TN | | X | X | | | | | | | | X | X | X | X | X | X | | X | X |
| TSS | X | X | X | X | X | X | X | X | X | | X | X | X | X | X | X | X | X | X |
| Chl-a | X | X | X | | X | X | X | | | | X | X | X | X | X | X | X | X | |
| Algae | | | | | X | X | X | X | X | | X | X | X | X | X | X | X | | X |

* Bolding, italics, and underlining indicate year groupings used in analysis.
^ Constituent codes defined above in Table 2.a.
+ “X” indicates constituent was sampled in year noted.

Information on discharge into the SFRR was taken from the U.S. Geological Survey (U.S. Geological Survey 1999). Values from the gage identified as Mechums River at White Hall, VA were used to describe general discharge into the reservoir. Correlations between the Mechums gage and others in the SFRR watershed were very strong and the Mechums gage was the only one still in service. Air temperature data from the Observatory Weather Station in Charlottesville were provided by the Virginia State Climatologist (National Climatic Data Center 2000). The Observatory station was the closest station to the SFRR with a reliable and digitally available record.

### *Statistical Approach*

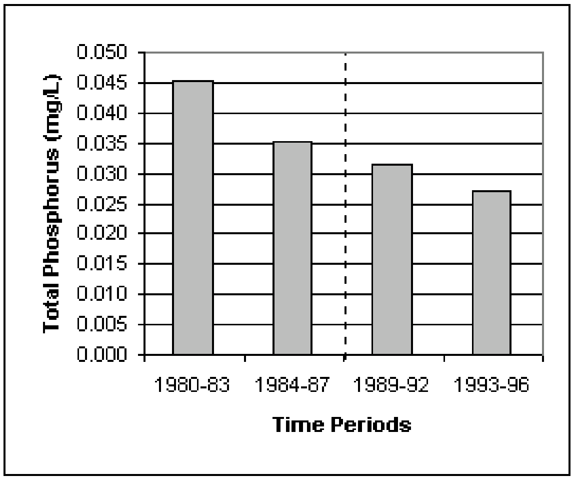
Changes in the SFRR were best addressed by considering four-year time periods (1980-83, 1984-87, 1989-92, and 1993-96) rather than on a year by year basis. Management strategies were applied on a multiple year time scale and year to year variation in climate was dampened by grouping years. The data generally fell into logical four-year groupings (Table 2.b). Two year groups (1980-83 and 1984-87) fell before 1988 and two (1989-92 and 1993-96) fell after. The year 1988 was a logical midpoint as it was the year the sewage interceptor and hydropower plant went on line. It was also a year of no data collection. In a few cases only two year groups were used (1980-87 and 1989-96).

Time period means were compared using an analysis of covariance multiple comparison test with a Bonferroni adjustment (SAS proc mixed, SAS Institute, Inc. 1999). An $\alpha$ of 0.05 was applied. In all cases a Levene test was performed to check the assumption of equal homogeneity of variance (Norusis 1998). Observations were assumed to be independent by nature of being at least two weeks separated in time and spanning multiple years. Dependent variables were transformed to meet the assumption of normality.
The SFRR is quite riverine in nature. The influence of tributary discharge would have been a strong covariate with almost any variable of interest. Therefore, each independent variable was correlated against a large set of descriptors of tributary discharge to the reservoir. The variable with the strongest correlation was used as a continuous covariate in the mixed effect analysis of covariance. The same procedure was performed for precipitation and air temperature variables where appropriate.

## RESULTS

### *Eutrophication Issues*

Mean phosphorus concentrations in the reservoir (near the dam) dropped from 0.045 mg/L in the early years of the study to 0.027 mg/L at the end (Figure 2). The trend appeared to be consistently downward over time. Reductions in concentrations were statistically significant between the 1980-83 years and 1989-92 years ($P = 0.032$) and between the 1980-83 years and the 1993-96 years ($P = 0.001$). The flow in the Mechums River two days before the sampling predicted some of the variation in phosphorus concentration ($P < 0.0001$).

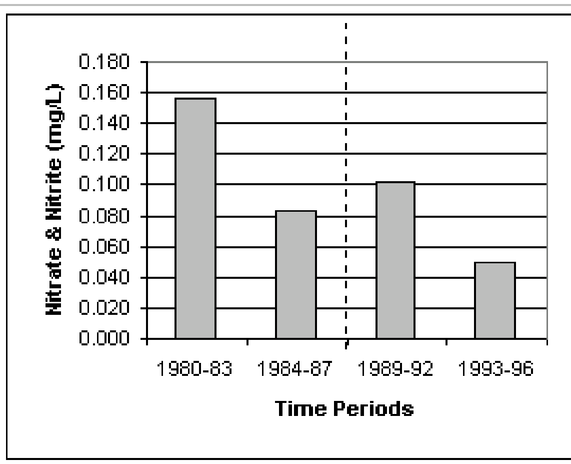



| *P-Values* | 1980-83 | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|---|
| 1980-83 | | *0.207* | ***0.032*** | ***0.001*** |
| 1984-87 | | | *1.000* | *0.171* |
| 1989-92 | | | | *1.000* |
| Mean (mg/L) | 0.045 | 0.035 | 0.031 | 0.027 |
| n | 24 | 29 | 23 | 21 |

| | |
|---|---|
| Transformation: | natural log |
| Covariate(s): | Inflow two days earlier (ln) |
| Covariate sig: | <.0001 |
| Model sig: | 0.001 |

Figure 2. Total phosphorus by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance multiple comparison test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-values highlighted in bold.

Mean nitrate/nitrite ($NO_x$) concentrations decreased from 0.157 mg/L at the beginning of the study period to 0.050 mg/L at the end (Figure 3). There was not a consistent downward trend. The reduction in concentration between the 1980-83 and 1993-96 time periods was statistically significant ($P = 0.003$). The mean flow for the ten days prior to the sample predicted for some of the variation in concentration ($P = 0.005$).

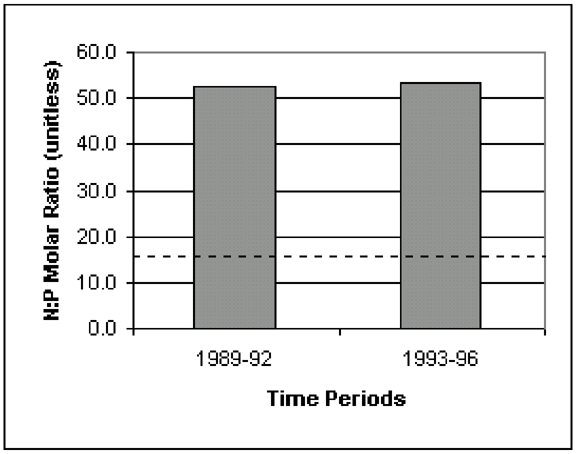



| *P-Values* | 1980-83 | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|---|
| 1980-83 | | *0.271* | *1.000* | ***0.003*** |
| 1984-87 | | | *1.000* | *0.518* |
| 1989-92 | | | | *0.118* |
| Mean (mg/L) | 0.157 | 0.084 | 0.102 | 0.050 |
| n | 17 | 23 | 21 | 21 |

| | |
|---|---|
| Transformation: | natural log (ln) |
| Covariate(s): | Mean inflow for 10 days (ln) |
| Covariate sig: | 0.005 |
| Model sig: | <.0001 |

Figure 3. Nitrate/Nitrite by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance multiple comparison test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-value highlighted in bold.

Data were available to calculate the mean (molar) total nitrogen to total phosphorus ratios for the latter two periods of the study (Figure 4). In both cases the ratio was roughly 53.

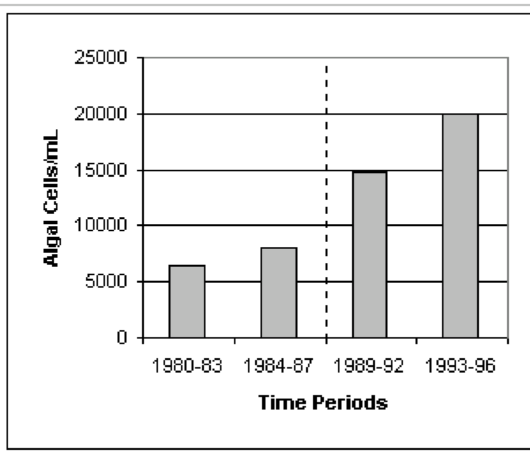



| *P-Value* | 1989-92 | 1993-96 |
|---|---|---|
| 1989-92 | | *0.966* |
| Mean | 52.6 | 53.2 |
| n | 21 | 16 |

Transformation: none
Covariate(s): Mean inflow for year (ln);
Covariate sig: 0.086
Model sig: 0.966

Figure 4. Nitrogen to phosphorus ratios by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance test. Dashed line signifies the Redfield ratio (roughly 16:1) where nitrogen and phosphorus are often in equilibrium.

The mean total number of algal cells appeared to show an increasing trend with time from 6434 cells/ml in the 1980-83 period to 19897 cells/ml in the 1993-96 period (Figure 5). Because of sample size issues, analysis of covariance could be carried out only by lumping data from 1980-87 and data from 1989-96. The increase in algal counts in the later period was statistically significant ($P < 0.0001$). The mean flow in the Mechums River for the year preceding the sample probably predicted some of the variation in algal counts though the covariate was not quite statistically significant ($P = 0.086$).

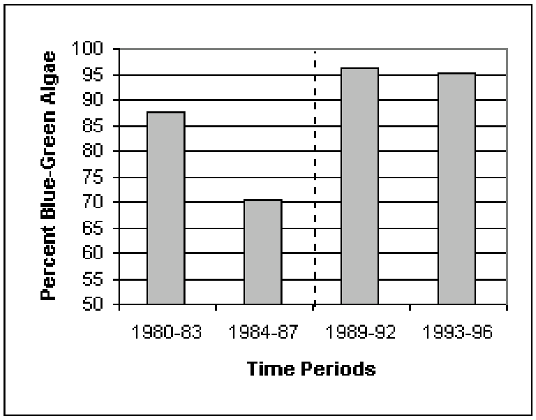



| *P-Value* | 1980-87 | 1989-96 |
|---|---|---|
| 1980-87 | | ***<0.0001*** |
| Mean (Cells/mL) | 7688 | 16440 |
| n | 37 | 31 |

Transformation: natural log (ln)
Covariate(s): Mean inflow for year (ln);
Covariate sig: 0.0856
Model sig: <0.0001
Note: The four period model did not meet the assumption of equal variance.

Figure 5. Algal cells per ml by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-value highlighted in bold.

The mean percentage of the blue-green algae in the reservoir (as compared to diatoms, green algae, and flagellates) ranged from 70% in the 1984-87 period to 96% in the 1989-92 period (Figure 6). As with the algal counts, a comparison of the four time periods could not be performed and a comparison of the two time periods was substituted. The1980-87 period and the 1989-96 periods were significantly different ($P < 0.0001$) in an analysis of covariance. However, the smooth upward trend over time that appeared in the algal counts was not paralleled in the community composition. The flow in the Mechums River for the year preceding the sample predicted some of the variance in the blue-green algae percentage ($P < 0.0001$).

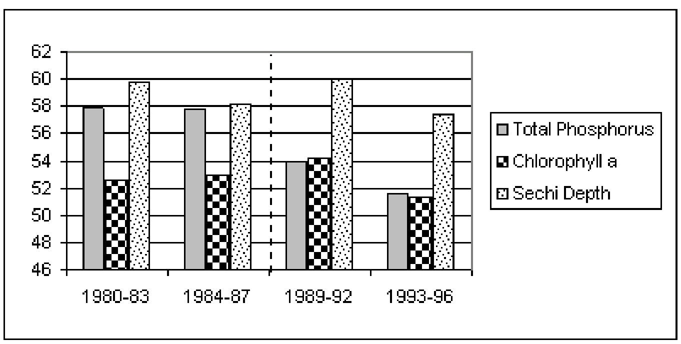



| *P-Value* | 1980-87 | 1989-96 |
|---|---|---|
| 1980-87 | | ***<0.0001*** |
| Mean (%Blue-Green) | 75 | 96 |
| n | 37 | 32 |

Transformation: arc sin
Covariate(s): Mean inflow for year (ln);
Covariate sig: <0.0001
Model sig: <0.0001
Note: The four period model did not meet the assumption of equal variance.

Figure 6. Percent blue-green algae by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-value highlighted in bold.

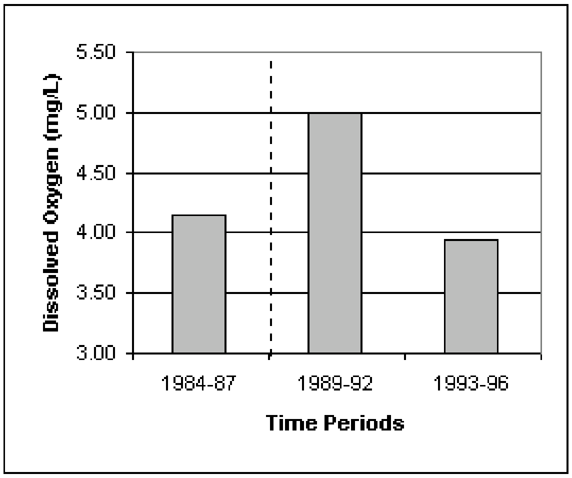
The Carlson Trophic State Index (TSI) calculated by three different methods showed mixed results regarding eutrophication (Figure 7). Scores based on total phosphorus concentrations ranged from 51.6 to 57.9 revealing a consistent downward trend with an apparent drop between the 1984-87 and 1989-92 time periods. Scores based on Chlorophyll a concentrations ranged from 51.3 to 54.2 and showed no apparent trend. Scores based on chlorophyll a were consistently lower than those based on total phosphorus. Scores based on sechi depth ranged from 57.4 and 60.0 and were consistently higher than total phosphorus and chlorophyll a scores.




| *TSI Score* | Total Phosphorus | Chlorophyll a | Sechi Depth |
|---|---|---|---|
| 1980-83 | *57.9* | *52.6* | *59.8* |
| 1984-87 | *57.8* | *52.9* | *58.2* |
| 1989-92 | *54* | *54.2* | *60* |
| 1993-96 | *51.6* | *51.3* | *57.4* |

Figure 7. Mean Carlson Trophic Index scores for the South Fork Rivanna Reservoir near Charlottesville, VA by time period. Scores are based on total phosphorus, chlorophyll a, and sechi depth readings. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant.

Dissolved oxygen data were available for the most recent three time periods (Figure 8). The mean for the water column ranged from 3.95 mg/L in 1993-96 to 4.99 mg/L in 1989-92. A pattern through time did not appear. The highest mean for 1989-92 was significantly different from that of 1993-96 ($P = 0.024$) and nearly significantly different from 1984-87 ($P = 0.079$). The mean flow in the Mechums River for the month preceding the sample predicted some of the variation in oxygen concentration ($P < 0.0001$).



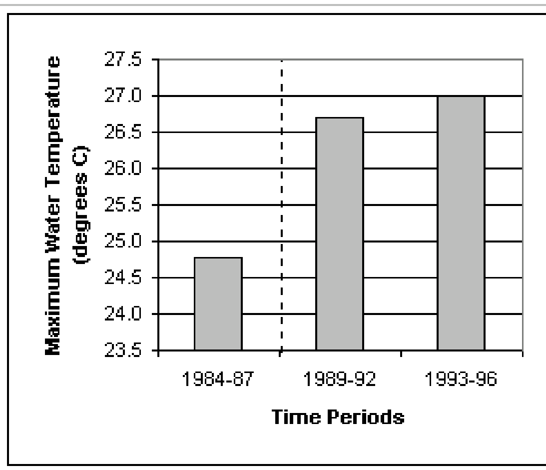

| *P-Values* | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|
| 1984-87 | | 0.079 | 1.000 |
| 1989-92 | | | **0.024** |
| Mean (mg/L) | 4.15 | 4.99 | 3.95 |
| n | 23 | 21 | 19 |

Transformation: none
Covariate(s): Mean inflow for 1 month (ln);
Covariate sig: <0.0001
Model sig: 0.017

Figure 8. Dissolved oxygen in the water column by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance multiple comparison test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-value highlighted in bold.

### *Physical Stratification (Water Temperature)*

Mean maximum temperature in the water column increased over time from 24.8°C in the 1984-87 period to 27.0°C in the 1993-96 period (Figure 9). The 1984-87 period was significantly lower than both the 1989-92 and 1993-96 periods ($P < 0.0001$ in both cases). The flow in the Mechums for the two weeks prior to sampling ($P = 0.039$) and the mean maximum air temperature for the ten days prior to sampling ($P < 0.0001$) explained some of the variance in water temperature.

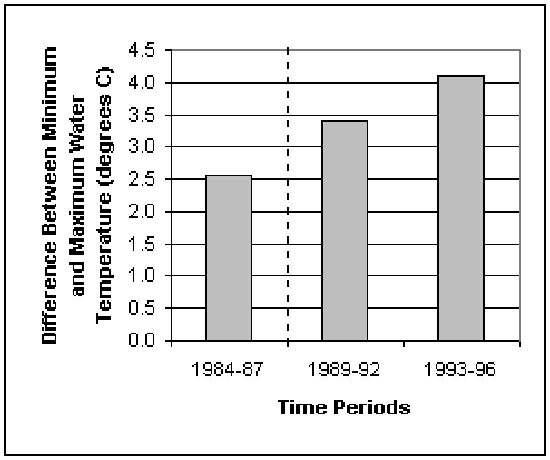



| *P-Values* | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|
| 1984-87 | | ***<0.0001*** | ***<0.0001*** |
| 1989-92 | | | *1.000* |
| Mean (degrees C) | 24.8 | 26.7 | 27.0 |
| n | 23 | 20 | 19 |

Transformation: none
Covariate(s): Mean inflow for 2 weeks (ln); maximum air temperature for 10 days.
Covariate sig: 0.039; <0.0001
Model sig: <0.0001

Figure 9. Maximum water temperature in the water column by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance multiple comparison test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-values highlighted in bold.

The mean difference between the maximum and minimum temperature in the water column increased from 2.6°C in the 1984-87 to 4.1°C 1993-96 (Figure 10). The 1984-87 period was significantly lower than the 1993-96 period ($P = 0.037$). The mean flow in the Mechums River for the two months prior to the sample ($P = 0.033$) and the mean maximum air temperature for the nine months prior to the sample ($P = 0.001$) predicted some of the variation in temperature contrast. The maximum air temperature on the sample date was close to significant as a covariate as well (P = 0.119).



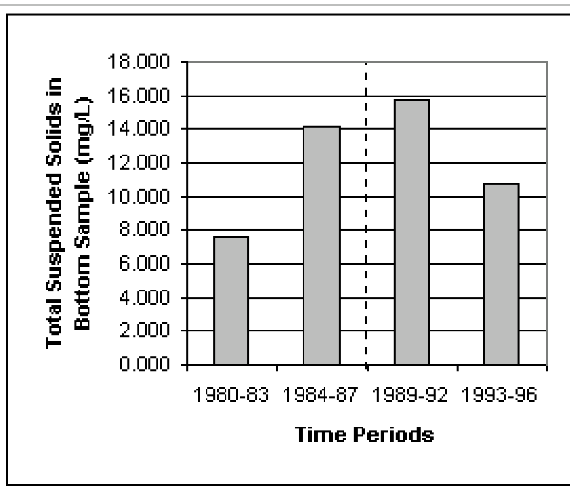

| *P-Values* | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|
| 1984-87 | | *0.419* | ***0.037*** |
| 1989-92 | | | *0.732* |
| Mean (degrees C) | 2.6 | 3.4 | 4.1 |
| n | 23 | 21 | 19 |

| | |
|---|---|
| Transformation: | none |
| Covariate(s): | Mean inflow for 2 months (ln); maximum air temperature for 9 months; maximum air temperature on sample date. |
| Covariate sig: | 0.033; 0.001; 0.119 |
| Model sig: | 0.042 |

Figure 10. Difference between maximum and minimum temperatures in the water column by time period for the South Fork Rivanna Reservoir near Charlottesville, VA with results of analysis of covariance multiple comparison test. Dashed line signifies installation of Crozet sewer interceptor and hydropower plant. Significant P-value highlighted in bold.

### *Sediment Issue*

Mean sediment concentrations in the bottom samples ranged from 7.6 mg/L in 1980-83 to 15.8 mg/L in 1993-96 (Figure 11). There did not appear to be a time trend. The 1980-83 period mean was significantly lower than both the 1984-87 (P = 0.041) and 1989-92 (P = 0.002) periods. The maximum flow in the Mechums River in the 5 days preceding the sample predicted some of the variation in sediment concentration (P = 0.050).




| *P-Values* | 1980-83 | 1984-87 | 1989-92 | 1993-96 |
|---|---|---|---|---|
| 1980-83 | | ***0.041*** | ***0.002*** | *0.922* |
| 1984-87 | | | *1.000* | *1.000* |
| 1989-92 | | | | *0.711* |
| Mean (mg/L) | 7.640 | 14.095 | 15.753 | 10.775 |
| n | 22 | 29 | 21 | 21 |

| | |
|---|---|
| Transformation: | natural log |
| Covariate(s): | Maximum inflow over 5 days |
| Covariate sig: | 0.050 |
| Model sig: | 0.013 |

Figure 11. Suspended sediment concentration in bottom samples take near the dam of the South Fork Rivanna Reservoir near Charlottesville, VA by time period with results of analysis of covariance multiple comparison test. Significant P-values highlighted in bold.

## DISCUSSION

### *Eutrophication Issues*

Phosphorus concentrations in the South Fork Rivanna Reservoir declined over the years. Possibly one cause was the construction of the Crozet sewer interceptor on the Mechums River which removed as much as 25% or more of the phosphorus load from the reservoir watershed by piping it to Charlottesville for treatment and release into the Rivanna River below the dam (Figure 2, Betz Environmental Engineers 1977). The interceptor went online in 1988 and phosphorus levels after 1988 were significantly different from those in the 1980-83 period. An EPA funded effort to install agricultural BMPs in the late 1980s and early 1990s may have helped as well. Other factors could have included the general introduction of the National Pollution Discharge Elimination System and a long term decline in row crop agriculture in the SFRR watershed. Similar factors may have been the sources of possible reductions (and certain stability) in nitrate/nitrite concentrations (Figure 3).

Whatever the cause, nutrient conditions in the reservoir appeared to have improved. With that improvement, particularly in phosphorus which appeared to be limiting relative to nitrogen (Figure 4), one might have expected improvements in typical eutrophication conditions. However, algal cell counts appeared to rise (Figure 5) and the blue-green algae proportion of the algal community increased (Figure 6). Dissolved oxygen conditions, which might have improved had algal bloom and die-off conditions been reduced, showed no sign of improvement (Figure 8).

The discrepancy between downward trends in nutrient conditions and upward trends in algal community suggested a limit on algal growth other than nutrients. A physical as well as a chemical limitation most likely existed. This physical limitation could have been the algal residence time in the reservoir. In riverine systems, there is little time in any one river reach for the phytoplankton population to increase significantly because algae are exported downstream too quickly. The SFRR is a long narrow reservoir with distinct riverine characteristics. This character was highlighted by the power of the Mechums River discharge variables in all of the analyses (though in some cases the Mechums flow data was probably a surrogate for rainfall and runoff closer to the sample site). The overall residence time of SFRR, typically eight days or less, probably limited the algal community (Betz Environmental Engineers 1977).

The change that allowed increased algal growth may have been the installation of the hydropower plant in 1988. The hydraulic character of SFRR appeared to change after that time. While the net residence time may or may not have changed, the residence time for warm surface waters probably increased. Without the plant, the major path out of the reservoir was over the dam, a process that skimmed surface (epilimnionic) water off the reservoir during stratified periods taking the warmer water and algae with it. The hydropower plant intake, resting on the bottom of the reservoir, drew from the cooler hypolimnion below the photic zone occupied by the algae.

The changes in the hydraulic pathways were reflected in changes in the temperature conditions. The maximum temperatures and differences between the coldest and warmest points in the water column both increased after the installation of the hydropower plant (Figures 9 and 10).

Water treatment plant operators observed the changes directly. For example, after a summer storm, cool, sediment-laden water moved along the bottom of the reservoir. If the hydropower plant was operating, much of that water left via the intake to the turbine. Meanwhile, the treatment plant could withdraw water from an intake higher in the water column allowing the plant to avoid the highest turbidity source water. If the algal conditions were not too bad, which they usually were not from a water treatment perspective, there was a significant water treatment benefit to the new hydraulic pathways (Golladay 2000).

The Carlson Trophic State Index (TSI) highlighted the complexity of assessing eutrophication in SFRR (Figure 7, Carlson 1977). The TSI is a mathematical attempt to put common eutrophication measures (total phosphorus, chlorophyll a, and sechi depth) on an even scale. Total phosphorus showed an improvement (lower score). Chorophyll a showed little change (where one would have expected an increase based on other algal measures). Chorophyll a scores were all lower than total phosphorus scores reinforcing the observation that phosphorus was not the limiting factor. Sechi depth scores showed little change. The fact that sechi scores were higher than chlorophyll a and total phosphorus scores indicated the riverine nature of the reservoir. Sechi depths were determined as much by sediment load as algal growth.

Did the reservoir become more or less eutrophic during the period of analysis? From the point of view of a strict algal assessment it did become more eutrophic. However, from the phosphorus perspective significant progress was made. Algal problems probably would have increased more significantly had the phosphorus reductions not occurred. Increased algal populations may have had negative impacts on the reservoir ecosystem and tailwater condition that were not seen in this study and require consideration. However, from a water treatment point of view the hydraulic changes appear to have been positive overall. The water treatment plant operators sometimes have found themselves balancing sediment issues against algal issues. Continued efforts to reduce nutrient loads (and sediment loads) will make their decisions easier and improve the reservoir as a drinking water source and an ecosystem.

***Sediment***

There was no clear pattern to the sediment concentrations in the bottom samples. Bottom samples were analyzed because sediment that had reached that depth was the most likely to be deposited in the reservoir. If that assumption was correct, deposition was quite variable and the rate of deposition did not show a clear time trend (Figure 11). However, this method was not the most appropriate for measuring sediment deposition. The next bathymetric survey will provide a more accurate indication of actual deposition patterns. It also would have been useful to measure sediment (and nutrient) concentrations in the reservoir tributaries so that the loads could be more closely tied to land use and management actions in the watershed.

## REFERENCES

Betz Environmental Engineers, Inc. 1977. Water quality management study of the South Rivanna Reservoir and tributary area.

Black & Veatch. 1995. Urban raw water management plan summary report.

Carlson, R.E. 1977. A trophic state index for lakes. Limnology and Oceanography, 22:2.

Cole, G.A. 1979. Limnology. The C.V. Mosby Company, St. Louis, MO.

F.X. Brown Associates, Inc. and Watershed Management Plan Committee. 1979. South Rivanna Reservoir watershed management plan.

F.X. Browne Associates, Inc. 1982. 208 watershed management study of South Rivanna

Reservoir.

F.X. Browne Associates, Inc. 1986. Evaluation of Lickinghole Creek Basin alternatives.

F.X. Browne Associates, Inc. 1993. Phase II report South Rivanna Reservoir restoration project.

F.X. Browne Associates, Inc. 1999. Evaluation of the Lickinghole Creek Detention Basin.

Golladay, David. 2000 (personal communication). Assistant Manager for Water Operations, Rivanna Water and Sewer Authority. Charlottesville, VA.

Greenberg, A.E., L.S. Clesceri, A.D. Eaton. 1992 (eds.). Standard Methods for the Examination of Water and Wastewater. American Public Health Association. Washington, DC.

Hirschman, D.J. 2000 (personal communication). Water Resource Manager for Albemarle County, VA.

National Climatic Data Center 2000. Data from Official National Oceanic and Atmospheric Administration Cooperative Observer Station Charlottesville 2 W (NCDC ID# 44-1593).

Norusis, M.J. 1998. SPSS 8.0 Guide to Data Analysis. Prentice-Hall, Inc. Upper Saddle River, NJ.

Reid, G.K. 1976. Ecology of Inland Waters and Estuaries. D. Van Nostrand Company, New York, NY.

SAS Institute Inc. 1999. The SAS System for Windows Version 8.

U.S. Geological Survey. 1999. Historical Stream Discharge Data. http://va.water.usgs.gov/.

Wedepohl, R.E., D.R. Knauer, G.B. Wolbert, H. Olem, P.J. Garrison, and K. Kepford. 1990. Monitoring lake and reservoir restoration. EPA 440/4-90-007.

Wilson, Stuart 2000 (personal communication). Laboratory Director,  Rivanna Water and Sewer Authority. Charlottesville, VA.

# Determining Sources of Fecal Pollution in the Blackwater River Watershed

Amy M. Bowman, C. Hagedorn and K. Hix.
Department of Crop and Soil Environmental Sciences
Virginia Polytechnic Institute and State University.
Blacksburg, VA 24060

E-mail: ambowman@vt.edu

## ABSTRACT

Antibiotic resistance analysis (ARA) was used to determine sources of fecal pollution in the Blackwater River in south central Virginia. The Department of Environmental Quality designated seven segments as impaired due to fecal coliforms with non-point source (NPS) agriculture the suspected source of impairment. The Blackwater River watershed encompasses 178,000 acres of dairy, beef, and intensive production agriculture, abundant wildlife populations and many homes with onsite septic systems.

A library of antibiotic resistance profiles based on 30 drug/concentration combinations was developed for 1,451 fecal streptococci isolates from human, cattle, chicken, horse, goat, sheep, deer, raccoon, muskrat, goose, duck, coyote, and wild turkey. Each isolate was classified as human, wildlife or livestock. Correct classification rates were 82.3% for human, 86.2% for livestock and 87.4% for wildlife isolates. Profiles were observed for 48 isolates from each stream sample, collected periodically from August 1999 through August 2000, and compared to the known sources using discriminant analysis. A human signature was found at each site at least once during the year, ranging from 2.1 to 56.2% of the sample isolates. The livestock signature varied from 12.5% to 91.7% over sites and months, and the wildlife signature varied from 4.2% to 70.8%. The results indicate that both humans and wildlife contribute to fecal pollution in addition to the suspected source, livestock, and reducing fecal pollution will require consideration of all three sources. The data from this project are being used to develop a total maximum daily load (TMDL) for fecal coliforms in the Blackwater River.

**Keywords:** water quality, fecal pollution, antibiotic resistance analysis, source tracking

## INTRODUCTION

Fecal bacteria are the major cause of impairments in Virginia's waterways according to the Virginia Department of Environmental Quality (DEQ), with agriculture listed as the primary source of contamination (Virginia DEQ 1998). Fecal coliforms, specifically *Escherichia coli* (*E. coli*), are used as an indicator organism used to demonstrate the potential presence of microbial pathogens and an increased risk of exposure to these pathogens in water. Knowing the sources of fecal pollution is fundamental to the total maximum daily load (TMDL) program currently in place by the EPA. Fecal coliforms, *E. coli*, and other enteric bacteria are present in warm-blooded animals and humans, as well as some cold-blooded animals and birds (Harwood et al. 1999, Alderisio and DeLuca 1999). A simple enumeration of fecal coliforms is not sufficient to implement best management practices for bringing impaired stream segments into compliance, especially in a mixed-use watershed.

Several bacteria source tracking (BST) methods are under development for use in determining sources of fecal pollution in a watershed. Both molecular methods such as ribotyping, pulse-field gel electrophoresis, and randomly amplified polymorphic DNA, and non-molecular methods such as antibiotic resistance analysis (ARA), nutritional patterning, cell wall fatty acid analysis, and strain specific coliphages have been used (and published) in source tracking projects (Hagedorn 2000). ARA has been developed for both fecal streptococci and *E. coli* (Harwood et al. 2000). ARA relies on different antibiotic resistance patterns that can be related to specific sources of fecal pollution. Variations of the ARA method used in this study have been successfully employed by other researchers and in other watersheds (Wiggins et al. 1999, Hagedorn et al. 1999). Benefits of ARA include use of simple laboratory techniques, basic equipment requirements, and can be performed at a relatively low cost compared to some other methods. In addition, high levels of separation have been found comparable to those reported for molecular methods (Dombek et al. 2000).

The Blackwater River watershed contains approximately 72,000 ha of rolling piedmont and mountain topography. The watershed contains abundant wildlife populations, many private septic systems from rural homes and several small communities, and supports an active dairy industry, scattered beef cattle farms, and intensive production agriculture. The upper mountainous areas are predominately forested, whereas most agriculture activities are centered in the lower piedmont region. The Blackwater River supplies drinking water for the Town of Rocky Mount and ultimately flows into Smith Mountain Lake, an 8,000 ha recreational and hydroelectric power generation impoundment. The Blackwater River encompasses numerous stream segments designated as impaired due to high fecal coliform bacterial counts. The Blackwater River is one of Virginia's high priority areas for TMDL development. Monthly compliance monitoring is performed by DEQ at five stations in the Blackwater River watershed. During the 1993-1995 assessment period for the 1996 303(d) list, 4 of the 5 stations greatly exceeded the 10% violation limit used for classification as impaired. For the 1998 303(d) list, a longer assessment period was used (1992-1997), and the list incorporated the results from a five-year 319 funded study, resulting in seven segments being designated as impaired. These segments represent 180.2 stream km, with the suspected source of impairment identified as non-point source (NPS) agriculture.

The goal of this study was to develop and test BST methodology in the Blackwater River watershed, Franklin County, Va (Figure 1, Appendix A). The project objectives were (1) to use ARA as a BST method to develop a profile library of fecal bacteria from known sources in the watershed, and (2) compare profiles of fecal bacteria from stream in the watershed (unknown sources) against the profile library to determine the sources of the fecal bacteria.

Our part of the Blackwater River watershed project is an opportunity that we will use to provide a proof-of-concept for BST methodology. Target audiences for BST results include the many small communities and the agricultural industry in the watershed, but the main target audience is regulatory agencies such as the Department of Conservation and Recreation (DCR), DEQ, and the EPA. We have the potential to provide agencies responsible for water quality and public health with a resource to determine sources of fecal contamination. Until sources of pollution are identified, risk to communities cannot be accurately assessed, and water quality improvements will remain a hit-or-miss affair.

## MATERIALS AND METHODS

### Construction of the Known Source Library

Known source sample collection. Isolates from fifteen different sources were collected in Franklin County, Va, from September 1999 to April 2000 to build a known source library. Sources included dairy cow, chicken, beef cow, horse, goat, deer, raccoon, muskrat, goose, duck, coyote, wild turkey and human. Livestock samples were collected from local farms, wildlife samples were from locations where wildlife had been observed and the scats could be identified, and human sources were collected from septic tank pump-out trucks. Solid fecal samples were placed in sterile Whirlpac bags and liquid manure and septic samples were collected in sterile polystyrene bottles. All samples were placed on ice in coolers for transport to the laboratory.

Isolation of fecal streptococci. Isolation from known sources was done by adding 10.0g of fresh fecal material or 10.0ml of liquid sample to 90ml of sterile distilled water. Three additional 1:10 dilutions were then made. Each dilution was plated on mEnterococcus Agar (BBL) and incubated at 37°C for 48 hours. Individual dark red colonies were transferred to one well containing 0.2 ml of Enterococcosel broth (BBL) in a 96-microwell plate using sterile toothpicks. Plates were incubated for 24 hours at 37°C and confirmation of fecal streptococci was indicated by black Enterococcosel broth, caused by esculin hydrolysis, after incubation.

Antibiotic resistance analysis. Thirty treatments of a combination of nine antibiotics and concentrations were used to determine antibiotic resistance patterns in fecal streptococci. Antibiotic stock solutions were prepared from commercial antibiotic powders (Sigma) as indicated in Table 1. Each of the thirty antibiotic/concentrations was added separately to flasks of autoclaved and cooled Trypticase Soy Agar (TSA, BBL) from the stock solutions to achieve the desired concentration (Table 2), and then poured into sterile 15x100mm petri dishes. Control plates (no antibiotics) were included with each set. Isolates were transferred from the microwell plate using a stainless steel 48-prong replica plater (Sigma). The replica plater was flame-sterilized (95% ethanol) after inoculation of each TSA plate. The inoculant was allowed to soak into the agar and the plates were then incubated for 48 hours at 37°C. Resistance to an antibiotic was determined by comparing each isolate to the growth of that isolate on the control plate. A one (1) was recorded if that isolate grew (a round colony, mostly filled) and a zero (0) was recorded for no growth. This was repeated for each isolate on each of the 30 antibiotic plates.

**Table 1. Concentration and solvent of antibiotic stock solutions**

| Antibiotic | Solvent | Stock Conc. (mg/ml) |
|---|---|---|
| Chlortetracycline | 1N sodium hydroxide | 10 |
| Oxytetracycline | 1:1 water-methanol | 10 |
| Streptomycin | Distilled water | 10 |
| Cephalothin | Distilled water | 10 |
| Erythromycin | 1:1 water-ethanol | 10 |
| Tetracycline | Methanol | 10 |
| Neomycin | Distilled water | 10 |
| Vancomycin | 1:1 water-ethanol | 10 |
| Amoxicillin | 1:1 water methanol | 2.5 |

**Table 2. Concentrations of antibiotic treatments**

| Antibiotic | Plate concentrations (μg/l) |
|---|---|
| Chlortetracycline | 60, 80, 100 |
| Oxytetracycline | 20, 40, 60, 80, 100 |
| Streptomycin | 40, 60, 80, 100 |
| Cephalothin | 10, 15, 30, 50 |
| Erythromycin | 10, 15, 30, 50 |
| Tetracycline | 10, 15, 30, 50, 100 |
| Neomycin | 40, 60, 80 |
| Vancomycin | 2.5 |
| Amoxicillin | 0.625 |

**Analysis of Stream Samples (Isolates of Unknown Origin)**

Collection and isolation of fecal streptococci and fecal coliforms. Stream samples were collected September, October, December 1999 and March, April, June, and August 2000, from sites within the Upper Blackwater River basin (Figure 2, Appendix A). Samples were screened for fecal coliforms using the commercial Colilert system. Positive samples were membrane filtered, placed on mENT (BBL) agar to isolate fecal streptococci and incubated for 48 hours at 37$^{o}$C. Results were recorded as colony-forming units (CFU) per 100 ml. Positive samples were also membrane filtered and placed on mFC (BBL) according to Standard Methods (American Public Health Association 1995) agar to isolate and enumerate fecal coliforms. Results were recorded as colony-forming units (CFU) per 100 ml.

Antibiotic resistance analysis. Dark red colonies isolated on mENT agar (fecal streptococci) were transferred to Enterococcosel broth in microwell plates, plated on antibiotic plates and growth or no growth was recorded as for the known source isolates described above.

**Statistical Analysis**

Data was analyzed by discriminant analysis using JMP-In statistical software (version 3.2.6 for windows, SAS Institute, Inc.). Each known source isolate was placed in human, livestock or wildlife categories to develop the database. Each isolate was compared to the control and also compared to every other isolate in the library. Discriminate analysis assigns a predicted source to each isolate: human, livestock, or wildlife based on the profile of growth on each antibiotic treatment. For each category a percent rate of correct classification was calculated by determining how many of each category were correctly identified. The unknown source stream isolates were then compared to the database to classify them as human, livestock, or wildlife in origin.

## RESULTS AND DISCUSSION

**Monitoring**

Fecal coliform counts were placed into three categories based on colony forming units (CFU) per 100 milliliters. Samples with less than 100 CFU/100 ml were designated as low, samples between 100 and 1000 CFU/100 ml were moderate and samples with fecal coliforms greater than 1000 CFU/100 ml were high. Samples in the high category of greater than 1000 CFU/100 ml exceed the Virginia recreational water use standard. Table 3 shows the number of samples taken during each sampling month and the number of samples in each of the three categories. Of 40 samples taken from September 1999 to August 2000, 3 (7.5%) were low, 22 (55.0%) were moderate and 15 (37.5%) were high. All low samples occurred in Oct. and Dec. 1999 while moderate and high samples occurred over all sampling periods.

**Table 3. Classification of fecal coliform counts into low, moderate and high categories**

| Month | No. of samples | Low | Moderate | High |
|---|---|---|---|---|
| September 1999 | 9 | 0 | 6 | 3 |
| October 1999 | 9 | 1 | 3 | 5 |
| December 1999 | 7 | 2 | 5 | 0 |
| March 2000 | 6 | 0 | 5 | 1 |
| August 2000 | 9 | 0 | 3 | 6 |
| Total | 40 | 3 | 22 | 15 |

*Low = < 100 CFU/100 ml; moderate = 100-1000 CFU/100 ml; high = >1000 CFU/100 ml

Sampling stations were grouped into four categories based on location: North Fork of the Blackwater River, South Fork of the Blackwater River, the upper section of the Blackwater River, and the middle section of the Blackwater, including Teels Creek (Figure 2, Appendix A). Teels Creek is a tributary of the Blackwater River that joins between sites BWR032.32 and BWR045.80. Fecal coliform counts (CFU/100 ml) for each station are shown below (Table 4). Based on the occurrence of high counts, site BNR000.40 on the North Fork, and sites BWR061.20 and BWR045.81 on the Blackwater River accounted for 9 of the

14 samples that exceeded 1000 CFU/100 ml. No high counts were recorded for December 1999 while 10 of the 14 high counts occurred in October 1999, and August 2000 (5 each month).

**Table 4. Fecal coliform counts for North Fork, South Fork, and Blackwater River, CFU/100 ml.**

| Stream | Rivermile | Sept 99 | Oct 99 | Dec 99 | Mar 00 | Aug 00 |
|---|---|---|---|---|---|---|
| N. Fork | BNR009.36 | 260 | 120 | 40 | * | 1,160 |
| | BNR000.40 | 49,000 | 10,000 | 320 | 200 | 3,920 |
| S. Fork | GCR002.44 | 940 | 1,070 | 30 | * | 590 |
| | BSF001.15 | 650 | 10,000 | 280 | 430 | 5,200 |
| Upper BW | BWR061.20 | 1,400 | 29,000 | 530 | * | 3,700 |
| | BWR054.81 | 2,120 | 1,170 | 460 | 700 | 2,400 |
| Middle BW | BWR045.80 | 270 | 760 | 360 | 390 | 840 |
| | BWR032.32 | 250 | 130 | 40 | 10,200 | 760 |
| | TEL001.02 | 940 | 30 | 490 | 370 | 1,850 |

* Sample not taken

**Antibiotic Resistance Analysis**

Known source library. The percentages of correct classification for the three source categories of human, livestock and wildlife are indicated in boldface type (Table 5). The discriminant analysis model compares each isolate to the entire library (1,451 isolates) and fits it into one of the predicted source categories. Because of variation in antibiotic profiles among individual sources, the correct classification is less than 100%. Of 424 human isolates in the library, 349 or 82.3% were identified as human by discriminant analysis. Ten percent, or 43 isolates, were incorrectly placed into the livestock category, and 32 or 7.5% were placed in the wildlife category. Of 646 livestock isolates, 557 or 86.2% were correctly identified. Discriminant analysis correctly identified 333 of 381 wildlife isolates, an 87.4% correct rate.

**Table 5. Antibiotic resistance analysis database of known source isolates**

| Predicted Source | Percent of known source isolates assigned to each predicted source (No. of isolates) | | |
|---|---|---|---|
| | Human | Livestock | Wildlife |
| Human | **82.3** (349) | 7.1 (46) | 2.4 (9) |
| Livestock | 10.1 (43) | **86.2** (557) | 10.2 (39) |
| Wildlife | 7.5 (32) | 6.7 (43) | **87.4** (333) |
| Total | 424 | 646 | 381 |

Source tracking. Up to 48 isolates from each water sample (unknown sources) were classified using antibiotic resistance analysis. The discriminant analysis model places each isolate into one of the three source categories if there is a greater than 50% probability that the isolate fits into that category based on its antibiotic resistance profile. Percentages of isolates in human, livestock and wildlife categories were recorded. The following tables (6-10) show the number of isolates and source tracking for each sample site during that sampling month.

The significant percentage of a source in a stream sample was determined for human, livestock and wildlife by calculating the expected frequency of misclassification using a variation of the method described by Harwood, et al. (2000). Expected frequency of misclassification is the percentage of isolates from a known source that are placed in the wrong predicted source category. A stream sample with less than this percentage of contamination from any source category could be attributed to error in the database. The significant percentage of human was determined by adding the number of livestock and wildlife isolates that were placed in the human predicted source category, then dividing by the total number of livestock and wildlife isolates. The percentage of non-human isolates that were misclassified as human was 5.4%. A stream sample with greater than 5.4% human can be treated as having significant human contamination because 5.4% percent of non-human isolates were misclassified by discriminant analysis as human. Similarly, the percentage of misclassification of livestock was determined to be 10.2% by adding the numbers of livestock isolates that were placed in human and wildlife categories and dividing by the total number of human and wildlife isolates. The significant level of wildlife contamination was calculated at 7.0% by adding the total number of human and livestock isolates misclassified as wildlife and dividing by the total number of human and livestock isolates.

Sites with greater than 5.4% of the isolates from human sources were designated as significant. A site with less than 5.4% has a higher probability of being incorrectly classified. In September 1999, two sites of 9 had more than 5.4% human (Table 6). October 1999 had one site above 5.4% human (Table 7). December 1999 had six of 7 sites with substantial human contamination (range of 6.2% to 41.7%, Table 8). In March 2000, every site sampled had greater than 5.4% human (range of 18.7% to 56.2%, Table 9), and in August 2000, every sample was greater than 5.4% human (range of 17.4% to 51.1%, Table 10). A clear human signature was present at every sampling site at some point during the year and indicates that human pollution is an important source of fecal bacteria at the sampling sites included in this study.

Sites with greater than 10.2% of the isolates from livestock sources were designated significant. A site with less than 10.2% has a higher probability of being incorrectly classified. In September 1999, all 9 sites had more than 10.2% livestock (range of 29.2% to 84.4%, Table 6). October 1999 also had all sites above 10.2% livestock (range of 39.6% to 91.7%, Table 7). December 1999 had every site sampled with substantial livestock contamination (range of 12.5% to 66.7%, Table 8). In March 2000, every site sampled had greater than 10.2% livestock (range of 21.9% to 48.9%, Table 9), and August 2000 had 9 of 9 samples greater than 10.2% livestock (range of 46.8% to 58.7%, Table 10). A clear livestock signature was present at every sampling site at some point during the year and indicates that livestock pollution is also an important source of fecal bacteria (as with human) at the sampling sites included in this study.

Sites with greater than 7.0% of the isolates from wildlife sources were designated as significant. A site with less than 7% has a higher probability of being incorrectly classified. In September 1999, all nine sites had more than 7% wildlife (range of 18.7% to 70.8%, Table 6). October 1999 had eight of nine sites above 7% wildlife (range of 8.7% to 58.3%, Table 7). December 1999 had all 7 sites with significant livestock contamination (range of 37.5% to 70.8%, Table 8). In March 2000, every site sampled had greater than 7% wildlife (range of 21.9% to 52.1%, Table 9), and August 2000 had 9 of 9 samples greater than 7% wildlife (range of 8.5% to 27.7%, Table 10). A clear wildlife signature was present at every sampling site at some point during the year and indicates that wildlife pollution is also an important source of fecal bacteria (as with human and livestock) at the sampling sites included in this study.

**Table 6. Antibiotic resistance analysis of fecal streptococci September 1999**

| Stream: | Rivermile: | No. of isolates: | Human | Livestock | Wildlife |
|---|---|---|---|---|---|
| North Fork: | BNR009.36 | 48 | 14.6 | 66.7 | 18.7 |
| | BNR000.40 | 48 | 2.1 | 29.2 | 68.7 |
| South Fork: | GCR002.44 | 45 | 0 | 84.4 | 15.6 |
| | BSF001.15 | 47 | 4.3 | 66.0 | 29.8 |
| Upper BW: | BWR061.20 | 48 | 18.7 | 39.6 | 41.7 |
| | BWR054.81 | 48 | 4.2 | 43.7 | 52.1 |
| Middle BW: | BWR045.80 | 47 | 0 | 42.5 | 57.4 |
| | BWR032.32 | 47 | 0 | 63.8 | 36.2 |
| | TEL001.02 | 48 | 0 | 84.4 | 70.8 |

**Table 7. Antibiotic resistance analysis of fecal streptococci October 1999**

| Stream: | Rivermile: | No. of isolates: | Human | Livestock | Wildlife |
|---|---|---|---|---|---|
| North Fork: | BNR009.36 | 48 | 4.2 | 91.7 | 4.2 |
| | BNR000.40 | 47 | 10.6 | 48.9 | 40.4 |
| South Fork: | GCR002.44 | 46 | 4.3 | 87.0 | 8.7 |
| | BSF001.15 | 48 | 2.1 | 39.6 | 58.3 |
| Upper BW: | BWR061.20 | 47 | 2.1 | 63.8 | 34.0 |
| | BWR054.81 | 47 | 4.3 | 72.3 | 23.4 |
| Middle BW: | BWR045.80 | 48 | 2.1 | 58.3 | 39.6 |
| | BWR032.32 | 48 | 2.1 | 83.3 | 14.6 |
| | TEL001.02 | 44 | 4.5 | 87.0 | 8.7 |

**Table 8. Antibiotic resistance analysis of fecal streptococci December 1999**

| Stream: | Rivermile: | No. of isolates: | Human | Livestock | Wildlife |
|---|---|---|---|---|---|
| North Fork: | BNR009.36 | * | | | |
| | BNR000.40 | 45 | 17.8 | 40.0 | 42.2 |
| South Fork: | GCR002.44 | * | | | |
| | BSF001.15 | 48 | 41.7 | 16.7 | 41.7 |
| Upper BW: | BWR061.20 | 48 | 6.2 | 22.9 | 70.8 |
| | BWR054.81 | 48 | 18.7 | 31.2 | 50.0 |
| Middle BW: | BWR045.80 | 48 | 18.7 | 43.7 | 37.5 |
| | BWR032.32 | 24 | 29.2 | 12.5 | 58.3 |
| | TEL001.02 | 48 | 4.2 | 66.7 | 29.2 |

* Source tracking not performed – too few isolates

**Table 9. Antibiotic resistance analysis of fecal streptococci March 2000**

| Stream: | Rivermile: | No. of isolates: | Human | Livestock | Wildlife |
|---|---|---|---|---|---|
| North Fork: | BNR009.36 | * | | | |
| | BNR000.40 | 32 | 56.2 | 21.9 | 21.9 |
| South Fork: | GCR002.44 | * | | | |
| | BSF001.15 | 48 | 37.5 | 33.3 | 29.2 |
| Upper BW: | BWR061.20 | * | | | |
| | BWR054.81 | 41 | 43.9 | 31.7 | 24.4 |
| Middle BW: | BWR045.80 | 48 | 22.9 | 31.2 | 45.8 |
| | BWR032.32 | 45 | 20.0 | 48.9 | 31.1 |
| | TEL001.02 | 48 | 18.7 | 29.2 | 52.1 |

* Sample not taken

**Table 10. Antibiotic resistance analysis of fecal streptococci August 2000**

| Stream: | Rivermile: | No. of isolates: | Human | Livestock | Wildlife |
|---|---|---|---|---|---|
| North Fork: | BNR009.36 | 47 | 42.6 | 46.8 | 10.6 |
| | BNR000.40 | 47 | 51.1 | 40.4 | 8.5 |
| South Fork: | GCR002.44 | 47 | 38.3 | 46.8 | 14.9 |
| | BSF001.15 | 47 | 17.4 | 58.7 | 23.9 |
| Upper BW: | BWR061.20 | 47 | 40.4 | 51.1 | 8.5 |
| | BWR054.81 | 47 | 31.9 | 40.4 | 27.7 |
| Middle BW: | BWR045.80 | 46 | 28.9 | 51.1 | 20.0 |
| | BWR032.32 | 46 | 17.4 | 54.3 | 28.3 |
| | TEL001.02 | 47 | 38.3 | 46.8 | 14.9 |

Human, livestock and wildlife categories each dominated the samples at various sites and months. The category with the largest percentage was considered to be dominant for any one sample. In September 1999 (Table 6), livestock dominated approximately half of the samples (5 of 9) and wildlife the others (4 of 9). In October 1999 (Table 7), livestock dominated 8 of 9 samples with wildlife dominating 1 of 9. In December 1999 (Table 8), livestock dominated two samples, and wildlife 4 of 7. One sample was split between wildlife and human at 41.7% each. In March 2000 (Table 9), livestock dominated 1 of 6, wildlife dominated two and human dominated 3 of 6. In August 2000 (Table 10), human dominated 1 of 9 samples, livestock dominated 8 of 9 samples, and wildlife dominated none. Livestock and wildlife clearly predominate as the sources of fecal pollution in the Blackwater River watershed. Human contributed a smaller proportion of contamination during all months except for one site in December 1999 and one site in August 2000.

One site from three sections of the Blackwater River was selected to illustrate changes in source tracking throughout the sampling period. Figures 3, 4 and 5 show the percentage of isolates in human, livestock and wildlife categories for each sampling event at the three selected sites.

Figure 3 (Appendix A) shows source tracking for BNR000.40 on the North Fork of the Blackwater River. The percentage of human increased from 2% in September 1999 to 56% in March, then declined to 40% in August 2000. Livestock percentages ranged from 22% to 49% and varied throughout the sampling period. Wildlife percentages generally declined during the year from 67% in September 1999 to 8.5% in August 2000.

Figure 4 (Appendix A) shows percentage of isolates in the three categories for BSF001.15 on the South Fork of the Blackwater River. The percentage of human isolates was highest during the high flow conditions in December 1999 and March 2000, and was the dominant category during those months. Livestock percentages were dominant and highest during low flow conditions of September and August and ranged from 66% to 17%. The wildlife was the dominant category in October 1999, which was also the peak wildlife percentage for the year.

Figure 5 (Appendix A) shows source tracking for BWR032.32 on the main stem of the Blackwater River. Human isolates were highest in December at 29%, but were not the dominant category for any month. Livestock dominated the samples in September, October, March and August and were lowest in December at 12%. Wildlife was highest in December at nearly 60% of the isolates; this was also the only month when wildlife was the dominant source.

Source tracking comparisons generally cannot be made among sites due to large distances between sites and land use patterns. Samples with a high percentage of human isolates occurred during months of high flows compared to the rest of the sampling period.

**Chemical and Physical Monitoring**

Eight sites in the upper Blackwater River watershed were monitored by Virginia Department of Environmental Quality in August 1999 and monthly from January to August 2000. Temperature, pH, dissolved oxygen, conductivity, and nitrates ($NO_3$-N) were measured (Table 11).

**Table 11. Chemical and Physical Measurements**

| Parameter | Temperature ($^oC$) | pH | Dissolved Oxygen (mg/L) | ($NO_3$-N) (μg/ml) | Conductivity (μmhos/cm) |
|---|---|---|---|---|---|
| Mean | 15.3 | 7.27 | 8.08 | .61 | 74.1 |
| Minimum | .2 | 5.97 | 5.42 | .32 | 8.8 |
| Maximum | 27.2 | 8.68 | 12.9 | 1.13 | 112.7 |

Chemical and physical analysis values were within acceptable ranges. Temperature ranged from 0.2$^o$C in January to 27.2$^o$C in August 1999 with an average of 15.3$^o$C. The pH values averaged 7.27, ranging from 5.97 to 8.68. Dissolved oxygen averaged 8.08 mg/L and ranged from 5.42 to 12.9 mg/L. Nitrate concentrations were low, averaging 0.61 μg/ml with a range of .32-1.13 μg/ml. Conductivity was relatively low, averaging 74.1 μmhos/cm, ranging from 8.8 to 112.7. During January the temperature was too low to measure conductivity.

Stream flows were measured for two sites, one on the North Fork and one on the South Fork, for selected months during the study period in cubic meters per second (Appendix A, Figure 6). Flow was lowest during August 1999, a period of drought for the region. The highest measured flows occurred during March 2000, but the stream was too high during April to take flow measurements. Flow dropped in May, June and July compared to the wet spring, but increased slightly in August due to heavy rain events.

## SUMMARY

The objectives of this research were to apply bacterial source tracking, specifically antibiotic resistance analysis (ARA), to a large watershed and to determine sources of fecal pollution in the impaired watershed for use in a TMDL study. Using ARA, correct category classifications for the known source library were 82.3% for human, 86.2% for livestock sources, and 87.4% for wildlife. Samples from the Blackwater River were found to contain bacterial isolates from livestock, wildlife, and human sources in varying proportions throughout the sampling period. Fecal coliform counts ranged from 30 to 49,000 CFU/100 ml over the course of the study period. Three sites accounted for 9 of 14 samples exceeding 1000 CFU/100 ml and October 1999 and August 2000 accounted for 10 of 14 samples exceeding 100 CFU/100 ml.

ARA has proven to be an effective tool in determining sources of fecal pollution in an impaired watershed. To improve water quality in the impaired stream segments, best management practices (BMPs) will be developed and implemented by regulatory and community officials to reduce fecal loading. The source tracking results presented here will become a critical component at this point in the TMDL process. Reliable identification of fecal pollution sources will ensure that BMP efforts and costs are directed at the correct sources. Before source tracking was made available, implementing BMPs to improve water quality was relative guesswork. Now, with source tracking methodology shown to work in a large watershed, a strong, science-based tool is now available for use in BMP planning and implementation. BST results can also be used as a modeling component to assess the impacts of BMPs and to monitor the reductions and changes in the ratio of fecal bacteria from human, wildlife and livestock sources.

## ACKNOWLEDGEMENTS

This work was supported by a TMDL development grant from the Virginia Department of Conservation and Recreation through MapTech, Inc.

## REFERENCES

Alderisio, K. A. and N. DeLuca. 1999. Seasonal Enumeration of Fecal Coliform Bacteria from the Feces of Ring-Billed Gulls (*Larus delawarensis*) and Canada Geese (*Branta canadensis*). Applied and Environmental Microbiology. Vol. 65, no. 12: 5628-5630.

American Public Health Association. 1995. Standard Methods for the examination of water and wastewater, 19th ed. American Public Health Association, Washington, D.C.

Dombek, Priscilla E., LeeAnn K. Johnson, Sara T. Zimmerley, and Michael J. Sadowsky. 2000. Use of Repetitive DNA Sequences and the PCR To Differentiate *Escherichia coli* Isolates from Human and Animal Sources. Applied and Environmental Microbiology. Vol. 66, no. 6: 2572-2577.

Hagedorn, Charles, Sandra L. Robinson, Jennifer R. Filtz, Sarah M. Grubbs, Theresa A. Angier and Raymond B. Reneau, Jr. 1999. Determining Sources of Fecal Pollution in a Rural Virginia Watershed with Antibiotic Resistance Patterns in Fecal Streptococci. Applied and Environmental Microbiology. Vol 65, no. 12: 5522-5531.

Hagedorn, Charles. 2000. Bacterial Source Tracking (BST): Identifying Sources of Fecal Pollution. [Online.] http://www.bsi.vt.edu/biol_4684/BST/BST.html. [22 September 2000, last date accessed.]

Harwood, Valerie J., Joseph Butler, Danny Parrish, and Victoria Wagner. 1999. Isolation of Fecal Coliform Bacteria from the Diamondback Terrapin (*Malaclemys terrapin centrata*). Applied and Environmental Microbiology. Vol. 65, no. 2: 865-867.

Harwood, Valerie J., John Whitlock, and Victoria Withington. 2000. Classification of Antibiotic Resistance Patterns of Indicator Bacteria by Discriminant Analysis: Use in Predicting the Source of Fecal Contamination in Subtropical Waters. Applied and Environmental Microbiology. Vol. 66, no. 9: 3698-3704

Virginia Department of Environmental Quality. 1998. 1998 305(b) Water Quality Assessment Report. [Online.] http://www.deq.state.va.us/water/98-305b.html. [19 September 2000, last date accessed.]

Wiggins, B. A., R. W. Andrews, R. A Conway, C. L. Corr, E. J. Dobratz, D. P. Doughery, J. R. Eppard, S. R. Knupp, M. C. Limjoco, J. M. Mettenburg, J. M. Rinehardt, J. Sonsino, R. L. Torrijos, and M. E. Zimmerman. 1999. Use of Antibiotic Resistance Analysis To Identify Nonpoint Sources of Fecal Pollution. Applied and Environmental Microbiology. Vol. 65, no. 8: 3483-3486.

## Appendix A. Figures

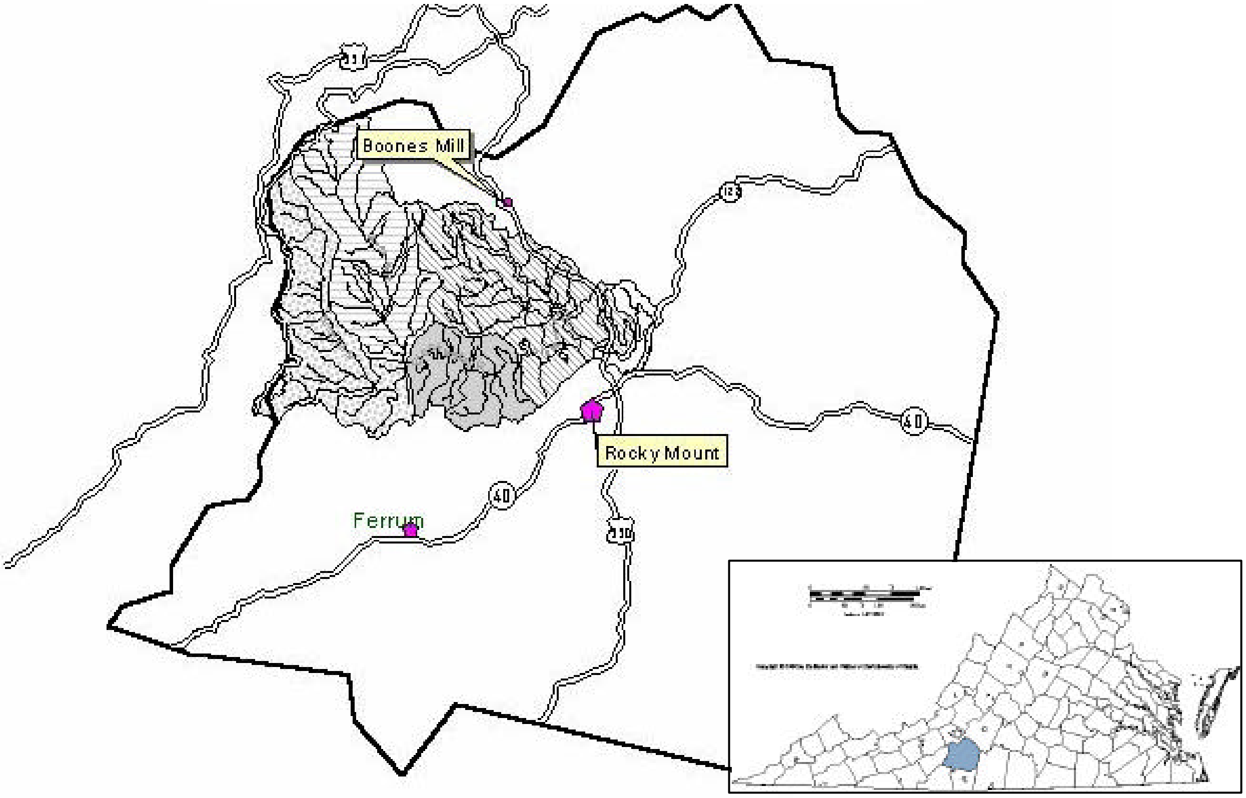



**Figure 1. Franklin County, Virginia; Upper Four Impairments of Blackwater River**

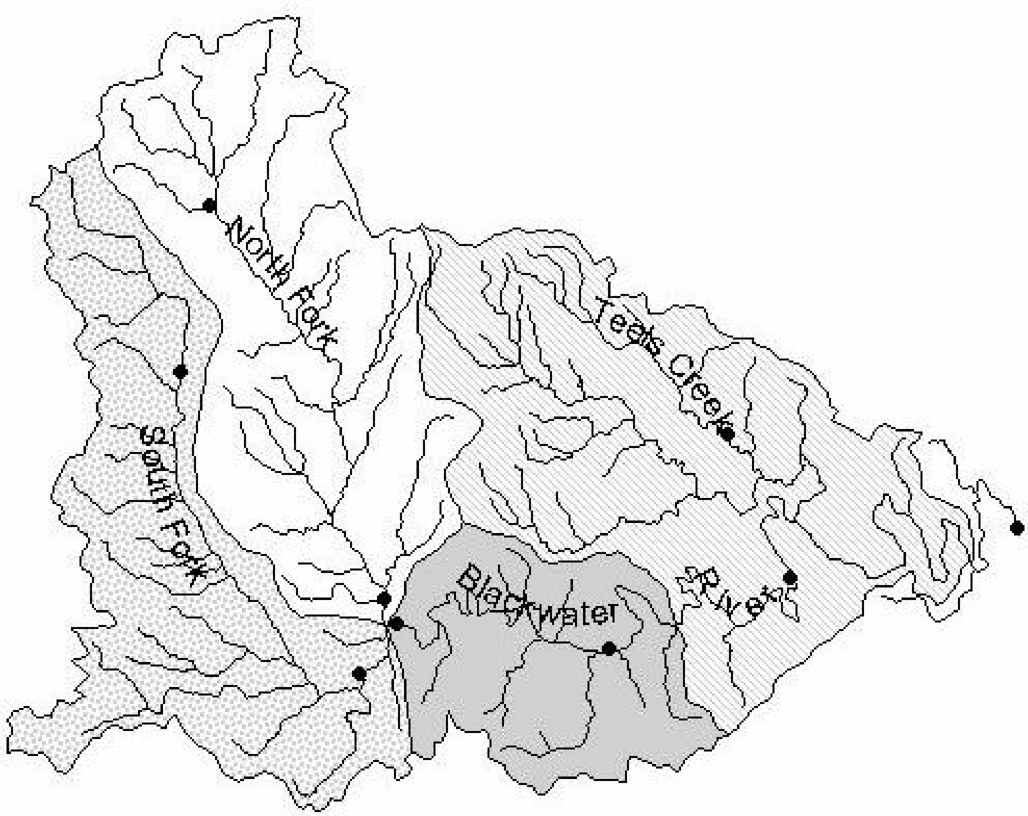



**Figure 2. Upper Four Impairments of Blackwater River with Sampling Sites**

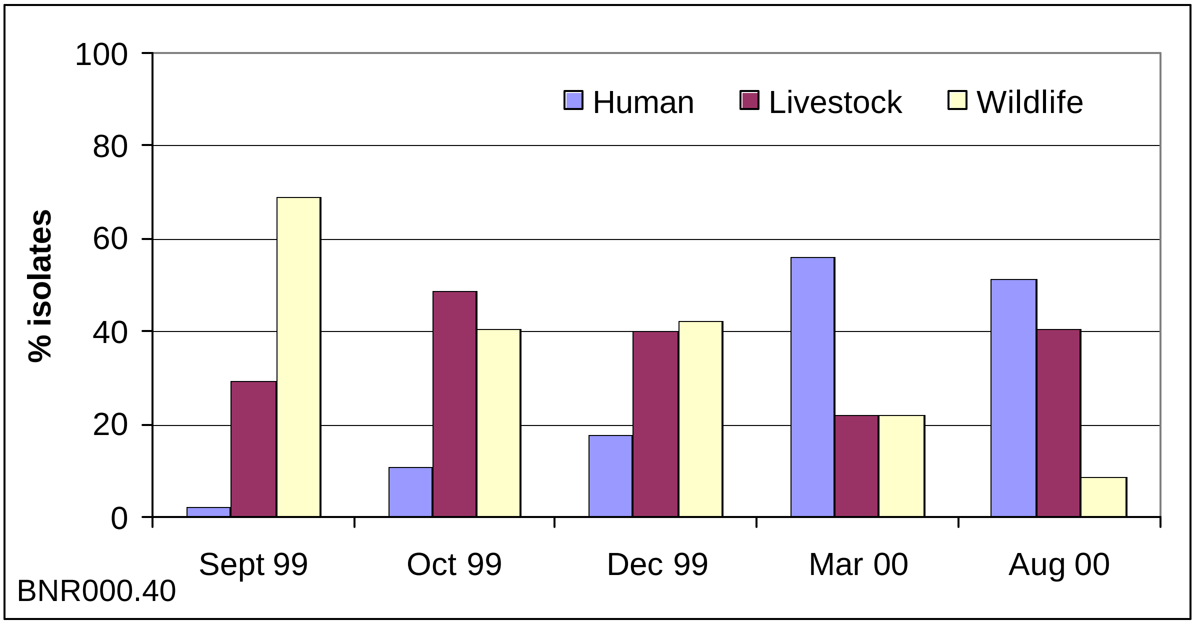



**Figure 3. Percentage of Isolates from North Fork in Human, Livestock, Wildlife Categories**

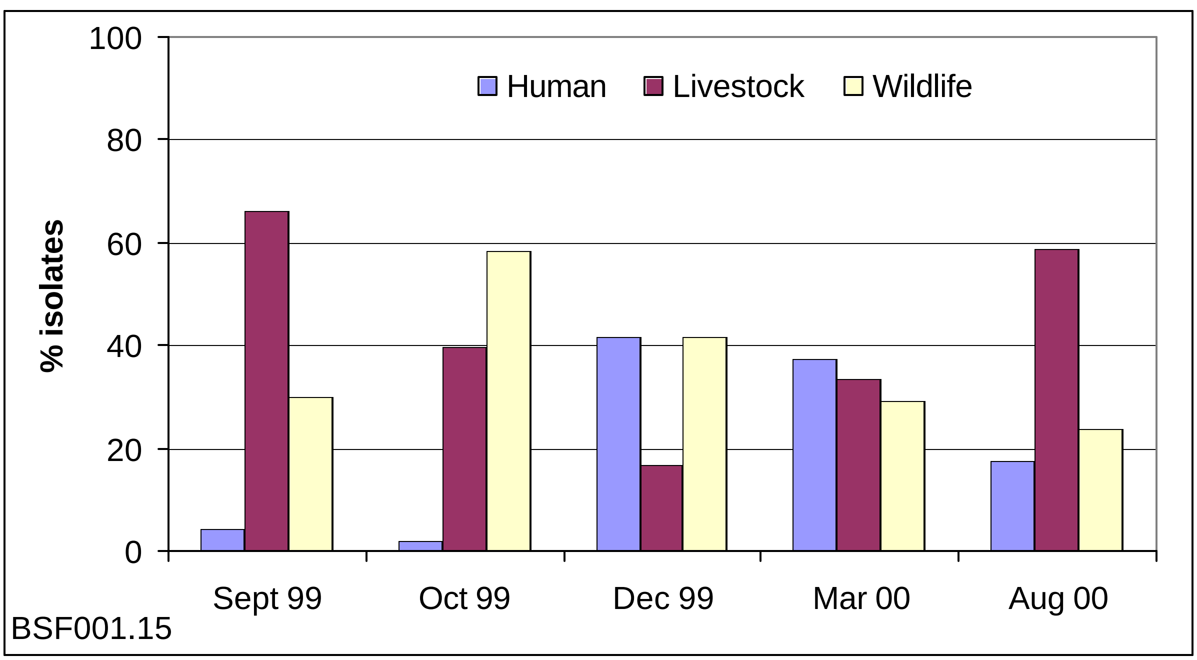



**Figure 4. Percentage of Isolates from South Fork in Human, Livestock, Wildlife Categories**

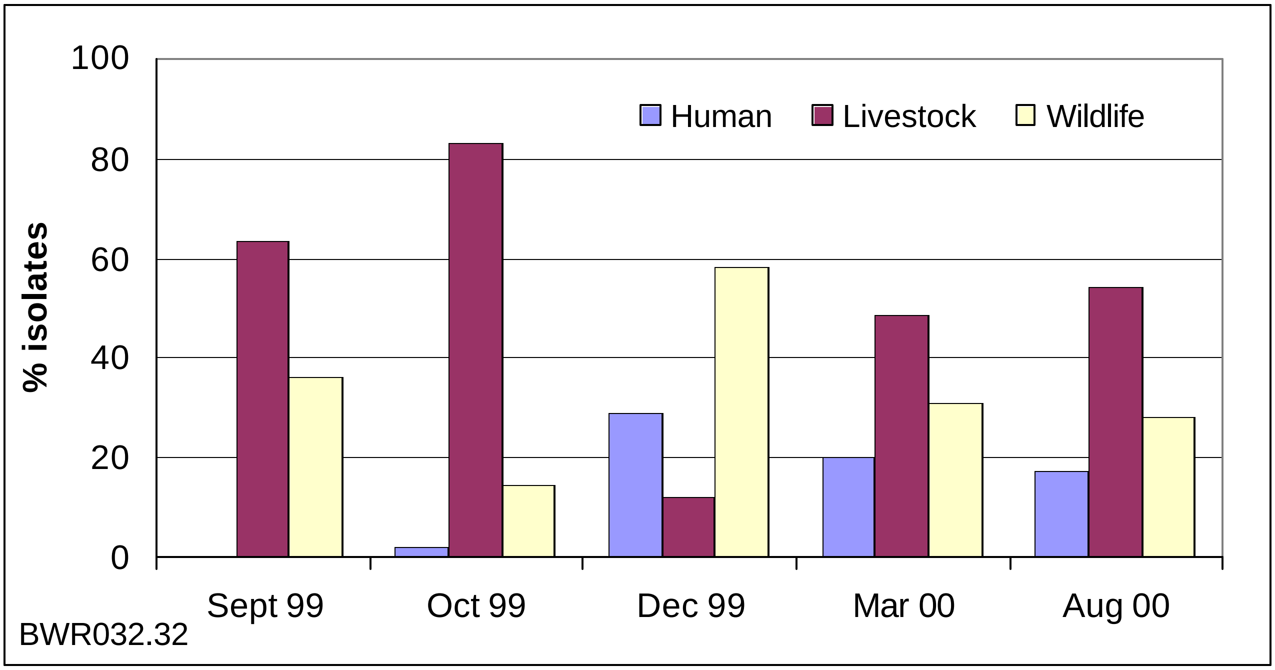



**Figure 5. Percentage of Isolates from Blackwater River in Human, Livestock, Wildlife categories**

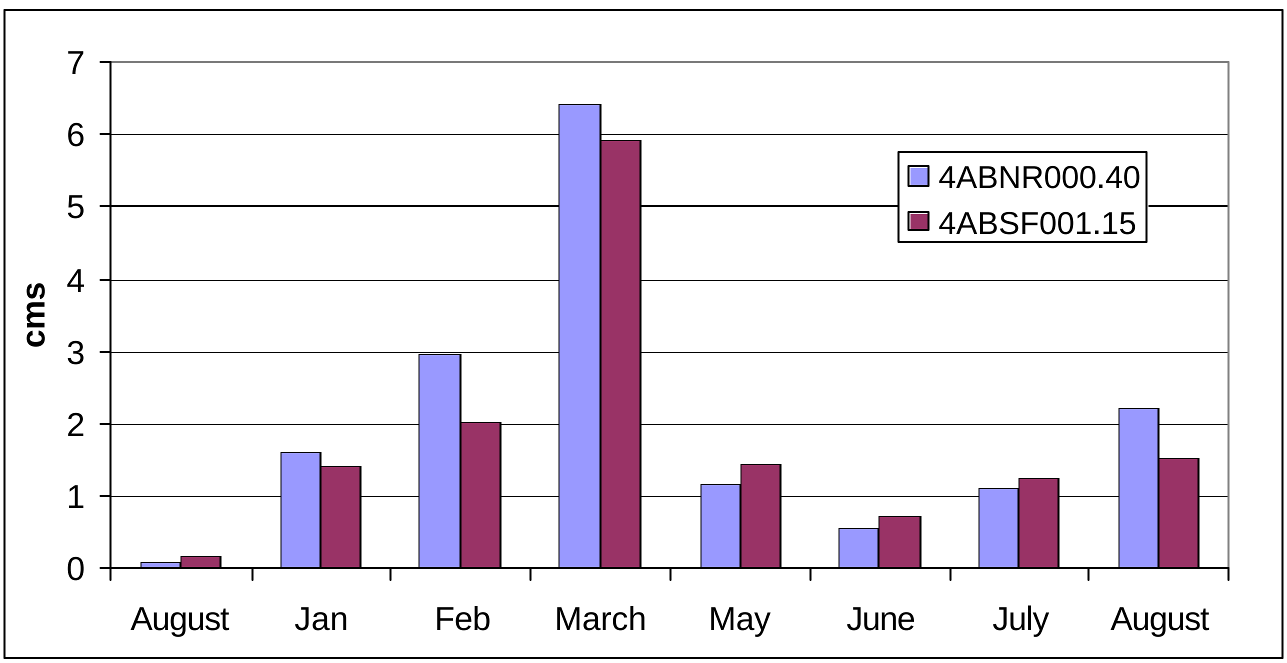



**Figure 6. Flow in $m^3/s$ for North Fork and South Fork August 1999 to August 2000**

# Monitoring the Impacts of Animal Waste BMPs on Nutrient Losses in Runoff from the Owl Run Watershed

K. M. Brannan
Research Associate
Biological Systems Engineering
Virginia Tech
Seitz Hall, Blacksburg, VA 24061
Email: kbrannan@vt.edu

S. Mostaghimi
H. E. and Elizabeth F. Alphin Professor
Biological Systems Engineering
Virginia Tech
Seitz Hall, Blacksburg, VA 24061
Email: smostagh@vt.edu

P. W. McClellan
President
MapTech, Inc.
Blacksburg, VA 24061
Email: pmcclellan@maptech-inc.com

## ABSTRACT

The results of a 10-year study conducted in the Owl Run watershed indicate the beneficial impacts of animal waste BMPs on surface water quality. The main objective of the study was to determine the effectiveness of a system of animal waste BMPs for improving surface water quality. Precipitation, stream flow, suspended solids, nitrogen (N), and phosphorus (P) were measured at the main outlet and at various locations in the Owl Run watershed. Stream flow and meteorological instrumentation installed in the watershed included 4 automatic water samplers, 8 recording rain gauges and a weather station. Pre- and post-BMP comparisons were performed for annual and monthly water quality parameters.

Reductions in all forms of N and most forms of P were observed due to the implementation of BMPs. The BMPs reduced average annual flow-weighted concentrations of total N by 49% and total P by 64% at the main outlet of the watershed. Analysis of average monthly values for flow-weighted concentrations revealed some periods of the year where BMPs were less effective in reducing N and P. In particular, increases in flow-weighted concentrations of nitrate-N were observed for some summer months at the main watershed outlet. Furthermore, the BMPs may have exacerbated problems associated with orthophosphorus-P. Increases in orthophosphorus were observed during several months of the year at the main watershed outlet. The system of BMPs implemented was effective in reducing nutrient levels in the streams of the Owl Run watershed, especially N levels. However, when P is the main water quality concern, BMPs that specifically address the management of excess P should be considered.

**Keywords**: Nonpoint Source Pollution, Animal Waste, Nutrient Management, Water Quality Monitoring

## INTRODUCTION

Agricultural activities are being increasingly blamed for deterioration of surface and ground water resources in the United States. Significant progress has been made in developing technologies for controlling point sources, while pollution from nonpoint sources, especially from agriculture, have been relatively neglected. Nonpoint source pollutants are transported primarily by runoff from urban, agricultural, mining areas, and construction sites. Runoff carries sediment, organic matter, bacteria, pesticides, metals, nutrients, and other chemicals. Nutrients, primarily nitrogen (N) and phosphorus (P), can be a major problem because they can cause eutrophic algae growth that may reduce oxygen availability and increase turbidity in water bodies. Livestock systems, which utilize pastureland for grazing animals and cropland for disposal of manure waste, are one segment of agricultural production for which the extent of nonpoint source pollution is neither clearly defined nor the effectiveness of best management practices (BMPs) adequately demonstrated.

In 1983, a study by the U.S. Environmental Protection Agency on the decline of the Chesapeake Bay indicated that point and nonpoint sources of pollution were among the main causes of the Bay's decline (USEPA, 1983). In particular, the study indicated that, nonpoint sources contributed about 67% of the nitrogen (N) and 39% of the phosphorus (P) entering the Bay. Furthermore, agriculture was estimated to be responsible for 60 and 27 percent of the N and P loadings from nonpoint sources, respectively. Consequently, in December 1987, the Governors of Pennsylvania, Maryland, and Virginia, the mayor of the District of Columbia, and the Administrator of the EPA pledged to address nonpoint source as well as other sources of pollution to restore and protect the Chesapeake Bay. This commitment, known as the Chesapeake Bay Agreement of 1987, required the signatory States to implement cost-sharing programs targeted at reducing nonpoint source pollution of the Bay and its tributaries.

Virginia's agricultural cost-sharing program was initiated in response to the Chesapeake Bay program and was designed to encourage voluntary application of BMPs such as conservation tillage and installation of animal waste storage facilities by farmers. Animal waste management involves both storage and proper application of waste as fertilizer on agricultural land in order to improve crop production and reduce transport of pollutants by runoff. Containment of animal wastes and its land application at the proper time can reduce P in surface runoff by 50 – 70% (USEPA, 1983). Timing, method, and rate of application are controllable management factors that influence both the effectiveness of the animal waste as a fertilizer and the degree to which runoff pollution is prevented (Sharpley et al., 1994).

While there is documented evidence on the water quality advantages of BMPs on field-size plots, the effectiveness of BMPs, especially animal waste management practices, on large watersheds with varying topography, land use, soils, and geology is relatively unknown. Thus, a comprehensive nonpoint source monitoring program was undertaken in 1986 to quantify the effects of animal waste BMPs on improving runoff water quality from the Owl Run watershed located in Fauquier County, Virginia. The objective of the study was to evaluate short-term and long-term effectiveness of animal waste BMPs in improving quality of surface runoff.

## METHODS

A pre versus post-BMP monitoring design (Spooner et al., 1985) was used for the Owl Run watershed project. The total duration of the monitoring project was ten years. The pre-BMP period started at the initialization of monitoring in 1986 and continued until 1989. This resulted in approximately three years of data for the pre-BMP period. The post-BMP period began on July 1, 1989 and continued until the end of the monitoring project (July 1, 1996), resulting in seven years of data.

### *Watershed Description*

The Owl Run watershed, which is 1,153 ha in size, is located in Fauquier County, Virginia. This watershed was selected for monitoring because of its high concentration of dairy farms and lack of animal waste management practices, at the commencement of the project. Erosion from the agricultural land was mainly attributed to the fact that a large area of cropland had to be left without protective vegetative cover over the winter to provide land for manure spreading (Mostaghimi et. al., 1999).

The climate of the Owl Run watershed is of the humid continental type with hot humid summers and mild winters. The average annual rainfall for Owl Run watershed is 1,000 mm. Precipitation is fairly well distributed throughout the year, although the greatest amount occurs in the spring and summer. Because of the characteristics of the summer rains, runoff is usually greater during summer period (Mostaghimi et. al., 1999).

The topography of the Owl Run watershed is consistent with both the Blue Ridge Mountains and Piedmont physiographic provinces (Mostaghimi et. al., 1999). The Piedmont Plateau comprises 80% of Fauquier County and is sub-divided into rolling to steep Piedmont plateau, undulating to rolling Triassic Plane of the Piedmont Plateau, and undulating to rolling Piedmont Plateau. The Blue Ridge Mountain Province in the northwest part of the county has terrain that is mainly steep and rugged.

Soils within the watershed are mostly shallow (0.3-0.6 m) silt loams, overlying Triassic shale. Approximately 72% of the soil series within the watershed are comprised of the Penn (40%, Ultic Hapludalfs), Bucks (16%, Ultic Hapludalts) and Montalto (16%, Ultic Hapludalfs) associations (SCS, 1956). The Penn soils are derived from Triassic Red shale and sandstone; specifically, the silt loam is from the shale and the loam is from the sandstone. These soils are shallow and excessively drained, occurring mainly in undulating to rolling relief. The Penn associations typically have medium runoff and medium to rapid internal drainage. Soil permeability is moderate while the water holding capacity is poor. The Bucks soil series, developed over Triassic red shale and sandstone, are moderately deep, well-drained upland soils. The Bucks typically occupy moderately large areas on broad, undulating ridges that join the Penn soils. The Bucks soils have slow to medium runoff with medium internal drainage. The Montalto soils, developed over fine grained Triassic diabase, are moderately shallow and well-drained. These soils have a medium runoff potential and internal drainage (SCS, 1956).

Nearly 70% of the Owl Run watershed is in agricultural production, including both cropping and livestock productions. The remainder of the watershed includes residential, commercial, transportation and forested areas. Corn production occupies approximately 26% of the watershed area, employing both conventional and no-tillage practices. Roughly half of the corn crop follows a rye cover or small grain rotation. The majority of hay fields remain as grass for three to four years followed by one year of corn planting. Typically, the corn crop is followed by grass legume hay, seeded with a small grain companion crop (Mostaghimi et al., 1999). Five major dairy and several beef operations were functional within the Owl Run watershed during the monitoring project and livestock numbers increased by 2% during the post-BMP period. This increase occurred only in the dairy operations, as the beef operations experienced a slight decline in numbers over the course of the investigation.

Prior to the construction of the animal waste storage facilities, farmers regularly land-applied livestock waste materials, regardless of the ground temperature. Researchers (Crane et al., 1983; Eghball and Power, 1994) have reported on the risks associated with this application schedule and suggest that storing waste material is an efficient means by which to reduce the potential for nonpoint source pollution. The selection of fields to receive waste applications prior to the implementation of the waste storage facilities was based primarily on the proximity during the pre-BMP period. The construction of waste storage facilities within the Owl Run watershed impacted the selection of the fields receiving manure applications in that land operators had the flexibility of applying animal waste to fields where it could be best utilized, rather than those fields most readily accessed. Personnel from the Virginia Division of Soil and Water Conservation developed nutrient management plans for each of the dairy operations. Technical support for the land-operators concerning these management methods was available through the local USDA NRCS office. Stream fencing was also constructed as a BMP throughout Owl Run in areas where animals had direct access to streams. A summary of the BMPs implemented in the Owl Run watershed is presented in Table 1.

***Monitoring System and Data Collection***

The Owl Run study was designed as a pre versus post-BMP implementation data collection and analysis (Spooner et al., 1985). The pre-BMP monitoring of the Owl Run watershed began in the spring of 1986. Approximately three years later, BMP implementation began and monitoring continued through June 1996. Thus, the 10 years of monitoring includes both the pre- and post-BMP data collection periods. Specific elements of the monitoring system included: wet and dry weather physical and chemical monitoring of surface and ground water; biological monitoring of surface water; physical and chemical analysis of soils; and chemical analysis of atmospheric deposition (Mostaghimi et al., 1999). The focus of the chemical component of surface runoff was nutrients, including both soluble and particulate forms.

Four surface runoff monitoring sites were established within the Owl Run watershed (Figure 1). Six pre-existing private drinking-water wells were employed to monitor ground water quality. However, after 12 months of data collection, the ground water monitoring was discontinued, as no water quality impairments were detected. Station QOA was at the outlet of the watershed and the data collected at this site was intended to illustrate the overall response of the entire 1,153 ha watershed to the implementation of the BMPs. The runoff collected at site QOB (45 ha) was installed as a means by which to exclude the urban runoff from the town of Calverton, Virginia. QOC collected mainly agricultural runoff and was installed to demonstrate the efficacy of cropland BMPs in reducing nonpoint source pollution throughout the 462 ha subwatershed. The fourth station, QOD, drained 331 ha of land, including runoff from two of the five dairy operations within the watershed. This monitoring station was intended to evaluate the effectiveness of intensive animal waste BMP implementations on stream water quality (Mostaghimi et al., 1999). During dry weather periods, there was no baseflow, thus the stream flow measured was primarily due to storm events.

Stream flow and meteorological instrumentation installed in the watershed included: 4 automatic water samplers; 8 recording rain gauges; meteorological instruments to record pan evaporation; ambient temperature; humidity; solar radiation, wind speed and wind direction, and a precipitation water quality sampler. Stream water quality samples were collected based on changes in the stream water level and automatic samplers, installed at all stations, were controlled by 21x microloggers (Mostaghimi et al., 1999). The particular monitoring data used in the study reported herein were precipitation, stream flow volume, total suspended solids (TSS), ammonium-N, total Kjeldahl N, filtered total Kjeldahl N, nitrate-N, total P, filtered total P, and orthophosphorus-P.

For all of the data categories, data sets were generated at a monthly time-step. Hourly data were accumulated to generate the monthly data. Sub-sets of monthly data sets were assembled for both the pre- and post-BMP periods. The cutoff date used to separate the pre and post-BMP periods was July 1, 1989. The flow-weighted concentrations were generated by dividing the monthly mass loads by the corresponding monthly accumulated stream flow. The average annual stream flow for each period were calculated by accumulating the monthly values for each period and dividing these sums by the ratio of the number of months in the period divided by 12. The average annual flow-weighted concentrations observed at the watershed outlet (QOA) are presented here. Reductions in average annual values for the other stations are not discussed in this paper and the reader is referred to Brannan et al. (2000) and Mostaghimi et al. (1999) for detailed discussions of annual values observed at the stations QOC and QOD. The contributions from subwatershed QOB were removed from the loads, and hence the flow-weighted concentrations reported for QOA, since no BMPs were implemented in the QOB subwatershed during the study period.

## RESULTS AND DISCUSSION

### *Precipitation and Runoff*

The average annual precipitation amounts for the Owl Run watershed during the pre- and the post-BMP periods were 1,054 and 1,075 mm, respectively (Table 2), indicating a slight increase (2%) in precipitation for the post-BMP period. These rainfall values are the average Theissen-weight values based on eight stations distributed across the Owl Run watershed (Figure 1). The long-term annual average rainfall for the Owl Run watershed is 1000 mm (Mostaghimi et al., 1999), indicating that precipitation during both the pre- and post-BMP periods were slightly above the long-term average. There was little difference in the average monthly rainfall amounts during the pre and post-BMP periods, as can be seen in Figure 2. Differences greater than 20 mm were observed for five of the months (March, May, July, October and November); the majority of these differences occurred during the post-BMP period. The slight increase in precipitation amount observed in the post-BMP period may result from the longer period of data collection for this period. However, the small increase in rainfall amount (2% in the average annual precipitation depth during the post-BMP period) is not expected to mask or bias the impact of the BMPs on the water quality of the Owl Run watershed.

Average annual stream flows (stream flow volume divided by watershed area) for the entire watershed (excluding the residential area measured at station QOB) during the pre and post-BMP periods were 345 mm and 440 mm, respectively (Table 2). Compared to the 2% increase in rainfall depth, the increase in stream flow at QOA during the post-BMP period was considerably higher (28%). The average annual stream flow discharged from the watershed was similar to the average annual values reported for the Piedmont region. Darling (1962) reported an average annual discharge of 410 mm/yr for the Piedmont region. The Owl Run watershed has a discharge similar to the regional average (the pre-BMP 345 mm and post-BMP 440 mm). Differences in the average monthly stream flows between the pre and post-BMP periods were consistent across all subwatersheds. The average monthly stream flow in watershed equivalent depth (stream flow volume divided by watershed area) observed at stations OQA are presented in Figures 3. For the months of January, March, and December, the stream flow tended to be much larger in the post-BMP period and for the months of May and June the pre-BMP values tended to be greater than the post-BMP stream flow (see Figure 3). The differences between the pre and post-BMP stream flow are the result of differences in precipitation amounts. There were no BMPs implemented at a significant level that would impact stream flow volume. Also, the elevated stream flow amounts observed in the post-BMP period could dilute nutrient concentrations in the streams. However, runoff is the main transport mechanism for pollutants in the Owl Run watershed and increased stream flow also results in increased generation and transport of pollutants. Therefore, the dilution effect is considered negligible with respect to the flow-weight concentrations observed during the post-BMP period.

***Nitrogen***

Reductions in monthly flow-weighted N concentrations were observed at watershed outlet due to implementation of BMPs. The changes in average annual flow-weighted concentrations observed at QOA are listed in Table 2. Reductions were observed in total N and in all the other forms of N investigated. There was a 49% reduction in the annual average flow-weighted concentrations for total N during post-BMP period at QOA, eventhough there was a 28% increase in stream flow for the post-BMP period as compared to the pre-BMP phase. Similar reductions were observed for the other forms of N at QOA. In comparison to the pre-BMP values, the average annual ammonium-N concentration was 57% less in the post-BMP period. Likewise, BMPs reduced the annual average flow-weighted concentrations of nitrate-N by 35%. BMPs were also effective in reducing particulate-N. The average annual flow-weighted concentrations of particulate-N decreased by 51% (Table 2). For the aggregated soluble forms of N, the average annual flow-weighted concentrations decreased by 48% after BMP implementation. The average annual flow-weighted concentrations of soluble organic N decreased by 62% in the post-BMP period as well. The general trend in N reductions for the entire Owl Run watershed in the post-BMP period is partially due to N losses during storage of the manure (Brannan et al., 2000). Organic N is converted to inorganic forms of N while manure is stored. The inorganic forms are prone to conversion to ammonia, which is then volatilized to the atmosphere.

The monthly distribution of flow-weighted N concentrations generally exhibits reduction in the post BMP period as observed in the average annual values. The average monthly flow-weighted concentrations for total N observed at QOA are shown in Figure 4. There was a consistent reduction in the flow-weighted total N concentration in the post-BMP period throughout the year (Figure 4). The greatest reduction in total N flow-weighted concentrations occurred in the late summer and fall. Smaller reductions were observed during the winter and spring, but the concentrations were low during both pre and post-BMP periods during these seasons, as compared to the concentrations during the late summer and fall. The large reductions in total N for the non-growing season are the result of the storage of manure and implementation of nutrient management plans. The manure is being applied to fields when and where the nutrients can be used by crops rather than applied where is most convenient, as necessitated during the pre-BMP period when no storage was available. Smaller reductions were observed at station QOA as compared to the reductions observed in total N flow-weighted concentrations at stations QOC and QOD (Data not shown). This may be due to N sources present in the lower portions of the Owl Run watershed not addressed by the BMPs. The effect of N originating from the residential areas around Calverton was removed by subtracting the loads and flow observed at station QOB from those observed at QOA. Other sources in the lower portion of the Owl Run watershed, such as the accumulation of N in-stream, may have masked the reductions observed in the upland watersheds. This is a problem inherent to watershed scale studies and is why multiple locations (especially upland watersheds) must be monitored in addition to the watershed outlet when investigating BMP impacts on water quality.

The reductions observed in total N were also observed for other forms of N during the post-BMP period. The average monthly flow-weighted concentrations for nitrate-N observed at station QOA are show in Figure 5. Reductions were observed in other forms of N but are not discussed in this paper in the interest of brevity. The reader is referred to Brannan et al. (2000) for a more detailed discussion of the reductions in other forms of N. Reduction in the post-BMP period were observed at station QOA for nitrate-N flow-weighted concentrations for all months, except for the month of June. The largest reductions for nitrate-N were observed in the late summer and fall.

The general trend in N reductions in the Owl Run watershed is partially related to the N losses due to the storage of the manure. The two main components of N found in manure are organic N and ammonia-N (Collins et al., 1995). The inorganic portion of N in fresh manure is commonly in the form of ammonium-N. Storage of manure, especially in slurry form, generally results in the loss of organic N through ammonification and then volitilization of the ammonia-N. Also, storage of manure at high moisture contents may result in the loss of nitrate-N by denitrification (Cabrera and Gordillo, 1995). However, the level of nitrate-N in manure depends on the presence of nitrifiers, which are microbes commonly found in the soil. The transformation of organic to inorganic forms of N is evident in the data collected from the Owl Run watershed. There was a dramatic change in the proportions of ammonium, nitrate, soluble organic, and particulate N from the pre to the post-BMP period (Brannan et al., 2000). There was reduction in soluble organic N, which resulted in increases in inorganic forms of N (Brannan et al., 2000). In the pre-BMP period, 22% of total N was soluble organic N, compared to only 17% in the post-BMP period. The reduction in soluble organic N resulted in increases in inorganic forms of N. The proportion of N in the soluble inorganic form (nitrate + ammonium) increased from 43% in the pre-BMP period to 50% in the post-BMP period (Brannan et al., 2000). There are both benefits and drawbacks to the transformation of N from organic to inorganic forms. A benefit is that the inorganic forms of N are available to plants, thus increasing the nutrient value of the manure. However, these same inorganic forms of N also promote the growth of nuisance aquatic plants and algae.

***Phosphorus***

As with N, there were reductions observed at QOA for flow-weighted P concentrations in the post-BMP period, but there was some increases observed in soluble forms of P. The changes in average annual flow-weighted concentrations observed at QOA are presented in Table 2. BMPs reduced the concentrations of all forms of P, except orthophosphorus-P. Despite the increase in stream flow (28%), the BMPs reduced average annual total P concentrations leaving the Owl Run watershed by 64%. The average annual soluble P flow-weighted concentration decreased by 39% in the post-BMP period. Likewise, BMPs reduced annual average flow-weighted soluble organic P concentration (74%) and flow-weighted concentrations of particulate-P (78%). The only increase observed, after BMPs were implemented, was for orthophosphorus-P, which increased by 7% (Table 2). The reduction in organic forms of P in the post-

BMP period could be related to the conversion of organic P to inorganic forms of P while the manure is stored. However, since inorganic forms of P are not lost to the atmosphere, as is the case with N, the management flexibility provided by the manure storage structures probably contributes more to the total, soluble, and particulate-P reductions. For P, this flexibility mainly impacts the timing of manure applications rather than its location. Since the nutrient management plans were based on crop N needs, some fields were likely to have received excessive levels of P. This over-application is likely the cause of elevated orthophosphorus concentration (Table 2) observed after the implementation of BMPs. However, spreading of manure did not occur daily when storage was available, but rather allowed for application of manure at times when it is less likely to be removed by runoff.

The monthly distribution of most forms of flow-weighted P concentrations exhibits similar reductions in the post BMP period as observed in the average annual values. The average monthly flow-weighted concentrations for total P observed at each station are shown in Figures 6. Reductions were observed in the flow-weighted total P concentration in the post-BMP period except for the months of May and September (Figure 6). There were very slight (less than 1%) increases in the flow-weighted total P concentration for these months. The greatest reduction in total P flow-weighted concentrations occurred in the winter-early spring, late summer and fall periods for station QOA. There were some large reductions in total P observed for several months at stations QOC and QOD (Data not shown), but for the months of March, April, June, and August, increases were observed in the post-BMP period for the upland subwatersheds.

The reductions observed in total P were also observed in some other forms of P during the post-BMP period, but not in orthophosphorus-P. The average monthly flow-weighted concentrations for orthophosphorus-P observed are show in Figure 7. The reader is referred to Brannan et al. (2000) for a more detailed discussion of the reductions in other forms of P. Reductions during the post-BMP period were observed at station QOA for orthophosphorus-N flow-weighted concentrations, except for the months of April, May and June (Figure 7). Increases were observed for these three months. The largest reductions for orthophosphorus-P were observed in the late fall.

The reductions in most forms of P, except the soluble forms, in the Owl Run watershed are related to the transformations of P due to the storage of the manure and the N-based nutrient management of manure applications to cropland. For N-based nutrient management plans, over-application of P on cropland may occur because P content of the manure may be higher than the that of N or because P needs of crops may be lower than the amount of N required by the crops (Sharpley et al., 1994). As with N, organic forms of P are converted to inorganic forms by microbial actions during storage. Unlike N, the inorganic P is not lost to the atmosphere and remains in the stored manure until its application. The transformation of organic P to inorganic forms is evident from the increases in the proportions of organic forms of P to inorganic forms after BMP implementation (Brannan et al., 2000). As with the increase observed in the proportion of soluble N, there are both benefits and drawbacks to the increases in the soluble forms of P. The main benefit is that the increase in soluble forms of P increases the nutrient value of the manure, but these same soluble forms of P also promote the growth of aquatic plants and algae. This is especially true for orthophosphorus-P, which is highly mobile by surface runoff and is a key nutrient in eutrophication. In the post-BMP periods, the average annual flow-weighted total P concentrations (Table 2) exceeded the eutrophication standard of 0.10 mg $L^{-1}$ total P concentration for streams not discharging directly into lakes (USEPA, 1986).

## CONCLUSIONS

The results of the 10-year study conducted in the Owl Run watershed clearly indicated the beneficial impacts of the BMPs on surface water quality with respect to nitrogen (N), but the results of the study were mixed with respect to phosphorus (P) levels observed in the streams. The main objective of the study was to determine the effectiveness of a system of animal waste BMPs for improving surface water quality. Some of the BMPs implemented as a part of the study included animal waste storage structures, nitrogen-based nutrient management, stream fencing, water troughs, and no tillage, among others.

Reductions in both N and P were observed due to the implementation of BMPs. A small increase in precipitation amount was observed in the post-BMP period and corresponding increases were observed in stream flow. Even with the increases in stream flow, large reductions were detected in monthly flow-weighted concentrations. Reductions in all forms of N and most forms of P were observed in the average annual values due to BMPs. The BMPs were effective in reducing average monthly flow weighted concentrations of total N for most of the year. BMPs were also effective in reducing other forms of N. At station QOA, reductions in nitrate-N were observed in the monthly average flow-weighted concentrations after BMP implementation. The BMPs were less effective in reducing nitrate-N concentrations for the summer months in the upland watersheds monitored.

Although BMPs were effective in reducing concentrations of most forms of P, orthophosphorus-P concentration increases in the average monthly flow-weighted concentrations were observed for at least part of the year after BMP implementation. Increases in the flow-weight concentrations of orthophosphorus-P were observed for April, May, and June. Reductions in orthophosphorus-P concentrations were observed at station QOA after BMP implementation for the remaining months.

In general, BMPs reduced the total amount of N and P, but increased the proportion of bioavailable forms of N and P reaching the streams of the Owl Run watershed. This is especially evident for P. The increases in inorganic forms of N and P can be attributed to specific BMPs. For example, waste storage facilities allow for the conversion of organic forms of N and P to inorganic forms during storage. Similarly, increases in soluble forms of P may be the result of N-based nutrient management plans. Over-application of P on cropland may occur due to the P content of the manure being higher than the N content or because P requirements may be lower than the amount of N needed by the crops. The increase in the proportion of bioavailable forms of both N and P in the streams of the Owl Run watershed could result in excessive aquatic plant growth and algae blooms. The system of BMPs implemented in the Owl Run watershed were effective in reducing flow-weighted concentrations of total N and P throughout the year. However, possible increases in bioavailable P indicates that P-based nutrient management plans should be considered where P is the major water quality concern.

## ACKNOWLEDGEMENTS

The Owl Run Project was funded, in part, by Virginia Department of Conservation and Recreation, Division of Soil and Water Conservation, Richmond Virginia.

## REFERENCES

Brannan, K. M., S. Mostaghimi, P. W. McClellan, and S. P. Inamdar. 2000. Animal waste BMP impacts on sediment and nutrient losses in runoff from the Owl Run watershed. Trans. of ASAE. (In press)

Cabrera, M. L. and R. M. Gordillo. 1995. Nitrogen release from land-applied animal manures. In: Animal Waste and the Land-Water Interface. K. Steele Ed. CRC Lewis Publs., New York. pp 589.

Collins, E. R., J. D. Jordan, and T. A. Dillaha. 1995. Nutrient values of dairy manure and poultry litter as affected by storage and handling. In: Animal Waste and the Land-Water Interface. K. Steele Ed. CRC Lewis Publs., New York. pp 589.

Crane, S. R., J. A. Moore, M. E. Grismer, J. R. Miner. 1983. Bacterial Pollution from agricultural sources: A review. Transactions of the ASAE 26(3):858-866, 872.

Darling, J. M. 1962. Maryland streamflow characteristics. Bulletin 25. U. S. Geologic Survey, Reston, VA.

Eghball, B. and J. F. Power. 1994. Beef cattle feedlot manure management. Journal of Soil and Water Conservation 49(2):113-122.

Mostaghimi, S., K. M. Brannan, P. W. McClellan, S. P. Inamdar and S. Shukla. 1999. Impacts of BMP on Surface Water in the Owl Run Watershed. Prepared for Virginia DCR, Div. of Soil and Water Conservation. Report No. 0999OR-F, Blacksburg: Virginia Polytechnic Institute and State Univ.

Sharpley, A. N., S. C. Charpra, R. Wedepohl, J. T. Sims, T. C. Daniel and K. R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environmental Quality. 23(3):437-451.

Soil Conservation Service. 1956. Fauquier County Soil Survey. United States Department of Agriculture.

Spooner, J., R. P. Mass, S. A. Dressing, M. D. Smolen, and F. J. Humenik. 1985. Appropriate design for documenting water quality improvements from agricultural nonpoint source control programs. In Proc. Perspectives on nonpoint source pollution. pp.30-34 United States Environmental Protection Agency. Washington, D. C. EPA 440/5-85/001.

USEPA. 1983. Chesapeake Bay: A Framework for Action. US EPA, Philadelphia, PA.

USEPA. 1986. Quality criteria for water. USEPA Rep. 400/5-86-001. USEPA. Office of Water Regulations and Standards. U. S. Gov. Print. Office (PB87-226759). Washington, D. C.

**Table 1. Major BMPs Implemented in Owl Run Watershed.**

| **Practice Name** |
|---|
| Manure Storage Structure |
| Nutrient Management (based on nitrogen crop needs) |
| Stream Fencing |
| Watering Troughs |
| Stream Crossing for Animals |
| Winter Cover Crops |
| Field Strip Cropping |
| Grassed Waterways |

**Table 2. Average Annual Hydrologic Parameters and Nutrient Flow-Weighted Concentrations at watershed outlet (QOA).**

| Parameter | Pre-BMP | Post-BMP | % Change[1] |
|---|---|---|---|
| Precipitation (mm) | 1,054 | 1,075 | 2% |
| Stream Flow (mm) | 345 | 440 | 28% |
| TSS (g $L^{-1}$) | 0.31 | 0.20 | -35% |
| Ammonium-N (mg $L^{-1}$) | 0.93 | 0.40 | -57% |
| Nitrate-N (mg $L^{-1}$) | 2.81 | 1.84 | -35% |
| Total N (mg $L^{-1}$) | 8.83 | 4.50 | -49% |
| Soluble N (mg $L^{-1}$) | 5.71 | 2.98 | -48% |
| Soluble Organic N (mg $L^{-1}$) | 1.96 | 0.75 | -62% |
| Particulate-N (mg $L^{-1}$) | 3.13 | 1.52 | -51% |
| Total P (mg $L^{-1}$) | 2.70 | 0.98 | -64% |
| Soluble P (mg $L^{-1}$) | 0.99 | 0.60 | -39% |
| Soluble Organic P (mg $L^{-1}$) | 0.57 | 0.15 | -74% |
| Particulate P (mg $L^{-1}$) | 1.71 | 0.38 | -78% |
| Orthophosphorus-P (mg $L^{-1}$) | 0.42 | 0.45 | 7% |

[1] Percent Change = $\left[\frac{(\text{Post} - \text{Pre})}{\text{Pre}}\right] \times 100$

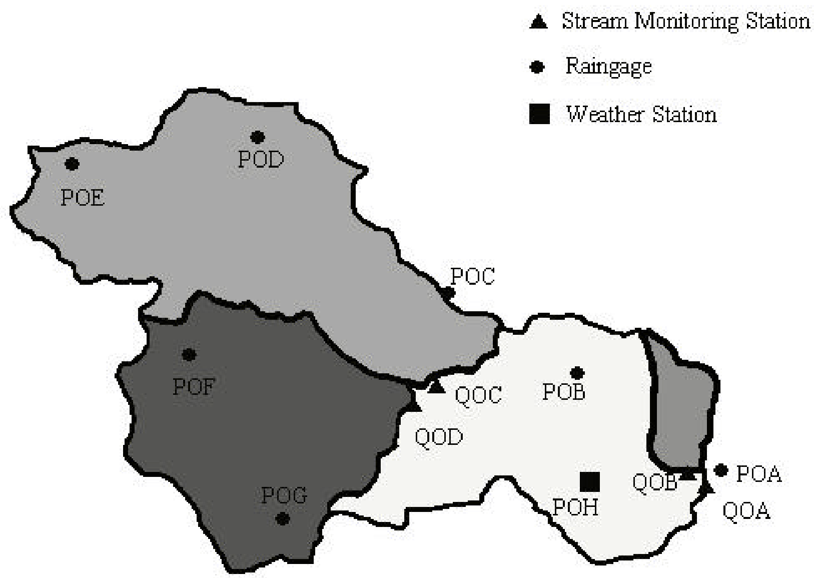



**Figure 1. Location of rain gauges, weather and monitoring stations.**

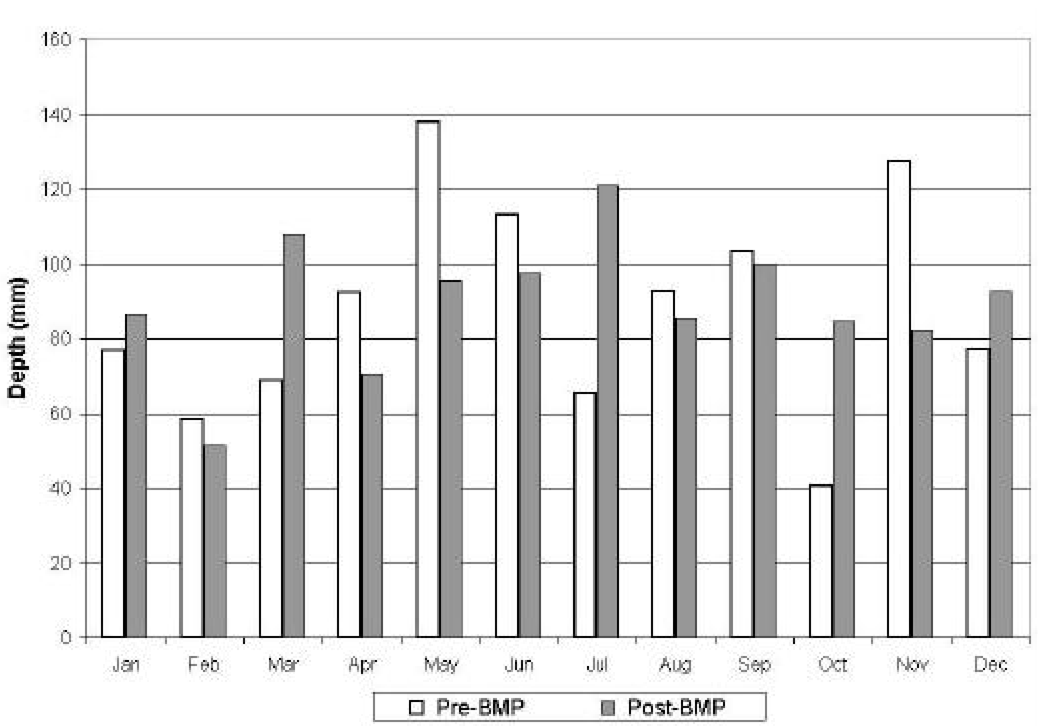



**Figure 2. Average monthly precipitation depth.**

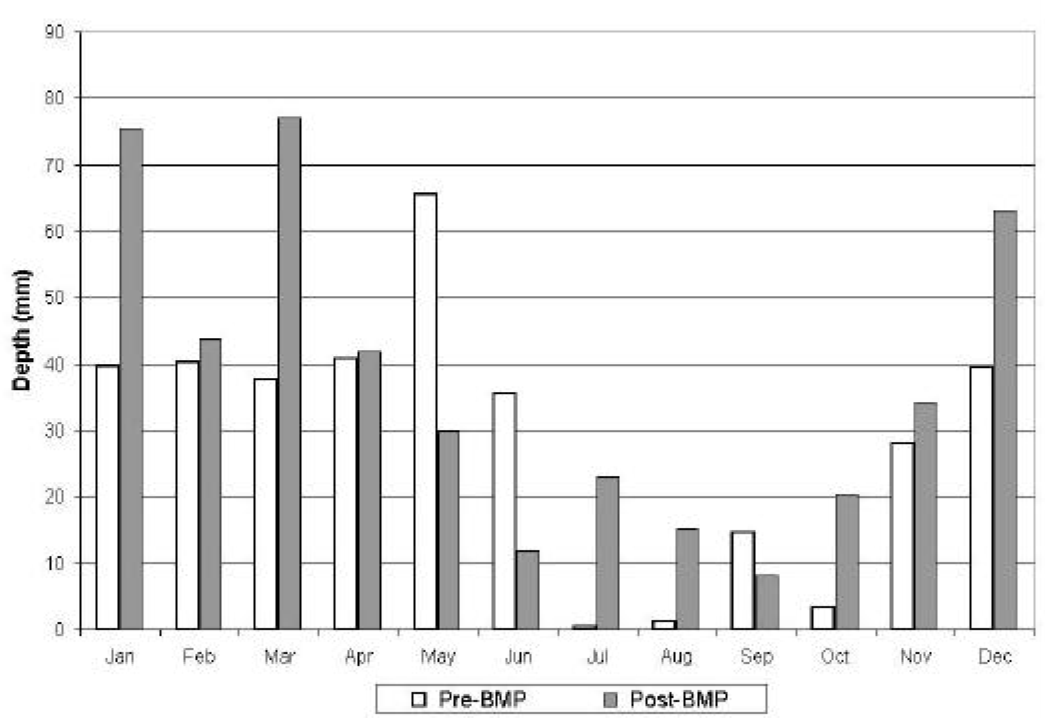



**Figure 3. Average monthly stream flow volume at watershed outlet.**

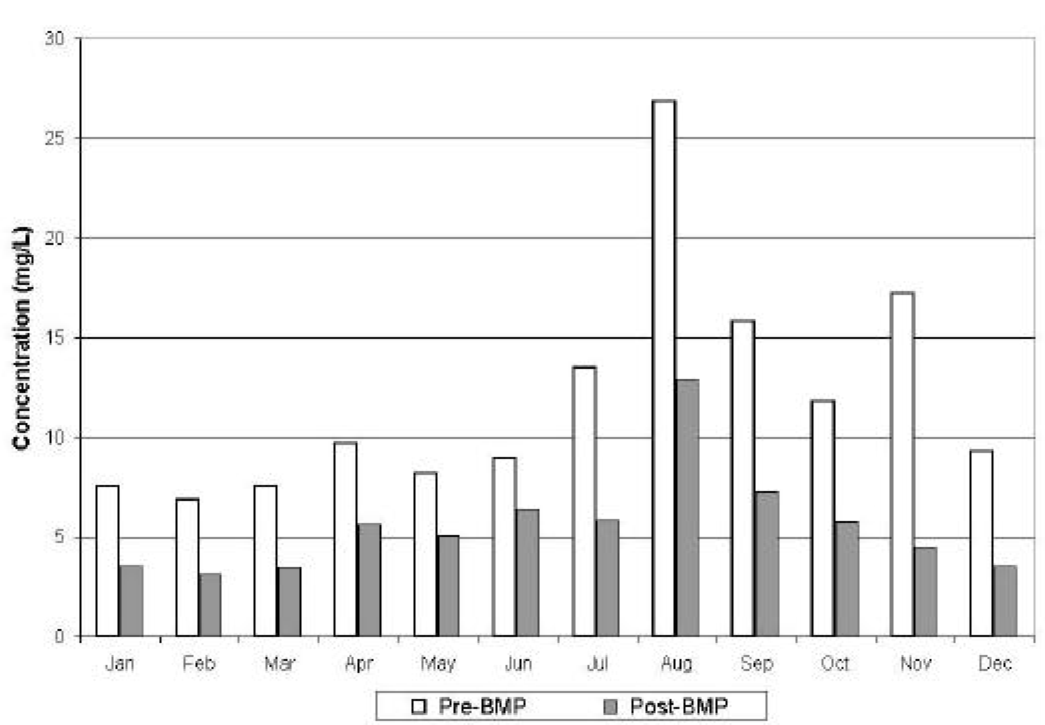



**Figure 4. Average monthly flow-weighted total nitrogen concentration at watershed outlet.**

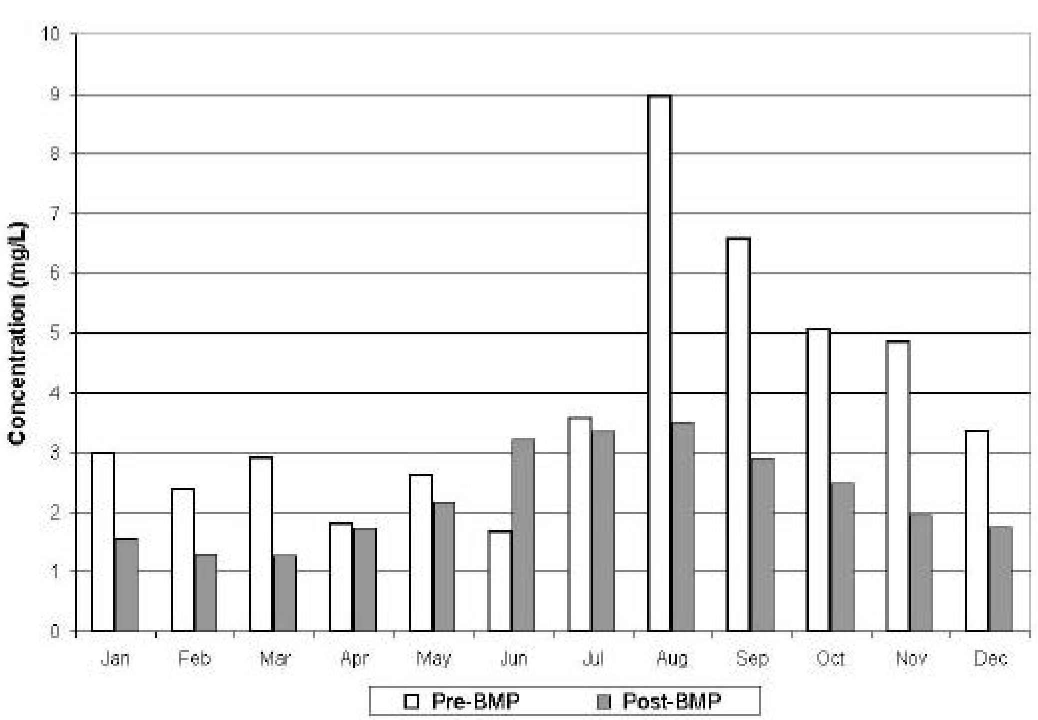



**Figure 5. Average monthly flow-weighted nitrate- nitrogen concentration at watershed outlet.**

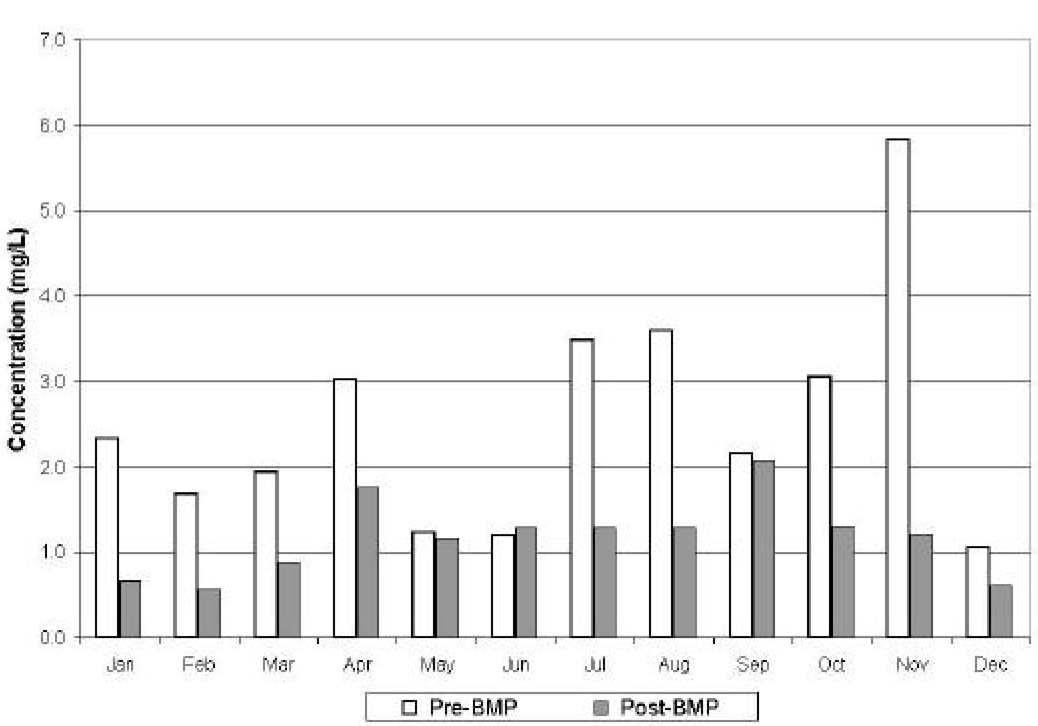



**Figure 6. Average monthly flow-weighted total phosphorus concentration at watershed outlet.**

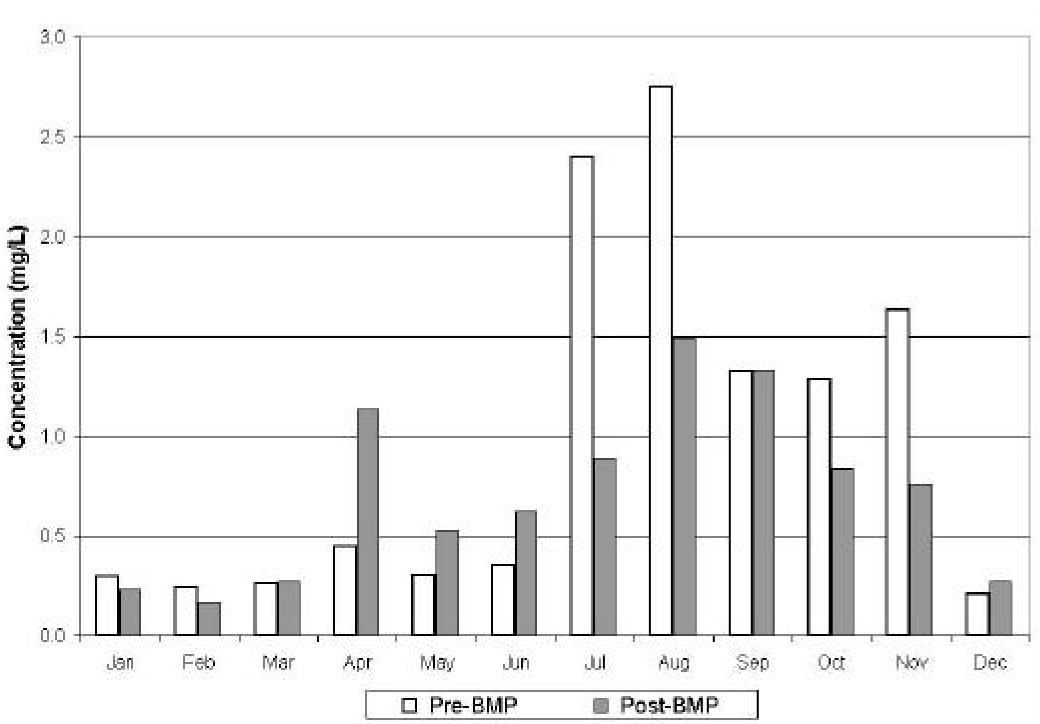



**Figure 7. Average monthly flow-weighted orthophosphorus concentration at watershed outlet.**

# A Preliminary Monitoring Program for Fecal Coliform Bacteria Within the Upper Appomattox River Watershed

David W. Buckalew and Mona M. Hafez
Department of Natural Sciences, Longwood College
201 High Street, Farmville, Virginia 23909
E-mail: dbuckale@longwood.lwc.edu

## ABSTRACT

In 1998, Virginia's Department of Environmental Quality (DEQ) identified a 77-mile portion of the Appomattox River and many of its tributaries as impaired (1998 305(b) Virginia Water Quality Assessment). Violations of VA's water quality standard for fecal coliform bacteria were recorded by DEQ at several monitoring stations on the Appomattox River, Sayler's Creek and other streams in and around Prince Edward County. The impairments are generally attributed to agricultural runoff, but have not been verified. Longwood College started the Appomattox River Water Quality Monitoring Program (ARWQMP) in conjunction with Clean Virginia Waterways to further delineate the extent of impairment and to characterize its causes and sources. The study area includes more than 30 sites in Prince Edward County, and portions of Appomattox, Buckingham, Cumberland, Amelia and Nottoway Counties. The goal of the program is to generate information with respect to: 1) pollution levels, 2) pollution sources in order to assist with land use management and stewardship, and community awareness, and 3) providing a baseline of information for future study. While many water quality parameters are monitored in this program, this effort has concentrated on building Longwood College's capacity to analyze water samples for indicator bacteria (total coliforms, fecal coliforms, *Escherichia coli*, fecal streptococci) of fecal contamination.

The upper Appomattox River watershed contains beef and dairy cattle farms, some chicken and hog confined animal feedlot operations (CAFOs), rural homes with possibly outdated septic systems, and an abundance of wildlife, all of which are possible sources of fecal pollution. While not all fecal bacteria are pathogenic, some present a public health risk. This paper presents the results of the ongoing coliform analyses using the traditional membrane filtration method, as well as a newer Colilert Defined Substrate Technology (DST) from IDEXX which automates the most probable number methodology for a more rapid and sensitive assessment of indicators of fecal pollution. Recent evidence indicates the lack of false positive results using DST as well as obviating the requirement for media preparation using the pre-packaged media. The results of the data suggest: 1) several tributaries (Angola Creek, Green Creek, Sayler's Creek, and Vaughan's Creek) occasionally introduce high levels of fecal pollution into the upper Appomattox River, 2) these occasional increases suggest agricultural nonpoint source runoff; 3) the Colilert method produces quicker and more reliable results for fecal coliform/*E. coli* testing, and 4) future bacterial source tracking will be required to ascertain fecal sources within this watershed.

## INTRODUCTION

Impaired water quality is emerging as one of the more critical public issues with respect to environmental quality. In 1998, Virginia's Section 303d (Clean Water Act) list consisted of 883 waterways exhibiting 1,002 impairments, the most extensive of which, is contamination by fecal coliform bacteria. These bacteria affect thousands of miles of stream length within the state (van der Leeden, 1993). According to the Total Maximum Daily Load (TMDL) priority list, surface waters are 'impaired' if they exceed a geometric mean of 200 fecal coliform bacteria per 100 ml of water for two or more samples over a 30-day period, or a level that exceeds 1,000 cells per 100 ml at any time.

Fecal coliform bacteria are within the prokaryotic Family Enterobacteriaceae. They are facultative, Gram negative, rod-shaped, non-sporing bacteria that exhibit lactose fermentation (among other fermentations). They produce a variety of acids (lactic, acetic, succinic, etc.) and gas (hydrogen and carbon dioxide). Fecal coliform bacteria thrive in the gastrointestinal tracts of a wide variety of animals, including man. Several members of this bacterial family are of concern to public health as they can cause enteric (gastroenteritis, salmonellosis, etc.) or systemic (typhoid, enteroinvasive/ enterohemorrhagic) illness in humans (Prescott et al., 1999).

Fecal coliform contamination typically enters surface water from improper sewage treatment discharge and/or agricultural runoff. The later form of contamination contributes the most significant numbers of coliform bacteria in Virginia's waterways (USDA, 1997). Agricultural nonpoint runoff causes widespread effects to surface water quality due to inputs of fertilizer nutrients and pesticides as well as organics and bacteria from fecal contamination. Groundwater can be affected through percolation of nutrients and pesticides through the soil column.

Animal waste is considered a significant nonpoint source of water pollution from farms that have not adopted best management practices (BMP) or sustainable land use practices. According to the most recent census by the U.S. Department of Agriculture (1997), Virginia farms raised more than 264 million chickens, 26 million turkeys, 710,000 hogs and pigs, and 1.6 million cattle. The wastes from these 292 million animals constitute a significant bacterial output. The total waste produced by that number of cattle alone is equal to the waste produced by 7 million people (VA Agric. Stats., 1991). In addition to the effects of nutrient loads to surface and groundwater is the considerable bacterial inoculum the waste contains. These effects pose an environmental as well as a public health problem.

The Appomattox River is a major river system in Virginia draining 1,023,851 acres of agricultural, rural residential, and urban land, joining with the James River in Hopewell, VA. Appomattox River drains portions of 12 counties within the Piedmont region, six of which are contained in the area of study for this project. Of the monitoring sites in the ARWQMP, ten were selected for this report, based on their histories of bacterial loading from the Virginia DEQ monitoring data.

The data obtained in this study are part of a larger water monitoring program initiated by Kathleen Register, Director of Clean Virginia Waterways in conjunction with Longwood College, its microbiology faculty and students. While many water quality parameters (temperature, pH, N&P, DO, stream depth, and turbidity) are monitored in the ARWQMP study, this work focuses on indicators of fecal contamination. It has promoted public awareness of the issue as well as provided an educational benefit to Longwood's students. The ultimate goal of this program is to increase the quantity and scope of water quality monitoring in the upper Appomattox drainage in order to form a basis for long-term assessment of water quality in the watershed.

## MATERIALS AND METHODS

From December 1999 through September 2000, approximately 100 samples of stream water, originating from each of ten (10) sites along the upper Appomattox River drainage were examined (refer to Table 1). Water samples were obtained from all sites once per month (3rd Tuesday) from technicians at the Piedmont Soil and Water Conservation District office in Farmville, VA. The water samples were collected in sterile polyvinyl (Whirl-Pak) bags according to Pepper et al., (1999), stored in ice chests and transported to a walk-in refrigerator at Longwood College. Sufficient water was collected from each site to perform simultaneous split sample analyses by the membrane filtration method and by the Colilert most probable number assay. All water samples were processed for the bacterial assays within six hours of collection according to the procedures in the Standard Methods reference (APHA, 1998).

## Membrane Filtration using m-FC broth media

Each sample was diluted (1:100) by placing 1 ml of a well-mixed sample with 99 ml of sterile deionized water. The diluted sample was drawn through a 0.45 um (Millipore) filter. The filter was incubated on m-FC broth at 45° C ± 0.5° C for 24 ± 2 hours. The number of deep blue colony forming units (CFU) times the dilution factor of the sample gave the total number of fecal coliform bacteria per 100 ml of original sample. Colonies exhibiting typical physical characteristics for suspect fecal coliforms on m-FC broth were confirmed by incubation on yeast peptone agar (YPA) for positive selection and tested for oxidase reactions. Oxidase negative isolates were aseptically transferred to lactose tryptose lauryl sulfate broth (LTLSB) tubes (with Durham tubes) and incubated at 45° C ± 0.5° C for 24 ± 2 hours followed by the addition of Kovac's reagent to the LTLSB tubes positive for gas production to test for indole formation. Several duplicate samples were run each month and at least two (2) blanks were run to detect inadequacies in rinsing between samples.

## Colilert Method using DST

For the first months of the study (Dec–Mar), 15-tube serial dilutions (5 tubes each at 100%, 10%, and 1% dilutions) were used with the Colilert (IDEXX Laboratories, Westbrook, ME) reagent (containing ortho-nitrophenyl-B-D-galactopyranoside (ONPG) and 4-methylumbelliferyl-B-D-glucuronide (MUG)) in a multiple tube fermentation. After a 24-hr. incubation at 35° C ± 0.5° C the number of positive tubes for Total Coliforms (exhibiting a yellow color equal to or greater in intensity than a comparator sample) were compared to a MPN table to estimate bacterial numbers per 100 ml of original sample. The tubes were illuminated with a 6 watt 365 nm ultraviolet light. The number of tubes testing positive (exhibiting fluorescence equal to or greater than a comparator sample) provided an MPN value for the number of *E. coli* per 100 ml of original sample.

Since May, all samples have been tested using the IDEXX Quanti-Tray automated dilution system. Colilert reagent was added to a diluted (25%) water sample in a capped, sterile bottle and inverted to mix. The entire contents of the bottle was aseptically poured into a Quanti-Tray envelope, heat sealed, and incubated at 35° C ± 0.5° C for 24 ± 2 hours. The number of large and small wells testing positive for utilization of ONPG provided a numerical estimate via MPN. The MPN value multiplied by the dilution factor gave total coliforms per 100 ml of original sample. The wells were illuminated with a 6 watt 365 nm UV light. The number of wells testing positive for utilization of MUG provided an MPN value. The MPN value multiplied by the dilution factor estimated the number of *Escherichia coli* per 100 ml of original sample.

# RESULTS

The overall results of the study are presented in Table 1. The following sample sites will be grouped together for the sake of brevity and discussion: Appomattox River sites 1 and 2 (APP1 and APP 2). Sayler's Creek sites 5, 6, 7, and 8 (SAY 5, SAY 6, SAY 7, and SAY 8). Vaughan's Creek site 14 and Buffalo Creek site 15 (VAU 14, BUFF 15). Green Creek site 16 and Angola Creek site 17 (GRE 16 and ANG 17) will be examined separately.

**Appomattox River sites 1 and 2.** Samples taken from the Appomattox River sites (Figures 1 and 2) reveal similar patterns of fecal coliform (fc) loads throughout the study period. Both the winter (Dec/Jan/Feb) and late spring/summer (May/June/July/Aug) counts were similar for both sampling sites.

Coliform levels at both locations increased to a peak (5,000-5,500 fc /100 ml) during March and April, followed by a rapid decline (to <100 fc /100 ml) from May through August. In nearly all months sampled, the counts of fecal coliforms approximated the counts of *E. coli* whenever both assay techniques were employed.

**Sayler's Creek sites 5, 6, 7, and 8.** Samples taken along Sayler's Creek exhibit two distinct patterns with respect to bacterial loads (Figures 3, 4, 5, and 6). One pattern reveals a sharp, high volume increase during March/April/May samplings. The other reveals a brief elevation in April. High in the drainage (SAY 7 and SAY 8, see Figures 5 and 6), bacterial counts remained low (low 100's fc/100ml) for most of the period, peaking to 2100-2400 fc/100 ml in April. Another sharp peak was observed at the Say 7 site in September. Lower in the drainage (SAY 5 and 6, see Figures 3 and 4), bacterial counts were low (low 100's fc/100 ml) between December and February and between June and August. Fecal coliforms peaked (7500-9500 fc/100 ml) during March, April, and May. At SAY 5 and SAY 6, the peaks of bacterial loading occurred over a 3 month period as compared with the 1 month peak measured at SAY 7 and SAY 8. The counts of *E. coli* appear lower during March for both SAY 5 and SAY 6 due to the sample dilutions required for the Multiple Tube Fermentation, hence our data was bound to a maximum of >1600 cells/100 ml of sample.

**Vaughan's Creek site 14 and Buffalo Creek site 15.** Vaughan's Creek site 14 (Figure 7) and Buffalo Creek site 15 (Figure 8) mirror the count patterns observed specifically for the Appomattox River sites and generally for the majority of the testing sites. The peaks in bacterial loads for Vaughan's Creek and Buffalo Creek are higher than those recorded at the Appomattox sampling sites and approximate the spring peaks measured within the downstream Sayler's Creek sampling sites (SAY 5 and SAY 6).

**Green Creek site 16.** Green Creek (Figure 9) exhibited a low increase (to 2800 fc/100 ml) during March and April. Low numbers (<100 fc/100 ml) were recorded during December, January, and February. Slightly elevated numbers (to 500 fc/100 ml) were observed during the summer months, June, July, and August. Another strong increase in cell counts similar in magnitude to the peaks observed from Sayler's Creek sites 5, 6, and 7 occurred during September resulting in a fecal coliform total that was 'too numerous to count' (TNTC) at the standard 1% sample dilution.

**Angola Creek site 17.** Of all sampling locations, Angola Creek site 17 (Figure 10) produced the highest numbers and greatest variability of fecal coliform counts. The winter months of December, January, and February revealed low counts (300, 400, and 100 fc/100 ml, respectively). The remainder of the samples exhibited a high degree of variability from a low of 700 fc/100 ml in June between peaks of high counts in April and May (15,900 fc/100 ml and TNTC) and July (8400 fc/100 ml).

## DISCUSSION

The data represent the initial 9 months of sampling data from our bacterial survey. This investigation has not attempted to correlate amounts of precipitation or water flow rates or wholly considered land use practices (even though the area drained by each of these streams is primarily agricultural land) nor performed bacterial source tracking. The data provide an initial point of reference to determine the health of the aquatic system in light of presence and degree of fecal coliform contamination. Assessing the water microbiologically has drawn our laboratory to investigate and compare two current protocols for the determination of fecal bacteria in natural waters – namely, the membrane filtration and the Colilert defined substrate methodologies.

The preliminary results of the Longwood College/Clean Virginia Waterways stream monitoring program for fecal coliform analysis are as follows:

1) **Counts of fecal coliform bacteria and *E. coli* increase universally at varying times during the year.**

The simultaneous increase in fecal bacteria in sites monitored during March and April suggests increased agricultural (or otherwise) activity or increased drainage/runoff into the streams or both. The precipitation totals recorded within one week of obtaining the monthly stream samples measured an average of 3.16" and 2.33", respectively for March and April 2000. These values are not appreciably different than those measured prior to other sampling dates for the study period. Generally, precipitation is well distributed (ranges from 42-44 in./yr) throughout the year, with no distinct wet or dry periods (van der Leeden, 1993). Stream flow is fairly uniform across Virginia, with annual peaks during March and annual troughs during late summer/early fall. Notably in September 2000, there were high rainfall amounts (~1.50") the night before samples were taken. It is also noteworthy that when increases in bacterial loads were measured, there was an increase across all sites tested.
Future analysis will reveal if this increase is seasonal and/or related to precipitation events overlapping sampling days. Additionally, since this study has not completed a full annual cycle, another period(s) of increased coliform loading may be measured. Clearly, several years of data would serve to illuminate these trends.

2) **Estimates of *E. coli* from the Colilert methodology roughly approximate membrane filtration counts of fecal coliforms…and with greater efficiency.**

A t-test analysis of our data revealed there to be no significant difference ($P > 0.05$) between counts of total fecal coliform bacteria recorded by membrane filtration and counts of *E. coli* by Colilert DST over the course of the study. In some cases (multiple tube fermentation), the dilution of the water sample permitted a maximum count of ">1600" *E. coli* /100 ml. These values and their appropriate comparisons were not included in the statistical test.

Throughout the first four months of testing, confirmatory tests were completed on colonies exhibiting the diagnostic blue coloration on m-FC broth. In most cases (91% of colonies tested), the confirmatory tests revealed oxidase negative, Gram negative bacilli that exhibited lactose fermentation and indole production. Subsequent cultivation on Eosin-Methylene Blue (EMB) agar revealed the characteristic dark/green colonies with a metallic sheen. Some of the colonies not confirmed as fecal coliforms were later identified as free-living heterotrophs (*Aeromonas* and *Pseudomonas* among others). In every case, aliquots of samples testing positive for *E. coli* (exhibiting fluorescence) with the Colilert DST showed a positive confirmatory result in the above tests.

Both membrane filtration and Colilert DST techniques have been widely studied and are approved for regulatory monitoring of waters (Bissonnette et al., 1977; Cowburn et al., 1994; Edberg et al., 1994; Eckner, 1998). By far, the most commonly used technique for detecting the presence of fecal coliforms from water has been membrane filtration using m-FC broth. Our lab has found that the Colilert DST provides a more rapid and accurate assessment of indicators of fecal pollution in our water samples. The greatest advantage being that it is highly accurate and does not appear to be affected by other bacterial flora due to the specificity of the Colilert reagent. Additionally, the tubes and Quanti-trays reveal a distinct color response that is easily read and apparent within 24 hrs. Moreover, counts of total coliforms and *E. coli* are produced simultaneously. A disadvantage resides in the cost per sample which is approximately four times greater for the IDEXX analysis vs. membrane filtration.

3) **Levels of fecal coliform bacteria increase as a function of location in the drainage.**

The sampling data along Sayler's Creek revealed cumulative effects of bacterial loading as a function of distance down the drainage. The spring peaks in coliform levels measured from high in the drainage at the SAY 7 and SAY 8 sites (Figures 5 and 6) were short lived and not as pronounced or prolonged as levels lower in the drainage at the SAY 5 and SAY 6 sites (Figures 3 and 4). There appears to be a direct correlation between fecal coliform levels and distance within the drainage. This bears significant impact on points downstream within the Appomattox River as well as points further east, including the James River and Chesapeake Bay.


## ACKNOWLEDGEMENTS

This work was supported, in part, by a grant (#99-15) from the Virginia Environmental Endowment to Longwood College based on a grant written by Clean Virginia Waterways and from generous support from the Longwood College General Trust Fund for the Natural Sciences.


## REFERENCES


American Public Health Association (APHA). 1998. Standard methods for the examination of water and wastewater, 20th Ed. L.S. Clesceri, A.E. Greenberg, A.D. Eaton (eds). Washington, D.C.

Bissonnette, G.K., J.J. Jezeski, G.A. McFeters, and D.G. Stuart. 1977. Evaluation of recovery methods to detect coliforms in water. Appl. Microbiol. 33: 590-595.

Cowburn, J.K., T. Goodall, E.J. Fricker, K.S. Walter, and C.R. Fricker. 1994. A preliminary study of the use of colilert for water quality monitoring. Lett. Appl. Microbiol. 19: 50-52.

Eckner, K.F. 1998. Comparison of membrane filtration and multiple tube fermentation by the Colilert and Enterolert methods for detection of waterborne coliform bacteria, *Escherichia coli*, and enterococci used in drinking and bathing water quality monitoring in southern Sweden. Appl. Environ. Microbiol. 64: 3079-3083.

Edberg, S.C., M.J. Allen, D.B. Smith, and the National Collaborative Study. 1988. National field evaluation of a defined substrate method for the simultaneous enumeration of total coliforms and *Escherichia coli* from drinking water: Comparison with the standard multiple tube fermentation method. Appl. Environ. Microbiol. 54: 1595-1601.

Edberg, S.C., M.J. Allen, and D.B. Smith. 1994. Comparison of the colilert method and standard fecal coliform methods. AWWA Research Foundation. Denver, CO.

Pepper, I.L., C.P. Gerba, and R.M. Maier. 1999. Environmental sample collection and processing. In: Environmental Microbiology. pp 177-194. Academic Press, San Diego, CA

Prescott, L.M., J.P. Harley, and D.A. Klein. 1999. Microbiology, (4th ed.). McGraw-Hill Publ. Boston, MA.

U.S. Department of Commerce. 1997 Census of Agriculture. Econ. And Stat. Admin. , Bureau of the Census, Wash. D.C.

van der Leeden, Fritz. 1993. Water Atlas of Virginia. Tennyson Press, Lexington, VA.

Virginia Department of Environmental Quality. 1998. Virginia 303(d) Total Maximum Daily Load Priority List Report. VA DEQ/VA DCR, Richmond, VA.

Virginia Department of Environmental Quality. 1998. Virginia 305(b) Virginia Water Quality Assessment Report. VA DEQ/VA DCR, Richmond, VA.

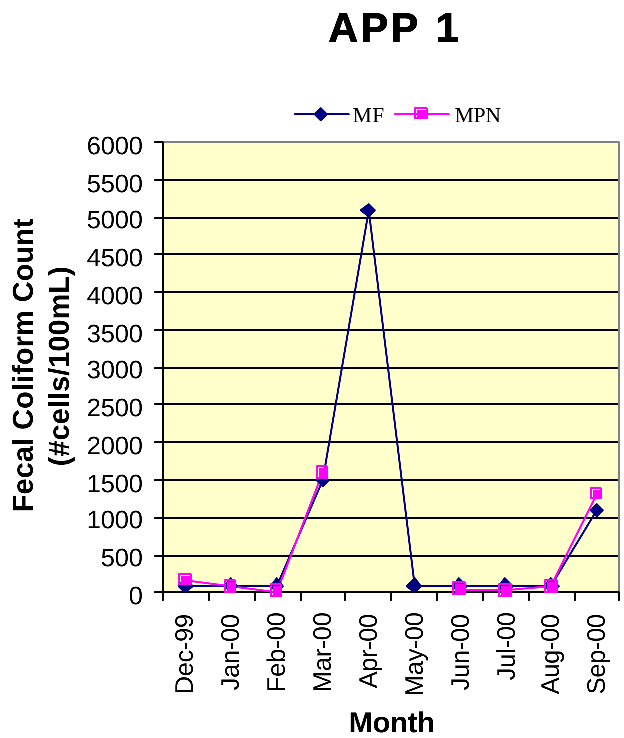



Figure 1. Appomattox River at Rt.. 609.  Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

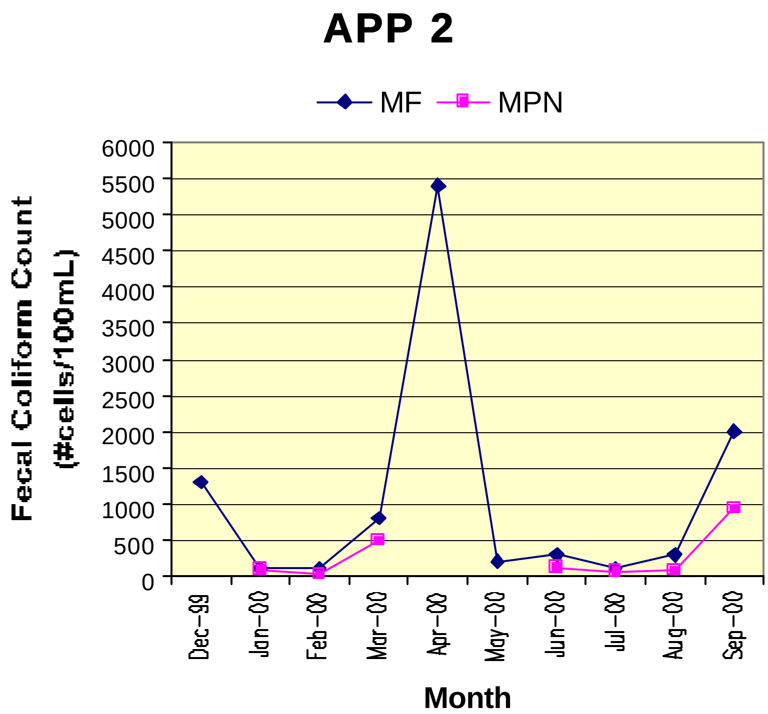



Figure 2. Appomattox River at Rt. 45. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

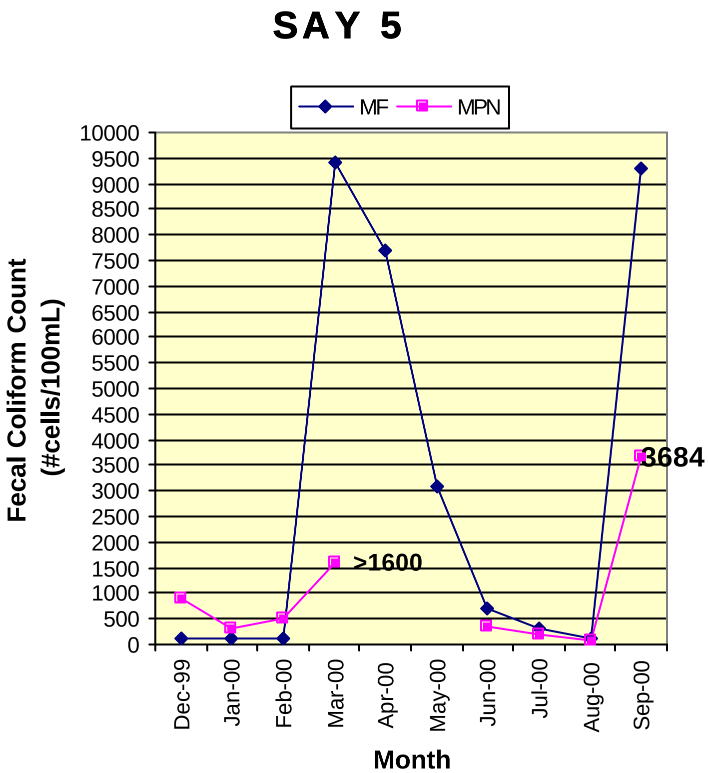



Figure 3. Little Sayler's Creek at Rt. 620. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

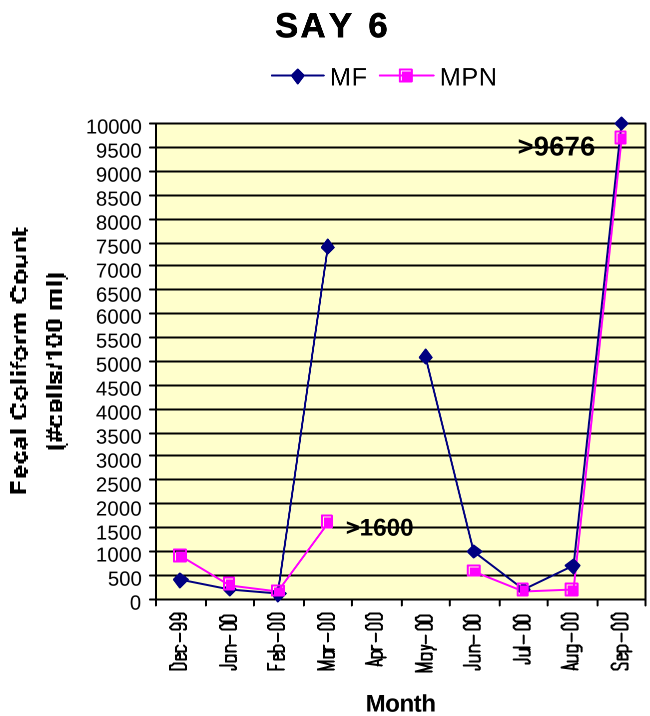



Figure 4. Little Sayler's Creek at Rt. 600. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

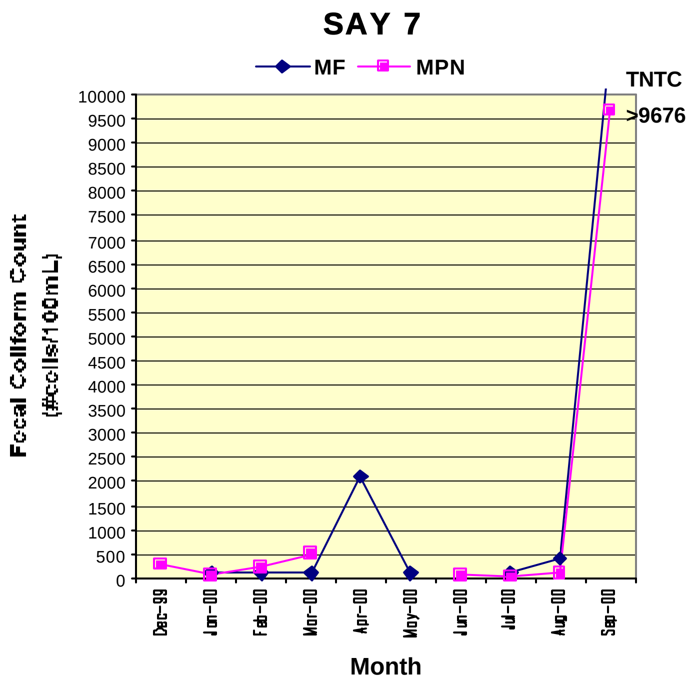



Figure 5. Big Sayler's Creek at Rt. 617. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

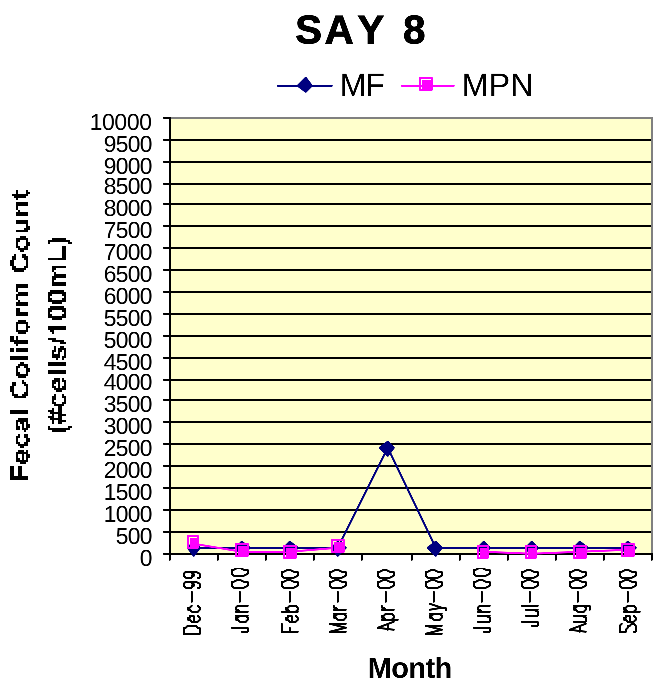



Figure 6 Big Sayler's Creek at Rt. 620. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

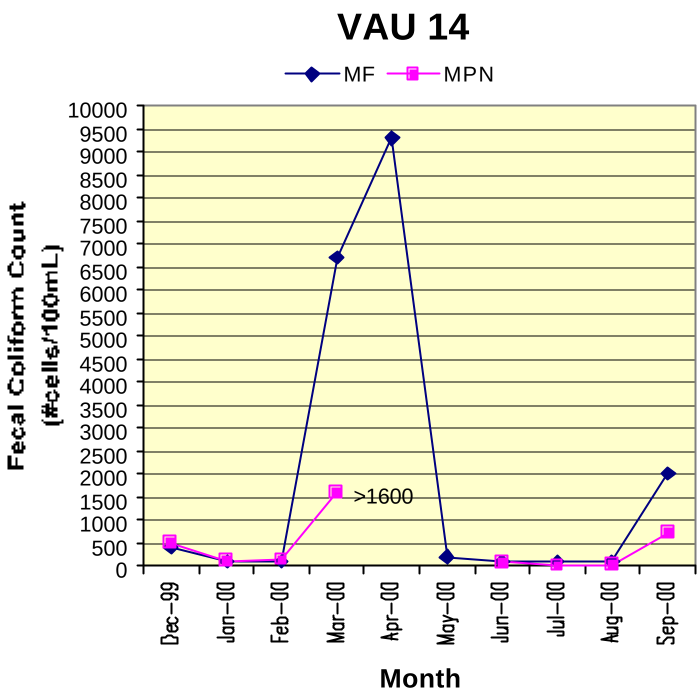



Figure 7 Vaughn's Creek at Rt. 609. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

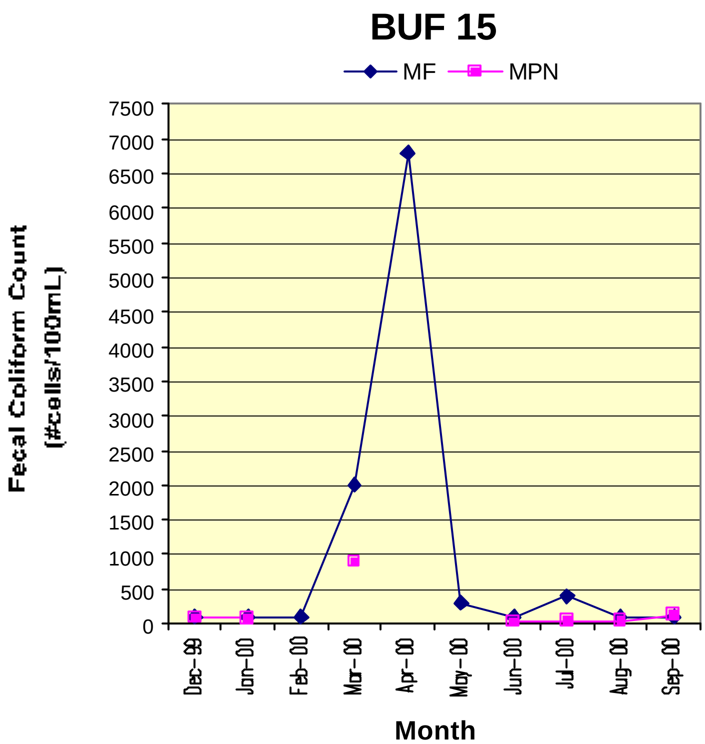



Figure 8. Buffalo Creek at Rt. 648. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

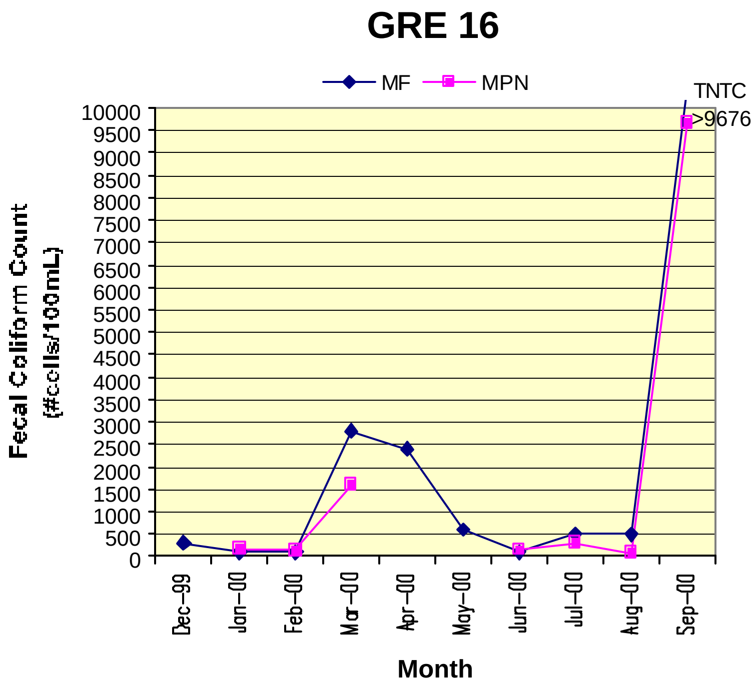



Figure 9. Green Creek at Rt. 600. Monthly counts of fecal coliform bacteria and E. coli at each of the following ten sampling sites.

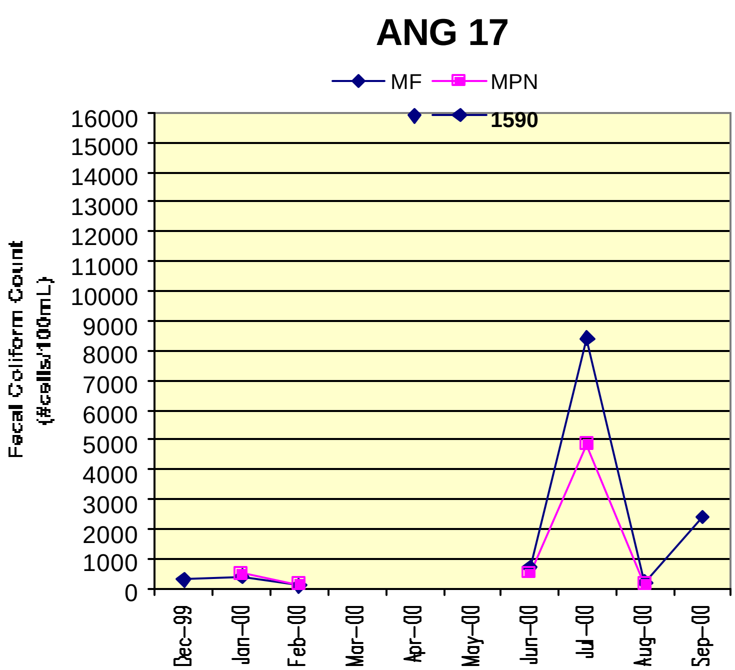



Figure 10. Angola Creek at Rt. 673. Monthly counts of fecal coliform bacteria and E. coli at each of the ten sampling sites.

**Table 1. Counts of fecal coliform bacteria and *Escherichia coli* from ten selected sites in the Appomattox River watershed**

Fecal coliform counts were obtained by membrane filtration (MF) and *E. coli* by most probable number (MPN) methodology.

| | Dec 1999 | | Jan 2000 | | Feb 2000 | | Mar 2000 | | April 2000 | | May 2000 | | June 2000 | | July 2000 | | Aug 2000 | | Sept 2000 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | MF | MPN | MF | MPN | MF | MPN | MF | MPN | MF | MPN | MF | MPN | MF | I-MPN | MF | I-MPN | MF | I-MPN | MF | I-MPN |
| App 1 | 100 | 170 | 100 | 80 | 100 | 17 | 1500 | 1600 | 5100 | ND | 100 | ND | <100 | 48 | <100 | 29 | 100 | 84.4 | 1100 | 1312 |
| App 2 | 1300 | ND | 100 | 90 | <100 | 33 | 800 | 500 | 5400 | ND | 200 | ND | 300 | 126 | <100 | 67.2 | 300 | 82.6 | 2000 | 944 |
| Say 5 | 100 | 900 | 100 | 300 | 100 | 500 | 9400 | >1600 | 7700 | ND | 3100 | ND | 700 | 368 | 300 | 191.8 | <100 | 81.2 | 9300 | 3684 |
| Say 6 | 400 | 900 | 200 | 300 | 100 | 170 | 7400 | >1600 | TNTC | ND | 5100 | ND | 1000 | 582 | 200 | 176.8 | 700 | 188.8 | TNTC | >9676 |
| Say 7 | ND | 280 | 100 | 50 | 100 | 220 | 100 | 500 | 2100 | ND | 100 | ND | ND | 50 | <100 | 31.2 | 400 | 107.6 | TNTC | >9676 |
| Say 8 | 100 | 220 | <100 | 35 | <100 | 9 | 100 | 140 | 2400 | ND | 100 | ND | <100 | 18 | <100 | 2 | 100 | <4 | 100 | 58 |
| Vau 14 | 400 | 500 | 100 | 110 | <100 | 140 | 6700 | >1600 | 9300 | ND | 200 | ND | <100 | 88 | <100 | 10.4 | 100 | 28.4 | 2000 | 716 |
| Buf 15 | 100 | 90 | <100 | 80 | <100 | ND | 2000 | 900 | 6800 | ND | 300 | ND | 100 | 32 | 400 | 36.6 | 100 | 42.2 | 100 | 128 |
| Gre 16 | 300 | ND | <100 | 170 | <100 | 140 | 2800 | >1600 | 2400 | ND | 600 | ND | 100 | 152 | 500 | 307.8 | 500 | 79.6 | TNTC | >9676 |
| Ang 17 | 300 | ND | 400 | 500 | 100 | 140 | ND | ND | 15900 | ND | TNTC | ND | 700 | 520 | 8400 | 4838.4 | 200 | 130.8 | 2400 | nd |

**Legend:**

| | |
|---|---|
| App 1 | Appomattox River at Rt. 609 |
| App 2 | Appomattox River at Rt. 45 |
| Say 5 | Little Saylers Creek at Rt. 620 |
| Say 6 | Little Saylers Creek at Rt. 600 |
| Say 7 | Big Saylers Creek at Rt. 617 |
| Say 8 | Big Saylers Creek at Rt. 620 |
| Van 14 | Vaughan's Creek at Rt. 609 (Cabin Branch) |
| Buf 15 | Buffalo Creek at Rt. 648 |
| Gre 16 | Green Creek at Rt. 600 |
| Ang 17 | Angola Creek at Rt. 673 |
| MF | Membrane Filtration Fecal Coliform Count |
| MPN | Most Probable Number *E. coli* Count using Multiple Tube Fermentation |
| I-MPN | IDEXX Most Probable Number *E. coli* Count |

## Chlorophyll as an Endpoint in Determination of Attainment Prior to TMDL Development

Arthur J. Butt
Virginia Department of Environmental Quality
P.O. Box 10009
Richmond, VA 23240

### ABSTRACT

Virginia is engaged in development of quantitative criteria for state nutrient/sediment enrichment water quality standards. This includes the selection of target levels or "endpoints" of selected environmental parameters for criteria derivation. Such endpoints, if adopted would become the living resource targets for water quality management. These selected parameters or "endpoints" for criteria derivation include dissolved oxygen, chlorophyll *a*, and water clarity. The purpose of this paper was threefold: 1) to determine what unit(s) of measure should be used for chlorophyll; 2) to characterize the temporal and spatial patterns of chlorophyll levels in Virginia's tidal waters; and 3) to compare current levels to proposed target levels of chlorophyll using models.

Although the mean concentration levels for surface chlorophyll had been used for comparisons in the past, the median was determined more appropriate metric for status and trends by the Chesapeake Bay Data Analysis Workgroup. An annualized median chlorophyll concentration for Virginia's tributaries was 6.1 µg/l for the period 1985 through 1999. The tidal fresh regions had median concentrations below 5 µg/l whereas those in the higher salinity reaches of each tributary were between 5 and 10 µg/l.

Spatial and temporal patterns of median surface chlorophyll levels varied widely. Concentrations were highest (at or above 10 µg/l) in the transitional zones in the three estuaries for both seasons. The tidal fresh regions had higher concentrations during the summer months in the James and Rappahannock Rivers (above 9 µg/l). Concentrations were below 5 µg/l in the tidal fresh during the spring with lowest levels observed in the York and highest in James. The lower estuary of each tributary had median concentrations between 5 and 10 µg/l. Observed concentrations were then compared to target chlorophyll thresholds proposed by EPA.

Computer models predicted surface chlorophyll responses to various nutrient options. Based on a series of nutrient reduction scenarios, it was determined that both the Rappahannock and James basins would need Full Voluntary Program Implementation of nutrient controls to meet chlorophyll levels in the transitional zone. Nutrient reductions would have to meet conditions for the Limit of Technology (LOT) in the York Basin. Again this was due to excessive chlorophyll levels in the transitional zone.

**Keywords:** chlorophyll, water quality standards, TMDL

### INTRODUCTION

Water quality monitoring, modeling, and assessment studies have shown that Virginia's tidal waters are highly enriched (Dauer et al. 1998; VSNR 1999). Symptoms of nutrient enrichment include high chlorophyll levels, blooms of some potentially toxic algae, acutely and chronicly low dissolved levels in some southern tidal rivers and the Bay mainstem, and loss of submerged aquatic vegetation (SAV). These in turn, adversely affect the abundance and diversity of fishes and the quality of their food.

The Bay Program partners have committed to the goal of improving water quality in the Bay and its tributaries. While a voluntary approach has been the primary means to address the nutrient enrichment problems within the Bay and its rivers, regulatory action may be implemented. For example, EPA-Region III included Virginia's portion of the Chesapeake Bay and portions of several tidal tributaries on the 1998 federal Section 303(d) list of impaired waters. All lists of impaired waters are scheduled to develop a Total Maximum Daily Load. EPA's actions are coupled with the desire to have Virginia's waters removed from the impaired waters list prior to implementation of a TMDL for the watershed as a whole. Part of the de-listing process deals with development of quantitative criteria for state nutrient / sediment enrichment water quality standards. This requires states to set standards needed to maintain a designated use for those impaired water. Such water quality goals or "endpoints" for criteria derivation should be based on key environmental parameters.

Routinely, the parameters most frequently targeted are nitrogen and phosphorus. The Chesapeake Bay Program has identified three new parameters: dissolved oxygen, chlorophyll *a*, and water clarity. If adopted, these would become the living resource targets for water quality management. A classic measure of ecosystem health has been the chlorophyll content of the water column. Not only is it a useful expression of algal biomass, it is arguably the single most responsive indicator of nitrogen and phosphorus enrichment (Harding and Perry 1997). Therefore, determination of the level or levels of chlorophyll that correspond to adequate habitat for living resources is a useful and necessary exercise.

While current monitoring programs assess the status and trends of various indicators of the heath of the ecosystem, a more comprehensive characterization of chlorophyll concentrations within Virginia's tributaries is wanting. The purpose of this paper was threefold: a) to determine what unit(s) of measure should be used for chlorophyll, 2) to characterize the temporal and spatial patterns of chlorophyll levels in Virginia's tidal waters, and 3) to compare current levels to proposed target levels of chlorophyll.

## METHODS

Previous exercises performed for the purposes of Chesapeake Bay Program water quality modeling demonstrated that it was necessary to evaluate water quality conditions on seasonal time frames and over broad salinity regimes (Thomann et al., 1994). For this reason, comparisons were made among tributary regions defined by salinity (Tidal Fresh = 0-0.5 ppt, River Estuarine Transition = 0.5-5.0 ppt, and Lower Estuary > 5.0 ppt) as opposed to the use of single stations. Surface chlorophyll data from Virginia's three major tributaries during the period 1985-1999 were employed in this analysis. Both annualized and seasonal comparisons for the same fifteen-year period were employed. Spring (March through May) and summer (June through September) were chosen since concentrations peaked during these warmer periods. A frequency distribution of surface chlorophyll concentrations for the study period was asymmetrical, failing normality. Therefore, the median rather than arithmetic mean was employed. The geometric mean was found to be less than the median for most comparisons, but exceeded the median for some seasonal comparisons. The Kruskal-Wallis one way ANOVA on ranks was performed along with pairwise multiple comparisons employing the Dunn's method ($P < 0.05$) to test differences in the median values among treatment groups.

### Study Area Characteristics

Chesapeake Bay is a shallow estuary with an average depth of 8.4 m (Cronin 1971). The land area to water surface area ratio is 17:1 (17 units of land for every unit of water). As a result, the watershed not only provides major freshwater inputs but nutrients and organic material from the extensive land area. Despite the large volume of freshwater inflows, the median residence time for Bay water is around 116 days (Hagy and Boynton 2000). Land to water surface area ratios varied greatly between the three tributaries compared. Both James and York Rivers showed land to water surface area ratios well above the average for the Bay watershed (42:1 and 30:1, respectively; Table 1). In addition, the time required for water to flow through the tidal portions of a tributary varied from as little as 31 days (James) to over 53 days (Rappahannock).

Light and nutrients regulate the production of phytoplankton-derived organic matter within the Bay. Field studies and model simulations indic ated that nutrient supply usually was the most important process determining net primary production and that light tended to regulate the algal portion of production (Fisher and Butt 1994). Virginia's segment of the Bay was nitrogen limited, although both nitrogen and phosphorus limitation was observed in the tributaries, varying by season. The tidal fresh regions of all three rivers were strongly light limited (Haas and Webb 1998) were either nitrogen or phosphorus limited. Below Richmond, James River was nitrogen limited during the spring and summer. The tidal fresh region of the Mattaponi and Pamunkey Rivers within the York basin were phosphorus limited during the spring. While only the upper reaches of the tidal portions were phosphorus limited during the summer, the rest of the tidal York River was nitrogen limited. Rappahannock River was phosphorus limited during the spring but changed to nitrogen limitation during the summer. Sources of total annual nitrogen loads to the Bay were dominated by nonpoint source (45%) mostly from Susquehanna River, ocean input (28%), point source (19%), and atmospheric deposition directly to the Bay surface (7%) (Thomann et al., 1994).

## RESULTS

The annualized median chlorophyll concentration for Virginia's tributaries was 6.1 μg/l with the upper 75% quartile at 12.0 μg/l and lower 25% quartile - 3.1 μg/l (Figure 1). The median chlorophyll concentrations varied widely within and among the three tributaries (Table 2). The tidal fresh regions had medians below 5.0 μg/l, whereas those in the higher salinity reaches of each tributary were between 5 and 10.0 μg/l. Highest concentrations were recorded in the transitional zone. Of Virginia's three major tributaries, chlorophyll levels were highest in tidal fresh and transitional zones of James River and lowest in the lower estuary of this tributary.

Results of seasonal comparisons of surface median chlorophyll levels indicated that in the spring, levels were higher in the transitional and lower estuary regions of each tributary (Table 3). The only exception was in the transitional region of York River where median concentrations were 3.5 μg/l. Concentrations were below 5.0 μg/l in the tidal fresh reaches of all three tributaries with highest levels in James (4.8 μg/l). Summer median concentrations were higher than spring levels in the tidal fresh region of the Rappahannock and James Rivers (9.1 and 10.5 μg/l, respectively) (Table 4). There was no change in the seasonal median of York River. As for spring time, the transitional zones showed the highest values with concentrations in all three tributaries at or above 10.0 μg/l. The lower estuary of each tributary had lower median concentrations ( 5.0 - 10.0 μg/l).

Statistical differences in the median values among treatment groups were tested. Most all groups differed significantly both spatially and temporally. The only groups showing no statistically significant differences were median concentrations in the transitional regions of the Rappahannock and James during the spring and summer and the tidal fresh York. Median chlorophyll concentrations were similar in the lower estuaries in all three estuaries except for James River during the summer, which displayed the lowest concentration levels (5.7 μg/l).

A set of target concentrations for chlorophyll *a* levels has been proposed by the EPA's Chesapeake Bay Program. Concentrations believed to be supportive of living resource conditions were 5.0 μg/l for transitional and lower estuary (mesohaline / polyhaline) waters, and 10.0 μg/l for tidal fresh / oligohaline waters. Both spatial and temporal comparisons were considered in light of the proposed target / threshold levels. With one exception, the upper threshold (10.0 μg/l) was met in all of Virginia's tidal fresh waters (Table 2); only the James failed during the summer (Table 4). The highest concentrations were in the transitional zone greater than median annualized concentration of 8.8 μg/l except in the York River (Table 2). Only the spring tidal York River met the threshold level of 10.0 μg/l (Tables 3 and 4). Concurrently, the lower estuary regions had median chlorophyll concentrations greater than 5.0 μg/l but less than 10.0 μg/l.

When annualized medians were converted to a percent change from the thresholds, it was concluded that substantial reductions in chlorophyll concentrations would be necessary to reach a set of proposed thresholds in the meso- to polyhaline (transitional and lower estuary) regions of each tributary (Table 5). Highest reductions would have to occur in the transitional zone, about 43% for the three tributaries combined. Highest reductions would be needed in the James and lowest in the York.

## DISCUSSION

A set of target concentrations for chlorophyll *a* levels has been proposed by the EPA's Chesapeake Bay Program. Concentrations believed to be supportive of living resource conditions were 5 μg/l for transitional and lower estuary (mesohaline / polyhaline) waters, and 10 μg/l for tidal fresh / oligohaline waters. The median chlorophyll concentration of 6.1 μg/l for the lower Chesapeake Bay tributaries was about 20% above the threshold value of 5.0 μg/l for the polyhaline region of the Bay. A series of nutrient reduction scenarios were performed to test the response of surface chlorophyll concentrations to various reduction levels. To obtain the threshold surface chlorophyll concentrations in any section of a tributary, it was assumed that nutrient reductions would be needed throughout the basin (or watershed). Based on model studies conducted as part of Virginia's tributary strategy process, significant nutrient reductions would be necessary to reach the proposed threshold chlorophyll concentrations (Table 6). Based on the assumption that nutrient controls were implemented throughout the basin, both the Rappahannock and James basins would need Full Voluntary Program Implementation (FVPI) (Table 7). York would need to meet the Limit of Technology (LOT) reductions. Through such analyses, it became apparent that the transitional zone was the region of greatest concern. The Rappahannock and York Rivers met conditions in the tidal fresh and would meet thresholds under the Interim Bay Agreement scenario. However, James River would need FVPI throughout the tidal portions.

The lower estuaries of each tributary are strongly influenced from outside the region (Butt et al. 2000). Ocean inputs mediated much of the influence in the lower portion of James River while the ocean in conjunction with Bay waters dominated water quality conditions up to the higher salinity waters of the lower Rappahannock and York tributaries; i.e., these waters were nitrogen limited. Median surface chlorophyll concentrations observed in this salinity zone within each tributary ranged from a summer time low of 5.7 μg/l in James River to a spring time high of 9.0 μg/l in York River. Such overall model results suggest that further nutrient controls within each tributary would do little toward altering chlorophyll levels in the lower, higher salinity regions.

A major feature common to each tidal river was the presence of the turbidity maximum located in the transitional zone. It was within this system that physical factors favored the accumulation of suspended matter. Just below this zone was a region of chlorophyll maxima (Harding et al., 1986, Malone et al., 1986, Fisher et al., 1988; Fisher and Butt 1994). Here, the water column cleared and algal accumulation resulted in elevated chlorophyll levels. Like the turbidity maximum, this biologically active region responded to hydrology, moving seaward under high flows but staying downstream of the turbidity maximum. Because of the unique physical, chemical, and biological processes in this region of Virginia's estuaries, controlling chlorophyll concentrations may also prove very difficult.

Therefore, it may be concluded that a surface chlorophyll threshold of 5.0 μg/l would be very difficult if not impossible to obtain in the transitional and lower estuaries based on targeted reductions solely within a tributary. This is primarily due to the complex physical, chemical and biological processes acting in these regions. In order to meet the 10.0 μg/l threshold level for the tidal fresh, only James River required further controls. However, this was a basin with the largest land to water surface ratio of the basins compared. Therefore, background loadings of nutrients and organic matter delivered to the tidal James may differ significantly from that delivered to other Bay tributaries to the north. Additional study of this problem and solutions to reduce chlorophyll levels in this region of the river may be warranted.


## ACKNOWLEDGEMENT

The author wishes to thank Dr. Brown of VCU for her comments and suggestions.


## REFERENCES


Butt, A.J., L.C. Linker, J.S. Sweeney, G.W. Shenk, R.Batiuk and C. Cerco. 2000. Technical Tools Used in the Development of Virginia's Tributary Strategies: A Synthesis of Airshed, Watershed, and Estuary Model Results. Technical Summary Report, September.

Cronin, W.B. 1971. Volumetric, Areal, and Tidal Statistics of the Chesapeake Bay Estuary and its Tributaries. Chesapeake Bay Institute Special Report 20, Ref 71-2.

Dauer, D.M., H.G. Marshall, K.E. Carpenter, M.F. Lane, R.W. Alden, K.K. Nesius, L.W. Haas. 1998. Virginia Chesapeake Bay Water Quality and Living Resource Monitoring Programs" Executive Report, 1985-1996. January.

Fisher, T.R., L.W. Harding, D.W. Stanley and L.G. Ward. 1988. Phytoplankton, nutrients, and turbidity in the Chesapeake, Delaware, and Hudson estuaries. Estuarine Coastal Shelf Sci. 27:61-93.

Fisher, T.R. and A.J. Butt. 1994. The role of nitrogen and phosphorus in Chesapeake Bay anoxia. **In:** Perspectives on Chesapeake Bay, 1994: Advances in Estuarine Sciences, Chesapeake Research Consortium Publication 147.

Haas, L. and K. Webb. 1998. Resource limitation of phytoplankton in the Virginia Chesapeake Bay and tributaries using nutrient-addition bioassays. Virginia Institute of Marine Sciences, Special Scientific Report #137.

Hagy, J.D. and W.R. Boynton. 2000. Chapter V. Controls on hypoxia in Chesapeake Bay and its major tributaries. **In:** Ecosystem models of the Chesapeake Bay relating nutrient loadings, environmental conditions, and living resources. Final Report to USEPA/CBPO UMCES[HPL] Contribution #3218.

Harding, L.W., B.W. Meeson and T.R. Fisher. 1986. Phytoplankton production in two east coast estuaries: Photosynthesis-light functions and patterns of carbon assimilation in Chesapeake and Delaware Bays. Coastal Shelf Sci. 23:773-806.

Harding, L.W. and E. Perry. 1997. Long term increases of phytoplankton biomass in Chesapeake Bay, 1950-1994. Marine Ecology Progress Series. 157:39-52.

Malone, T.C., L.H. Crocker, S.E. Pike and B.W. Wendler. 1988. Influences of river flow on the dynamics of phytoplankton production in a partially stratified estuary. Mar. Ecol. Prog. Ser. 48:235-49.

Virginia Secretary of Natural Resources. 1999. "Annual Report on the Development and Implementation of Nutrient Reduction Strategies for Virginia's Tributaries to the Chesapeake Bay." Virginia Dept. Environmental Quality Report.

Thomann, R.V., J.R. Collier, A.J. Butt, E. Casman, and L.C. Linker. 1994. Response of the Chesapeake Bay Water Quality Model to Loading Scenarios. CBP/TRS 101/94.

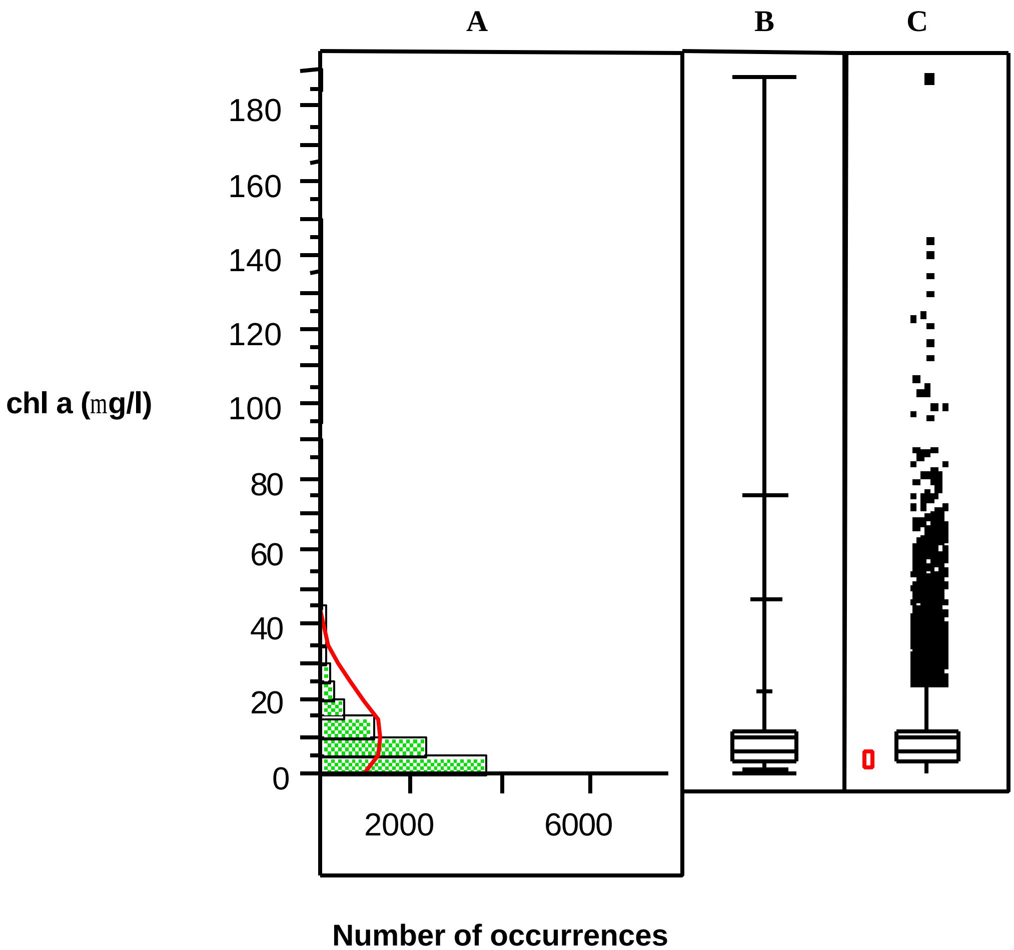



**Figure 1. A. Frequency distribution of median surface chlorophyll concentrations (mg/l) in Virginia's three major tributaries 1985-1999. B. Quartile Box Plot C. Outlier Box Plot**

**Table 1. River basin characteristics obtained from Cronin (1971) and Hagy and Boynton (1999).**

| River Basin | Watershed area ($km^2$) | Water surface area ($km^2$) | Land to water ratio | Depth (m) | Residence time (days) |
|---|---|---|---|---|---|
| James | 26,519 | 634 | 42:1 | 3.3 | 31 |
| York | 7769 | 256 | 30:1 | 4.3 | 35 |
| Rappahannock | 6940 | 401 | 17:1 | 4.8 | 53 |

**Table 2. Annualized median surface chlorophyll concentrations (mg/l) by tributary and salinity regime, 1985-1999. Asterisk "*" indicates that for York River, stations in Mobjack Bay (WE) were included in the lower estuary comparison.**

| Section / Basin | Rappahannock | York* | James | VA Tributaries |
|---|---|---|---|---|
| Tidal Fresh | 4.5 | 3.1 | 4.8 | 3.6 |
| Transitional | 8.9 | 8.2 | 10.0 | 8.8 |
| Lower Estuary | 7.1 | 7.8 | 5.8 | 7.0 |

**Table 3. Median spring (March through May) chlorophyll concentrations (mg/l) by tributary and salinity regime, 1985-1999.**

| Section / Basin | Rappahannock | York | James |
|---|---|---|---|
| Tidal Fresh | 3.1 | 3.0 | 4.8 |
| Transitional | 11.6 | 3.5 | 11.4 |
| Lower Estuary | 8.7 | 8.5 | 8.8 |

**Table 4. Median summer (June through September) chlorophyll concentrations (mg/l) by tributary and salinity regime, 1985-1999.**

| Section / Basin | Rappahannock | York | James |
|---|---|---|---|
| Tidal Fresh | 9.1 | 3.0 | 10.5 |
| Transitional | 10.4 | 10.0 | 11.4 |
| Lower Estuary | 7.7 | 7.7 | 5.7 |

**Table 5. Percent improvements in median chlorophyll concentrations required to meet proposed chlorophyll threshold of 10 mg/l.**

| Section / Basin | Rappahannock | York | James | VA Tributaries |
|---|---|---|---|---|
| Transitional | 44% | 39% | 50% | 43% |
| Lower Estuary | 30% | 36% | 14% | 29% |

**Table 6. Model scenario results based on changes to summer surface chlorophyll concentrations for Virginia tributaries.**

| Section / Basin | Rappahannock | York | James |
|---|---|---|---|
| Tidal Fresh | Met | Met | FVPI |
| Transitional | FVPI | LOT | FVPI |
| Lower Estuary | Interim Bay Agreement | Interim Bay Agreement | FVPI |

**Table 7. Descriptions of model scenario assumptions employed to evaluate nutrient reduction strategies. Abbreviations are: PS (point source), TN (total nitrogen), TP (total phosphorus), NPS-Ag (non-point source from agriculture), and BMPs (best management practices).**

| Model Scenario | Scenario Description |
|---|---|
| Interim Bay Agreement. / Tributary Strategies Above | Virginia's lower tributaries at an interim 40% nutrient reduction run for 10 years using 1985-94 flows. Tributary Strategy nutrient reductions applied in the Potomac River and all basins to the north including Maryland and Pennsylvania. |
| Full Voluntary Program Implementation (FVPI) | Full voluntary program implementation throughout the entire watershed. PS concentrations of 5.5 mg/L TN and 0.5 mg/L TP with flows projected to 2000. NPS-Ag assumes 75% cropland conservation till, 25% conventional till, 10% forest buffers, BMPs to animal wastes (80%), streambank protection (15%), nutrient management (75%), & septic connections (50%). |
| Limit of Technology (LOT) | The maximum practical level of implementation given unlimited resources and 100% land application based on "do everything, everywhere" using current available technologies throughout the entire watershed. Assumes PS concentrations of 3.0 mg/L-TN and 0.075 mg/L-TP with flows projected to 2000. NPS-Ag assumes 75% cropland conservation till and full forest buffers, 100% BMPs to animal wastes, streambank protection, nutrient management, and septic connections. |

# El Niño-La Niña Events and Water Resources of Lake Okeechobee, Florida

Hongsheng Cao
Department of Geology, Florida State University, Tallahassee, Florida 32306
E-mail: cao@gly.fsu.edu

## ABSTRACT

El Niño and La Niña events are responsible for the heavy rainfall and dry conditions in the Lake Okeechobee basin. Some El Niño-La Niña events resulted in severe flood events or drought events in the past several decades. The precipitation in the Lake Okeechobee basin is partially controlled by the El Niño-La Niña events. During El Niño years, strong rainfall and flood events are expected; whereas in La Niña years, weak rainfall and dry conditions occur. The future local and regional annual precipitation, the lake water level, and flood-drought events can be predicted if El Niño-La Niña events and their relative strengths are accurately forecasted.

The study contributes to a better understanding of the effects of El Niño/La Niña events on the water resources of the Lake Okeechobee basin. It is helpful to realize flood and drought potential and to minimize flood and drought damages when the future El Niño-La Niña years approach.

**Keywords:** El Niño, La Niña, precipitation, lake level, Lake Okeechobee.

## INTRODUCTION

El Niño-La Niña events warm or cool the waters in the equatorial central Pacific Ocean slightly and produce a warm or cold ocean current that flows southward along the coast of northern Peru in El Niño-La Niña years, but their effects are not limited to Peru locally and South America regionally.

Global changes in precipitation, streamflow, and related flood-drought events associated with El Niño events have been documented worldwide. Quinn et al. (1987) discussed the El Niño occurrences and the relative strengths of the El Niño events over the past four and a half centuries. Annual natural discharge of the River Murray-Darling River system of Southeastern Australia is often inversely related to sea surface temperature (SST) anomalies in the eastern topical Pacific Ocean (Simpson et al. 1993). Dry conditions in Australia tend to be associated with El Niño and below-normal rainfall and streamflow are consistently identified in the El Niño years (Chiew et al. 1998). Eltahir (1996) realized that 25% of the natural variability in the annual flow of the Nile River is associated with El Niño events. By analyzing the sea surface temperature of the Pacific Ocean and the streamflow of the Nile River, Eltahir developed a flood prediction procedure to improve the prediction of Nile floods. Waylen and Caviedes (1990) investigated the properties of annual and winter precipitation totals and streamflow characteristics in the Aconcagua River basin of Chile to identify flood- and drought-generating processes and their possible linkage to the El Niño Southern Oscillation phenomenon. Amarasekera et al. (1997) examined the relationship between the annual discharges of the Amazon, Congo, Paraná, and Nile rivers and the sea surface temperature (SST) anomalies of the eastern and central equatorial Pacific Ocean, an index of El Niño-Southern Oscillation (ENSO). The strongest relationship between El Niño events and extreme drought years is found in the northwest United States and a strong relationship is also noticed between dry conditions and La Niña events in the southern United States (Piechota and Dracup 1996). Sun and Furbish (1997) showed that El Niño and La Niña events are responsible for up to 40% of annual precipitation variations and up to 30% of river discharge variations in Florida.

The cold La Niña events have received less attention than the warm El Niño events because the cold La Niña phase is less distinct and causes less catastrophe than the warm El Niño phase (Dracup and Kaya 1994). The precipitation anomalies associated with La Niña events are opposite in sign to the precipitation anomalies during the El Niño events. A strong relationship between streamflow and the cold La Niña events has been recognized in four regions of the United States: the Gulf of Mexico, the Northeast, the North Central, and the Pacific Northwest (Dracup and Kahya 1994). They concluded that the sign of the seasonal streamflow anomaly associated with the La Niña events is the opposite of that associated with the El Niño events.

Lake Okeechobee provides South Florida with a wide range of benefits as a primary reservoir, natural navigational waterway, and important environmental resources. It supports several endangered species and a valuable sports fishing industry. It is the drinking water supply for the people who live in the vicinity of the lake, the backup source of water for millions of people in South Florida, and an irrigation supply for the extensive agricultural industry. It is also a source of water for the remaining Everglades system which includes Water Conservation Areas (WCA) and the Everglades National Park. The lake is also a major component of the Central and Southern Florida Flood Control System.

The threats of flood and drought, the pollution of the lake water, and the special value of the lake make the evaluation of the relationship among precipitation, lake level, and El Niño-La Niña events necessary. No special effort has been made previously to provide information on the correlation among precipitation, flood-drought events, and El Niño-La Niña in the Lake Okeechobee basin, though the lake is known to experience frequent flood-drought cycles with a varied duration and intensity. This paper will investigate the annual regional precipitation, lake level, and flood-drought events in Lake Okeechobee and how these factors are related to the El Niño-La Niña events in the past fifty years.

**El Niño and La Niña Events**

The interaction between atmosphere and sea surface in the equatorial Pacific Ocean causes an oscillation of temperature and pressure between the eastern equatorial Pacific Ocean and the western equatorial Pacific Ocean. This phenomenon is collectively referred to as the El Niño-La Niña Southern Oscillation (ENLNSO), where ‘El Niño-La Niña’ refers to the temperature oscillation in the eastern equatorial Pacific Ocean, and 'Southern Oscillation' refers to the pressure fluctuation between the east equatorial Pacific Ocean and west equatorial Pacific Ocean.

During El Niño events, unusually a high atmospheric pressure develops at sea surface in the western equatorial Pacific Ocean regions, and unusually low sea level pressures develop in the southeastern equatorial Pacific Ocean. Weakening trade winds allow the warmer water from the western Pacific Ocean to flow toward the eastern Pacific Ocean. El Niño refers to the irregular increase in sea surface temperatures from the coasts of Peru and Ecuador to the equatorial central Pacific Ocean. It is an ENLNSO warm event or the warm phase of ENLNSO, in which warm eastern and central equatorial Pacific sea surface temperature (SST) positive anomalies exist. El Niño is just a part of the result of an equatorial Pacific Ocean oriented cycle that occurs irregularly at intervals of three to five years. The El Niño event itself typically lasts about 12 to 18 months.

The cold La Niña events sometimes (but not always) follow El Niño events. During La Niña events, tendencies for unusually low atmospheric sea surface pressures in the west Pacific Ocean and high atmospheric sea surface pressures in the east Pacific Ocean are linked to periods of anomalously cold equatorial Pacific sea surface temperatures, which is referred to as La Niña. The term La Niña is used to describe those times of cold eastern and central equatorial Pacific SST negative anomalies, which sometimes are referred to as ENLNSO cold event or cold phase of ENLNSO.

Not all El Niño/La Niña events have the same strength because the atmosphere can not always react in the same way from one El Niño/La Niña event to another. In the past there have been some strong, moderate, and weak El Niño/La Niña occurrences. A list of El Niño and La Niña years is provided by the National Center for Environmental Prediction (NCEP) (Table 1 and Table 2). El Niño/La Niña may be getting stronger as time goes on and the El Niño event of 1990-1995 supports this assertion. The El Niño events have become more frequent in recent years than they ever did in the past. Also, the events in the 1990s have been "strong" compared to former recorded occurrences. Since March 1997, SSTs in the central and eastern equatorial Pacific have been higher than normal. The sea surface temperature for September 1997 was the highest in the last 50 years.

**Hydrogeology**

The most prominent topographic feature in southern Florida is the Lake Okeechobee-Everglades basin (Figure 1). Lake Okeechobee is the central component of South Florida's interconnected Kissimmee River-Lake Okeechobee-Everglades ecosystem. Seas covered the area during the Pleistocene interglacial stages and marine calcareous materials were deposited. During glacial stages the seas retreated and fresh-water marls were deposited in shallow depressions. The Lake Okeechobee-Everglades basin probably was a lake or a swamp during a part of each glacial stage (Meyer 1971).

Lake Okeechobee, within the Coastal Lowlands of Florida, was formed on an ancient sea floor by recession of the sea during or prior to Pleistocene time (about 6000 years ago) (Joyner 1974). Lake Okeechobee is a shallow saucer-like depression within the broad flat plain, the northern half of the lake is almost completely surrounded by sandy prairies that range in altitude from 6.1 to 9.1 m above mean sea level. The southern half of the lake lies in the Everglades where the altitude of land surface ranges from 4.3 to 6.1 m above mean sea level (Meyer 1971). The deepest point in the lake is slightly below mean sea level. The temperature of the water ranges from 16 °C in the winter to about 32 °C in the summer.

Lake Okeechobee is the largest fresh-water lake in Florida and is the second largest fresh-water lake wholly within the United States, covering an area of approximately $1.9 \times 10^9$ $m^2$ with an average depth of 2.6 m when it is at a stage of 4.7 m above mean sea level. The lake is nearly circular and is 56 km long from north to south and 48 km wide from east to west. The shoreline, approximately 170 km long, is rimmed by the Hoover Dike. Maximum water levels ranged from 6.1 to 6.4 m above mean sea level (VanArman et al. 1998). Approximately 4300 million $m^3$ of fresh water are stored in the lake at average stage of 4.3 m.

Historically, water entered the lake primarily from rainfall and flow across adjacent wetlands. Runoff from an area of about $14.7 \times 10^9$ $m^2$ drains southward into the lake. Largest of the inflowing streams is the Kissimmee River which discharges about $21 \times 10^8$ $m^3$ into the lake annually. Other principal inflowing streams or canals are Fisheating Creek and Taylor Creek. Most of the water enters the lake from rainfall, local runoff, and discharge from the Kissimmee River, Fisheating Creek, and Taylor Creek. Therefore, the principal water inputs to Lake Okeechobee are surface inflows and rainfall (Figure 2). Rainfall is responsible for almost as much of the total water input as surface inflow. The principal water outputs are surface outflows, evaporation and groundwater seepage (Figure 3). Evaporation is actually responsible for a larger proportion of total water output than surface outflow with groundwater seepage, a minor component of the overall water balance. Due to the large surface area, about 70 percent of this water is lost to evaporation annually. Because evaporation is a major component of total water outflow from Lake Okeechobee, determination of hydraulic residence time for the lake must be consistent with the intended use of this parameter. Hydraulic residence time can be defined as lake volume divided by the sum of surface outflows, evaporation and groundwater seepage. Annual hydraulic residence time ranges between approximately 0.8 and 1.5 years if we consider both seepage and evaporation (Figure 4). Mean hydraulic residence time represents the residence time of water in the lake and is the approximate residence time for describing the water budget. There is substantial variation in lake volume, reflecting sensitive responses to changes in precipitation and evaporation and there are corresponding changes in lake depth and lake surface area, respectively (Figure 5).

Three major aquifer systems have been identified in the study area. The Upper Florida aquifer system is a deep, regionally extensive, artesian limestone system with generally high transmissivity. The Upper Floridan aquifer system is the primary source of water supply and is used as a source of supplemental irrigation water. This system receives direct recharge along a structural high, which occurs in the central part of the state. From these recharge areas, the Upper Floridan aquifer dips southward and becomes confined by the clays and silts of the younger Hawthorn Group. The potentiometric surface is highest at the point of primary recharge area and decreases to the south. The intermediate aquifer system is developed within the Hawthorn Group, which is a major confining unit, separating the surficial aquifer system from the Upper Floridan aquifer. These leaky artesian aquifers consist of moderately transmissive sandstone or sandy and shelly limestone beds and produce water of fair to poor quality. The surficial aquifer system consists of sands, sandy and shelly limestone, sandstone, and silts and contains water table and semi-confined aquifers.

**Precipitation**

The climate of the Lake Okeechobee region is subtropical and is characterized by warm and humid summers and moderately cool winters (Meyer 1971). The average annual temperature during 1931-1995 was 22.6 °C. Mean monthly temperatures ranged from 18.3 °C in December to 27.2 °C in August (Figure 6). The lowest temperature recorded was -4.4 °C in January 1940 and the highest was 37.8 °C in July 1931. The region experiences long, warm, and humid summers and mild winters. The Lake Okeechobee region has its distinct wet and dry seasons (VanArman et al. 1998). Rainfall varies considerably within this region. But average annual rainfall in the Lake Okeechobee region is 128.3 cm. The driest month is December, when average monthly rainfall ranges from less than 3.2 cm to 6.4 cm. The wettest month is September, when average rainfall ranges from 24 cm over the adjacent land areas to 15.2 cm over Lake Okeechobee itself. Rainfall is seasonal with about 75 percent of the yearly total accumulative from May through October with most water supply replenishment occurring at this time. In extremely dry years, this can translate to low water availability during the late winter and spring. Mean monthly rainfall ranges from 4.1 cm in December to 21.6 cm in September (Figure 6). The maximum daily rainfall recorded was 27.7 cm in November 1931, and the maximum monthly rainfall there was 61.2 cm in June 1945. Rainfall during 1963-1965 was below normal despite severe weather conditions that accompanied the passage of three hurricanes. Rainfall during 1966 was above average. Although rainfall averages 128.3 cm/yr in the watershed, wide variations occur between locations and from year to year (Figure 7 and Figure 8). Substantially less rainfall occurs over Lake Okeechobee than occurs over adjacent land areas (VanArman et al. 1998).

Normally, rainfall is greatest from June through September and least from November through January. Much of the rainfall that occurs during the wet season results from convection due to differences in temperature between land and water. Summer rainfall is associated with localized thunderstorm activity. Rainfall during spring and summer is unevenly distributed. In winter, fronts bring sweeping bands of rain and cooler temperatures. Frontal rains are usually more evenly distributed and are of longer duration than summer rainfall. Because evaporation and plant transpiration are significantly lower during the winter, these frontal rains are important for recharging groundwater. The mean seasonal rainfall for spring, summer, fall, and winter is 25.4 cm, 50.8 cm, 30.48 cm, and 22.86 cm, respectively. About half of the average annual rainfall falls from June through September. A shorter rainy season occurs from late February to late April. Most summer rain comes from short duration afternoon or early evening local shower and thunderstorms. These rainstorms occasionally produce 5.08-7.62 cm of rain in 1 to 2 hours.

The winter and early spring rains are generally associated with large-scale weather frontal developments and are occasionally of long duration, from 12 to 36 hours.

The combination of concentrated periods of rainfall and flat terrain produces a continually swampy, flooded condition throughout much of the region during the wet season, a subtropical characteristic which, for a long time, made the Lake Okeechobee region a less-than desirable spot for human settlement. The most severe floods in the watershed are associated with storms, which produce widespread distribution of rainfall for several days. Although flooding can occur in all seasons, maximum annual surface water stages occur most frequently from February through April as a result of a series of frontal-type rainfall events over the watershed. The area is also subject to summer and fall tropical storms and occasional hurricanes, which may occur from June through November. Storms and hurricanes are the major causes of widespread, excessive rainfall and associated flooding. Since 1948 three major floods and several minor floods have occurred in the watershed. Three extreme floods have occurred in the last 55 years (1948, 1959, and 1973).

Drought conditions, as ordinarily defined in humid areas, exist when there is insufficient moisture in the soil to maintain plant life or when precipitation is insufficient to meet the needs of established human activities (Pride and Crooks 1962). Spring and summer droughts of varying severity occur, but not in any predictable patterns. Severe droughts usually occur during the fall and late spring. November is normally the driest month of the year. A major drought in the Lake Okeechobee region, occurred between 1980-1981, and caused a natural drawdown of the lake to a record low level (Jones 1983). The year 1981 was highly unusual with respect to the Lake Okeechobee water supply. The drought that began in 1980 continued into 1981 and water inputs to the lake were very small. The lake stage declined until it reached a historic low of 2.97 m above mean sea level on July 29, 1981. Droughts occurred between 1931 and 1945, bringing saltwater intrusion along the coasts and causing extensive fires in the Everglades. The area occasionally experiences extended periods of below average rainfall, such as occurred during the drought of 1988-1991 (VanArman et al. 1998).

The duration of the period of rainfall deficiency is one of the more important factors that influence the severity of a drought and is probably the most significant cause of extreme drought in the watershed. Variation in rainfall has an important influence on droughts. Rainfall is not uniformly distributed nor is it properly timed for optimum utilization. Drought could occur even though the rainfall for a given period was higher than the average, when the distribution is such that most of the rainfalls during a short period. Records of rainfall and streamflow provide data for studying droughts by defining the area covered, the severity, the frequency, and the duration of drought.

The most severe recorded drought event occurred in the watershed during 1954-56. The drought resulted from rainfall deficiencies in amounts ranging from 17.8 to 27.9 cm during each of the 3 years. Annual rainfall totals in the watershed for 1954, 1955, and 1956 were 116.6, 107.5, and 107.8 cm, respectively. The statewide runoff during 1955 was estimated to be 15.2 cm, compared with 35.6 cm for an average year. The 1954-56 drought caused critical shortages of surface water in the watershed, because its 3-year duration was an event of unusual occurrence. The rainfall pattern shows that the 1954-56 drought in the watershed ranks as the most severe in the past 55 years. The drought of 1973-77 also was widespread; the recurrence interval for that drought ranged from about 15 to 35 years, depending on location. Dry conditions occurred in 1938-1943 (except 1941), 1961-1963, 1967-1968, 1977-1978, and 1989-1990 though the watershed did not experience drought in some of these periods. A period of record-low groundwater levels in 1989-1990 throughout most of the watershed was observed (Fernald and Purdum 1998).

## DISCUSSION

Since 1970 El Niños have been occurring every 2.2 years, up from every 3.4 around 1870, every 4.5 year around 1750, and every six years in the late 1600's (Dunbar et al. 1994). That means flooding frequency will be increased in years to come. Typically, El Niño occurs more frequently than La Niña. Though La Niña events occur after some (but not all) El Niño events, the probability of drought will also be increased.

On the basis of the SST anomalies and the June-November Southern Oscillation Index (SOI) selection criteria (SOI ≤ -0.50) in the eastern Pacific Ocean, the El Niño composite years should include the years listed in Table 1. Water warmer than usual in the eastern Pacific during 1986-1988, 1991-1992, 1993, 1994 and 1997-1998 is well documented. The storm of 1982-1983 was perhaps the strongest occurrence of El Niño in the century. During the 1986-1988 El Niño, the warm water penetrated eastward in the Spring of 1987. The 1990-1995 El Niño event is believed to be the longest recorded incidence in history. It is unusual for El Niño events to occur in such rapid succession. In the 1991-1992 El Niño year, the warm water penetrated towards the east in the northern hemisphere in the spring of 1992. Strong El Niño condition occurred in December 1997 with warm water extending all along the equator. El Niño events vary in strength. The 1997-1998 El Niño is unusually strong. The strength of the 1997 El Niño event could equal or surpass that in 1982-1983, making it the strongest El Niño this century. The precipitation in the El Niño years is above the annual mean precipitation. In the El Niño years, such as 1953, 1957-1959, 1965-1966, 1972-1973, 1976, 1983, 1986, 1988, 1991-1992, and 1995, the precipitation in More Heaven and Ft. Myers is above annual precipitation (Table 1, Figure 7, and Figure 8). The high rainfall in 1964 and 1985 are the two cases, which are not correlated with the El Niño event. The flood of 1959 resulted from the high precipitation, which occurred in an El Niño year. The 1973 flood can be explained by the 1972-1973 El Niño event. The storm of 1994-1995 is now believed to be nearly as strong as the 1982-1983 storm. This storm is the most recent until now, and supposedly was hard to distinguish with the storm of 1982. The high rainfall and related flood in 1947-1948 is possibly correlated with an El Niño event (Figure 7 and Figure 8).

On the basis of the sea surface temperature (SST) anomaly and the June-November Southern Oscillation Index (SOI) selection criteria (SOI ≥ +0.50) in the eastern Pacific Ocean, La Niña composite years include 1950-1951, 1954-1956, 1964-1965, 1970-1971, 1973-1976, 1983-1985, 1988-89, 1996-97 (SOI = +0.47), and 1998-1999 (Table 2). The very cool water (negative anomalies) in the Eastern Pacific occurred in 1988-1989, which is a strong La Niña event. The somewhat less cool water in 1995 has been recorded. 1995-1996 was a weaker La Niña year. Strong La Niña conditions occurred during December 1998: the Eastern Pacific was cooler than usual, and the cool water extended farther westward. La Niña events also vary in strength. For example, the 1987 La Niña was stronger than the 1995 La Niña. During most of the La Niña years, the rainfall in More Heaven and Ft. Myers is below the annual precipitation (Table 2, Figure 7, and Figure 8), for example, in 1950, 1954-1956, 1973-1975, 1989-1990. The lowest 1954-1956 precipitation and the 1954-1956 severe drought were matching the long 1954-1956 La Niña event.

Lake stage displays considerably variation (Figure 9), and several years of prolonged high lake stages have occurred and are correlated to El Niño years. Then low water level conditions followed and seemed to response to La Niña events. Moderate to strong El Niño events occurred during 1957-1959, 1965-1966, 1968-1969, 1972-1973, 1982-1983, 1986-1988, 1990-1995, and 1997-1998 (Table 1). In longer or stronger El Niño years, Lake Okeechobee has a relatively higher water level (Figure 9). In some short or week El Niño years, lake levels seem to be not sensitive to those events. During 1950-1951, a long moderate La Niña event was followed by a very short week El Niño event, so we have a lower lake level during 1950-1951. Other longer La Niña events occurred during 1954-1956, 1964-1965, 1970-1971, 1973-1976, 1983-1985, 1988-1989, and 1995-1996 (Table 2). During those years, lower lake levels were generally observed (Figure 9). During 1973-1978, a longer and stronger La Niña event (1973-1976) is followed by three week El Niño events, the La Niña's effects on the lake levels are obvious though they are weakened by 1972-1973 and 1976-1978 El Niño events. If a short or week El Niño (or La Niña) event is followed by a short or week La Niña (or El Niño), their effects on the lake level are canceled out, such as during 1963-1964. During 1973-1976, 1984-1985, and 1988-1989 La Niña years, lake water volume, lake surface area, and lake depth were lower than usual, whereas during 1982-1983, 1986-1988, and 1990-1993 El Niño years, lake water volume, lake surface area, and lake depth were higher than usual (Figure 5). On the basis of the reasonable correlation between El Niño-La Niña events and the lake water level, lake water volume, lake surface area, and lake depth, I suggest that El Niño events also occurred during 1933-1935, 1936-1938, 1940-1942, 1946-1947, and 1979-1980, and La Niña events occurred in 1935, 1938-1939, 1943-1944, and 1981 (Figure 5 and Figure 9).

Therefore, annual precipitation, lake level, and lake volume can be predicted reasonably well from the SST anomalies and the Southern Oscillation Index (SOI). This can provide some guidance for the water management policy and planning. It is helpful to realize flood and drought potential and flood and drought hazards in land use planning and for management decisions concerning flood plain utilization when the future El Niño-La Niña years approach. This provides the basis for further study and planning and the solutions to minimize future flood and drought damages during the El Niño-La Niña years. The study of correlation between the Lake Okeechobee flood-drought events and El Niño-La Niña events would improve flood routing and forecasting and would also be an important contribution to the judicious implementation of land use planning.

## CONCLUSIONS

The precipitation in the Lake Okeechobee basin is partially controlled by the El Niño-La Niña events. During El Niño years, strong rainfall is expected and flood events result; whereas in La Niña years, weak rainfall and dry conditions occur. During El Niño years, lake water volume, lake surface area, and lake level are much higher than those during La Niña years. On the basis of the recorded precipitation, lake volume, lake surface area, and lake depth, several unidentified El Niño and La Niña events are suggested. Several severe flood and drought events historically correlated with El Niño and La Niña events. Therefore, the El Niño-La Niña events of the Pacific increase the chances of high rainfall in the Lake Okeechobee area, and result in destructive flooding, drought, and devastating forest fires. The annual precipitation, lake level, and flood and drought events can be predicted from the sea surface temperature (SST) anomalies, the Southern Oscillation Index (SOI), and the resulted El Niño-La Niña events.

The results of this study contribute to a better understanding of the effects of El Niño/La Niña events on the environments of the Lake Okeechobee basin, and may be used for future planning and forecasting in the area. The developing methods can predict droughts and excessive rainfall periods (floods) on the basis of the analyses of El Niño-La Niña events and relationships to precipitation trends in south Florida.

## ACKNOWLEDGMENTS

I gratefully acknowledge the financial supports from the Florida Department of Environmental Protection (FDEP) and Department of Geological Sciences at the Florida State University. The author is indebted to Andy Feldman for reviewing an earlier version of the manuscript and suggesting significant improvement.

## REFERENCES


Amarasekera, K. N., R. F. Lee, E. R. Williams, and E. A. B. Eltahir. 1997. ENSO and the natural variability in the flow of tropical rivers. *Journal of Hydrology*, *200*, 24-39.

Berndt, M. P., H. H. Hatzell, C. A. Crandall, M. Turtora, J. R. Pittman, and E. T. Oaksford. 1998. *Water Quality in the Georgia-Florida Coastal Plain, Georgia and Florida, 1992-96*. Denver, CO: U.S. Geological Survey.

Ceryak, R., M. S. Knapp, and T. Burnson. 1983. *The geology and water resources of the Upper Suwannee River basin.* Tallahassee, FL: Florida Geological Survey.

Chiew, F. H. S., T. C. Piechota, J. A. Dracup, and T. A. McMahon. 1998. El Niño/Southern Oscillation and Australian rainfall, streamflow and drought; links and potential for forecasting. *Journal of Hydrology*, *204*, 138-149.

Crane, J. J. 1986. *An investigation of the geology, hydrogeology, and hydrochemistry of the Lower Suwannee River Basin.* Tallahassee, FL: Florida Geological Survey.

Dracup, J. A., E. Kahya. 1994. The relationships between U.S. streamflow and La Niña events. *Water Resources Research*, *30*, 2133-2141.

Dunbar, R. B., G. M. Wellington, M. W., Colgan, and P. W. Glynn. 1994. Eastern Pacific sea surface temperature since 1600 A.D.; the delta $^{18}$O record of climate variability in Galapagos corals. *Paleoceanography*, *9*, 291-315.

Dunne, T., L. B. Leopold. 1978. *Water in Environmental Planning*. New York: Freeman and Company.
Eltahir, E. A. B. 1996. El Niño and the natural variability in the flow of the Nile River. *Water Resources Research*, *32*, 131-137.

Piechota, T. C., J. A. Dracup. 1996. Drought and regional hydrologic variation in the United States: Associations with the El Nino-Southern Oscillation. *Water Resources Research*, *32*, 1359-1373.

Pittman, J. R., H. H. Hatzell, and E. T. Oaksford. 1995. *Spring contributions to water quantity and nitrate loads in the Suwannee River during base flow in July 1995*. Denver, CO: U. S. Geological Survey.
Pride, R. W., J. W. Crooks. 1962. *The drought of 1954-56, its effect on Florida's surface-water resources.* Tallahassee, FL: Florida Geological Survey.

Raulston, M., C. Johnson, K. Webster, C. Purdy, and R. Ceryak. 1998. Suwannee River Water Management District. In E. A. Fernald and E. D. Purdum (Eds.), *Water Resources Atlas of Florida* (pp. 194-215). Tallahassee, FL: Institute of Science and Public Affairs, Florida State University.

Quinn, W. H., V. T. Neal, and S. E. Antunez de Mayolo. 1987. El Niño occurrences over the past four and a half centuries. *Journal of Geophysical Research*, *92*, 14,449-14,461.

Simpson, H. J., M. A. Cane, A. L. Herczeg, S. E. Zebiak, and J. H. Simpson. 1993. Annual river discharge in Southeastern Australia related to El Niño-Southern Oscillation forecasts of sea surface temperatures. *Water Resources Research*, *29*, 3671-3680.

Sun, H., D. J. Furbish. 1997. Annual precipitation and river discharges in Florida in response to El Niño- and La Nina-sea surface temperature anomalies. *Journal of Hydrology*, *199*, 74-87.

Waylen, P. R., C. N. Caviedes. 1990. Annual and seasonal fluctuations of precipitation and streamflow in the Aconcagua River basin, Chile. *Journal of Hydrology*, *120*, 79-102.

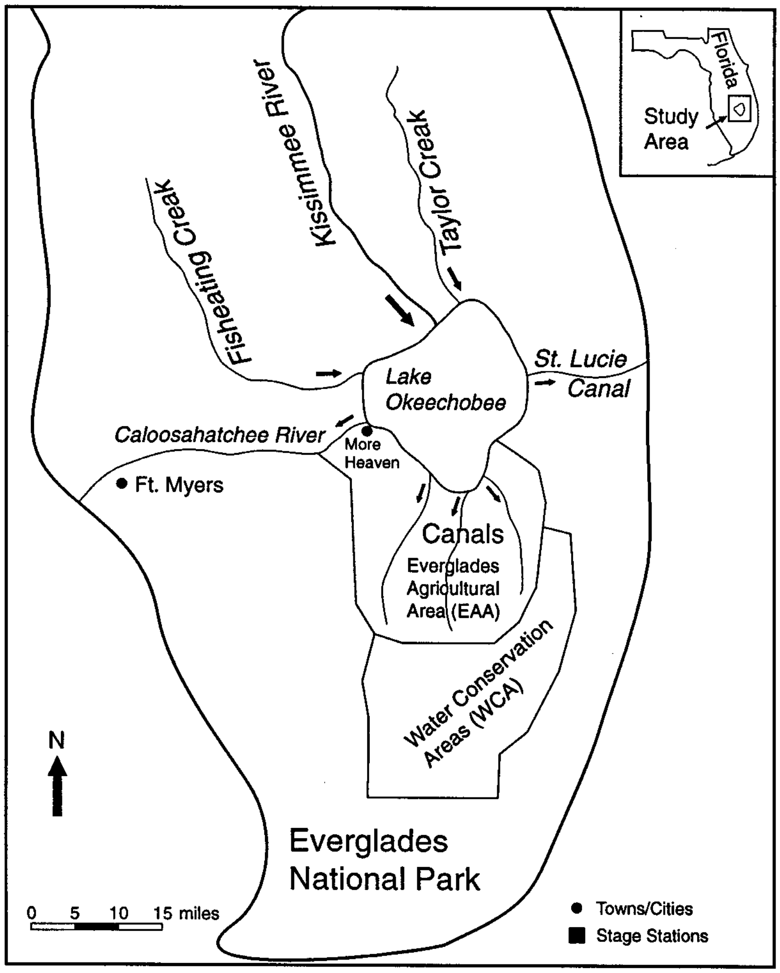



**Figure 1. Location of Lake Okeechobee Basin, Florida.**

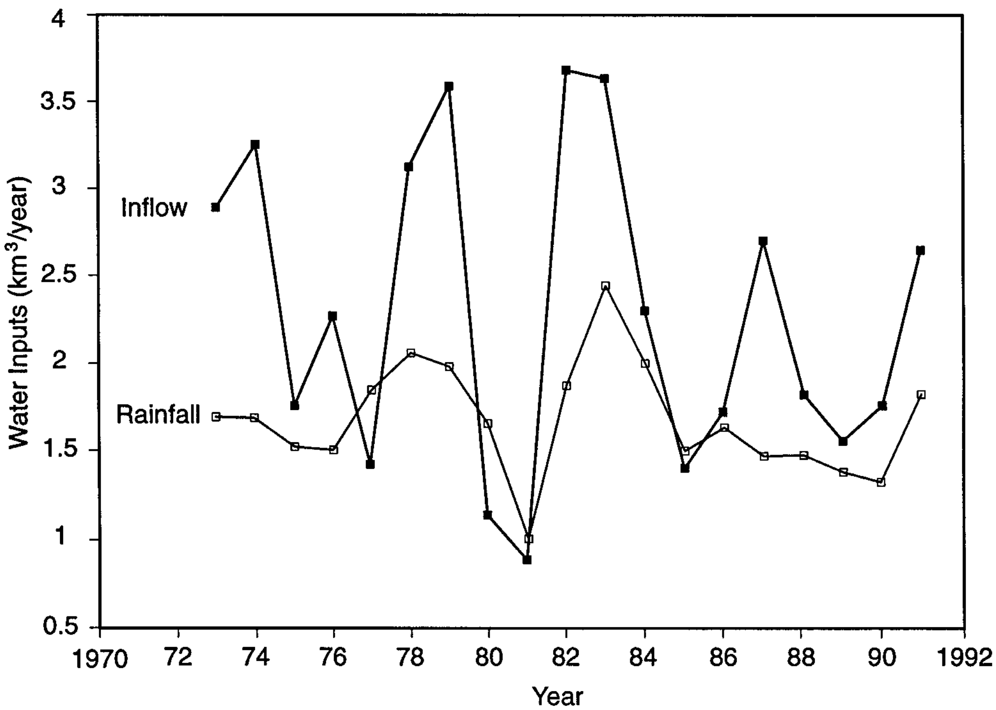



**Figure 2. Annual Mean Lake Water Inputs as Surface Inflow and Rainfall.**

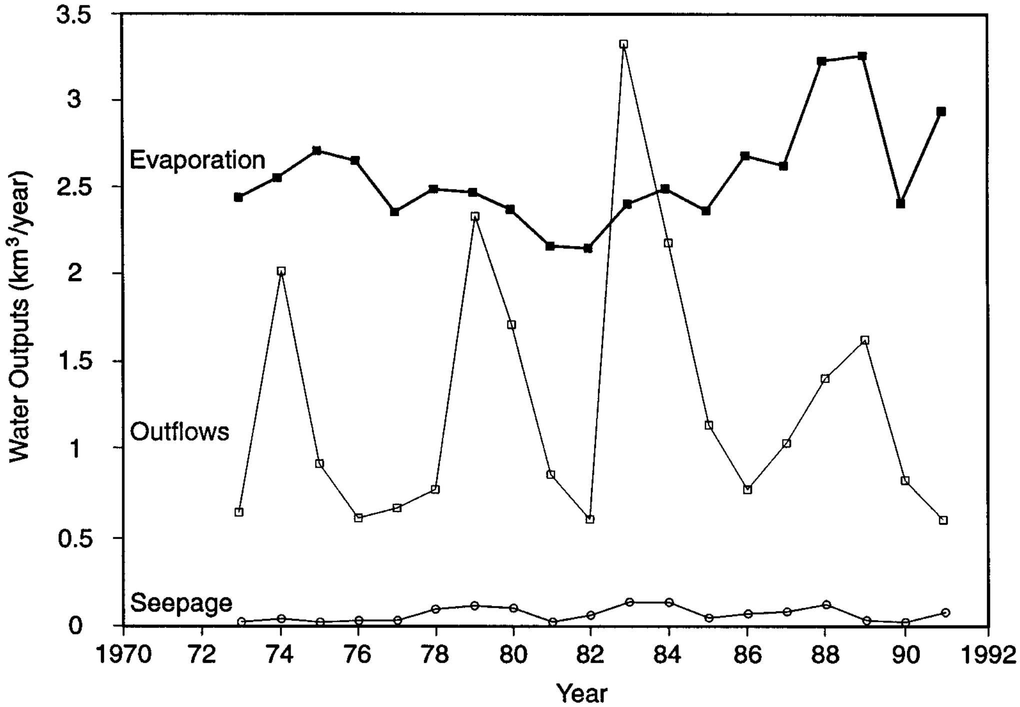



**Figure 3. Annual Mean Lake Water Outputs as Evaporation, Surface Outflows, and Seepage.**

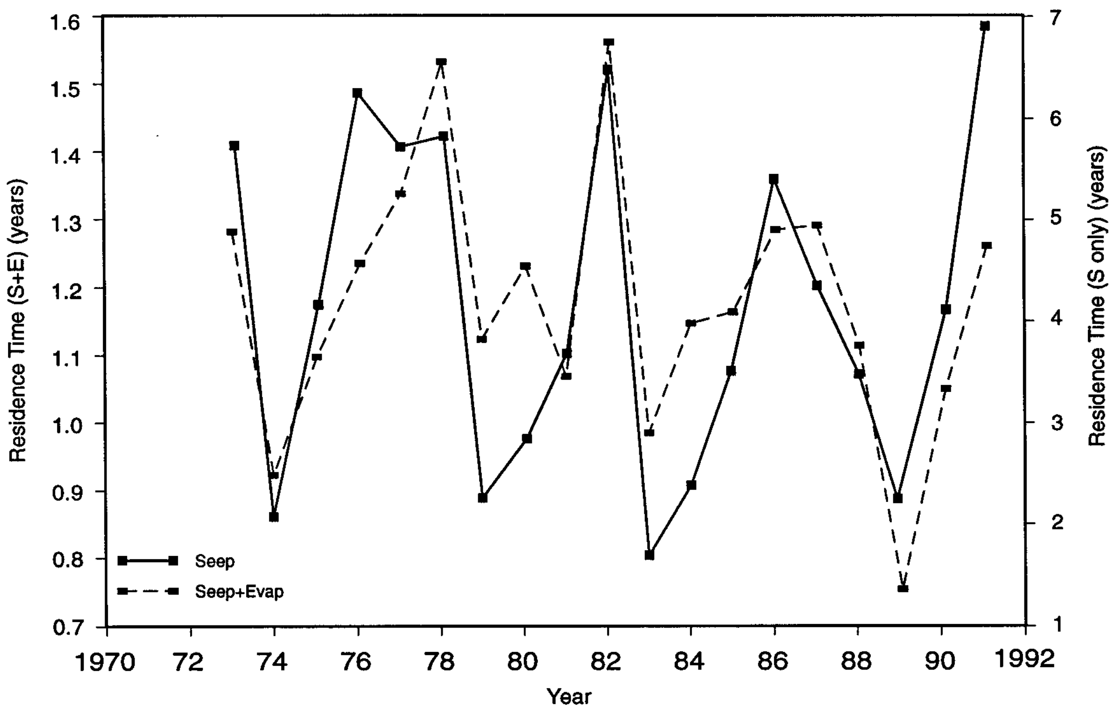



**Figure 4. Annual Mean Hydraulic Residence Times.**

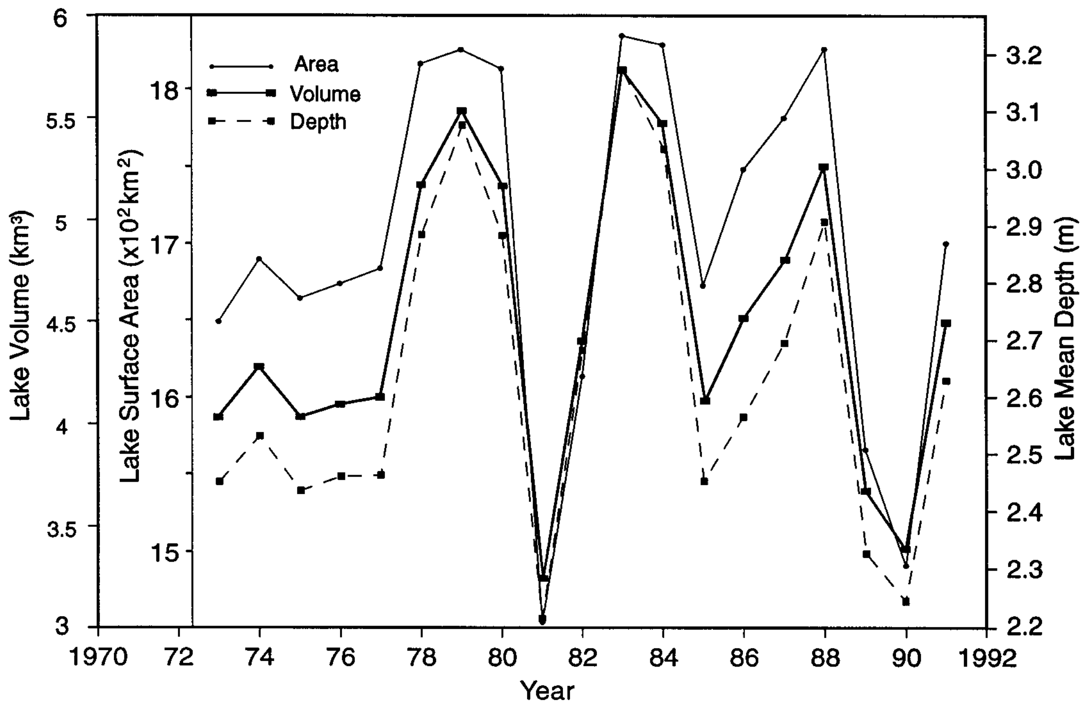



**Figure 5. Annual Mean Lake Volume, Lake Surface Area, and Lake Depth.**

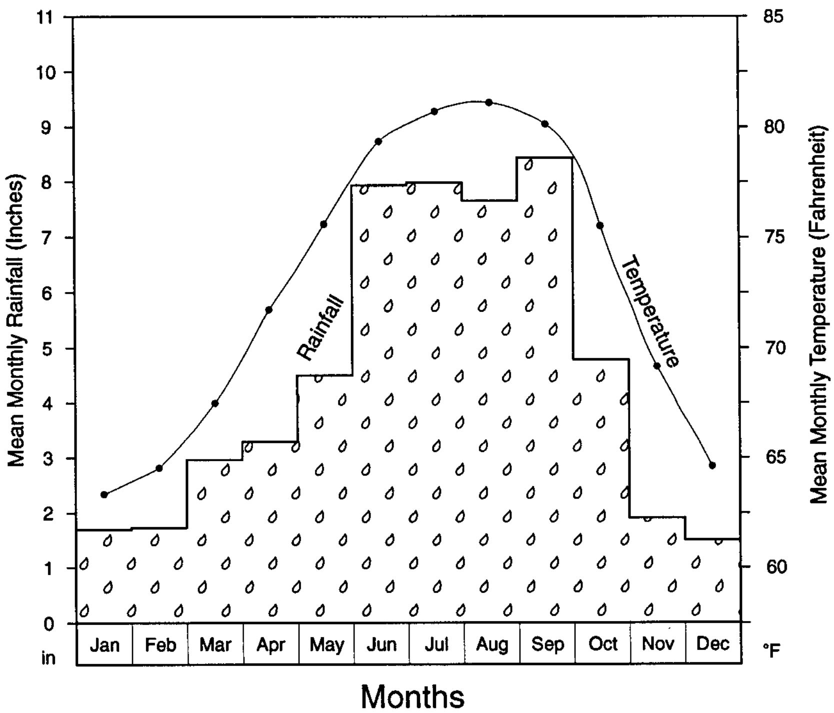



**Figure 6. Mean Monthly Rainfall and Monthly Temperature.**

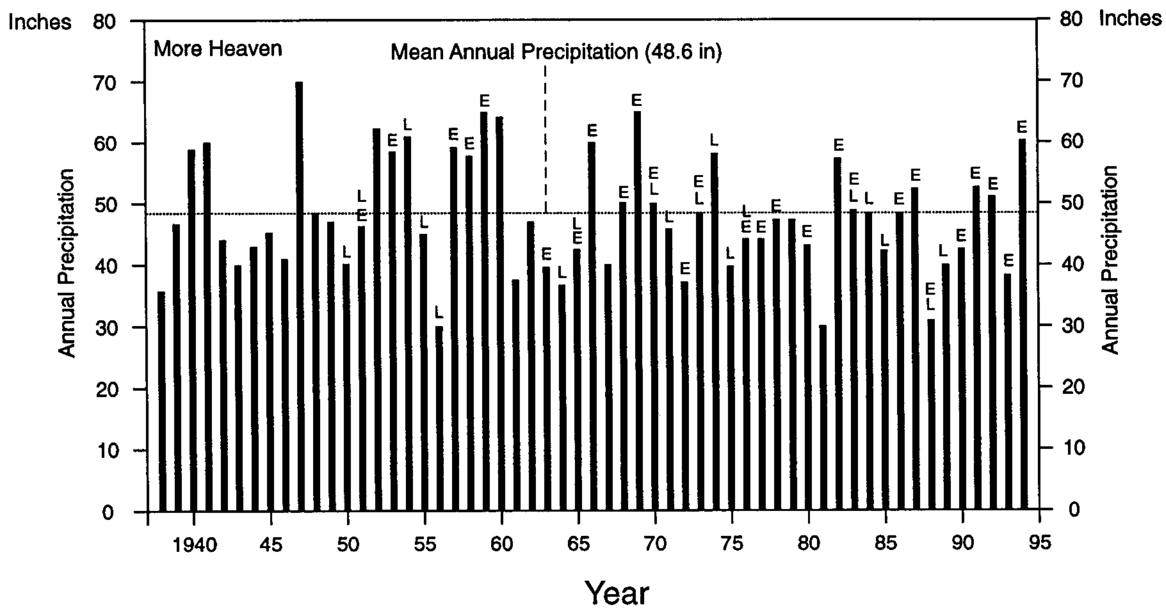



**Figure 7. Annual Mean Precipitation at More Heaven, South Florida. E Stands for El Niño, and L Stands for La Niña.**

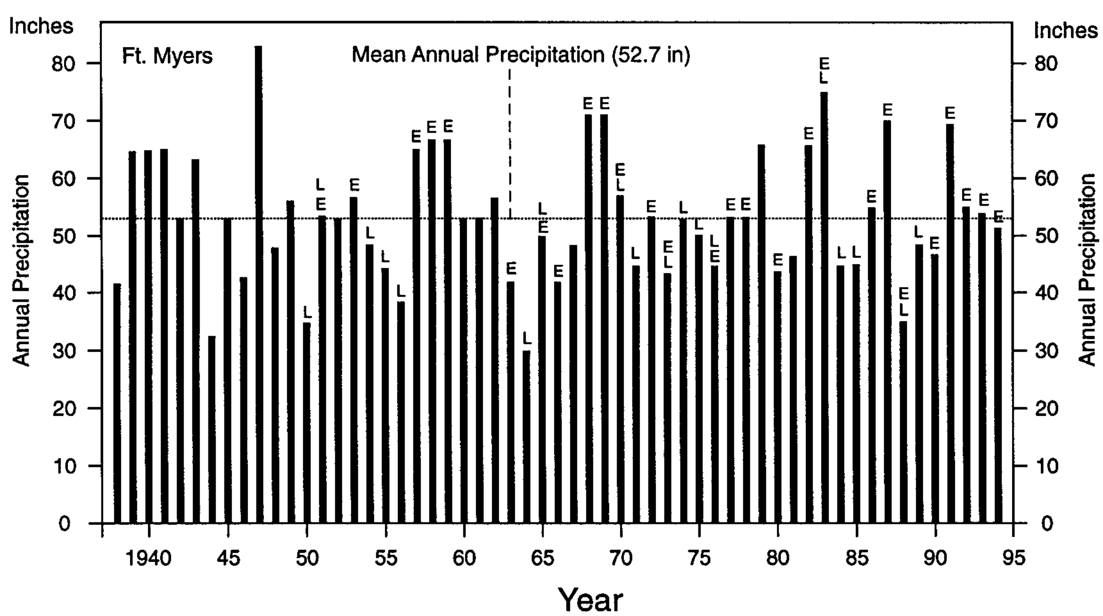



**Figure 8. Annual Mean Precipitation at Fort Myers, South Florida. E Stands for El Niño, and L Stands for La Niña.**

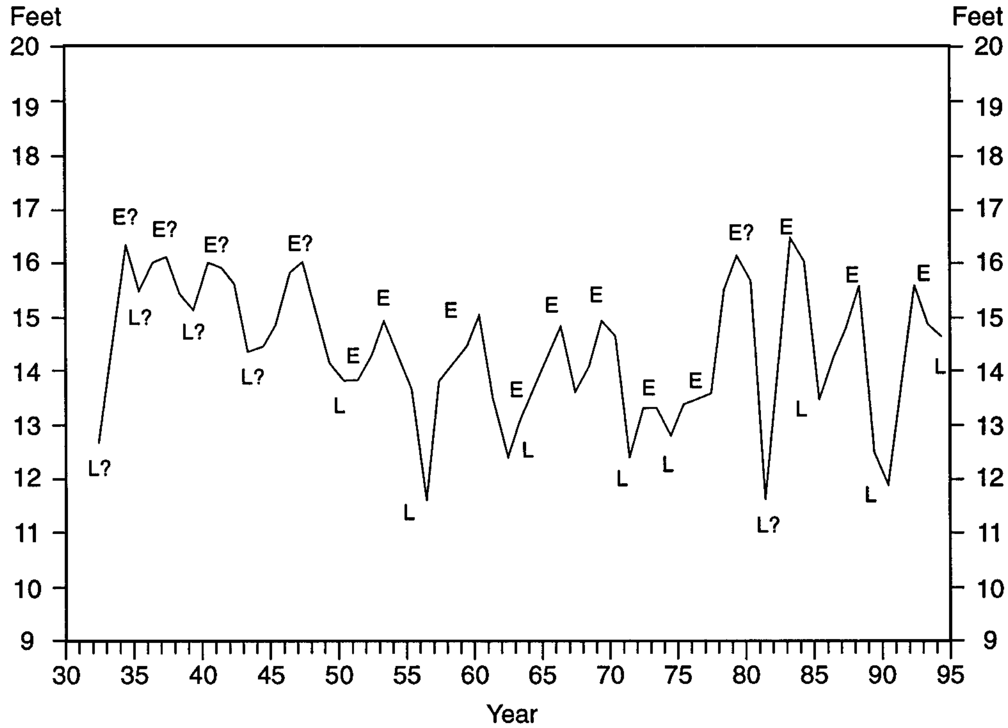



**Figure 9. Yearly Average Water Level Above Mean Sea Level in Lake Okeechobee (1932-1995). E Stands for El Niño, and L Stands for La Niña.**

**Table 1. The Season-by-Season List of Warm El Niño Conditions: Weak, Moderate, and Strong Periods.**

| | Jan-Mar | Apr-Jun | Jul-Sep | Oct-Dec |
|---|---|---|---|---|
| 1951 | | | | week |
| 1953 | | week | week | |
| 1957 | | week | week | moderate |
| 1958 | strong | moderate | week | week |
| 1959 | week | | | |
| 1963 | | | week | moderate |
| 1965 | | | moderate | strong |
| 1966 | moderate | week | week | |
| 1968 | | | | week |
| 1969 | moderate | week | week | week |
| 1970 | week | | | |
| 1972 | | week | moderate | strong |
| 1973 | moderate | | | |
| 1976 | | | | week |
| 1977 | | | | week |
| 1978 | week | | | |
| 1980 | week | | | |
| 1982 | | week | moderate | strong |
| 1983 | strong | moderate | | |
| 1986 | | | week | moderate |
| 1987 | moderate | moderate | strong | moderate |
| 1988 | week | | | |
| 1990 | | | week | week |
| 1991 | week | week | moderate | moderate |
| 1992 | strong | strong | week | week |
| 1993 | week | moderate | moderate | week |
| 1994 | | | moderate | moderate |
| 1995 | moderate | | | |
| 1997 | | moderate | strong | strong |
| 1998 | strong | moderate | | |

Data from Database of National Centers for Environmental Prediction/Climate Prediction Center, 1999.

**Table 2. The Season-by-Season List of Cold La Niña Conditions: Weak, Moderate, and Strong Periods.**

| | Jan-Mar | Apr-Jun | Jul-Sep | Oct-Dec |
|---|---|---|---|---|
| 1950 | moderate | moderate | moderate | moderate |
| 1951 | moderate | | | |
| 1954 | | | week | moderate |
| 1955 | moderate | week | week | strong |
| 1956 | moderate | moderate | moderate | week |
| 1964 | | | week | moderate |
| 1965 | week | | | |
| 1970 | | | | moderate |
| 1971 | moderate | week | week | week |
| 1973 | | | week | strong |
| 1974 | strong | moderate | week | week |
| 1975 | week | week | moderate | strong |
| 1976 | moderate | | | |
| 1983 | | | | week |
| 1984 | week | week | | week |
| 1985 | week | week | | |
| 1988 | | | week | strong |
| 1989 | strong | week | | |
| 1995 | | | | week |
| 1996 | week | | | |

Data from Database of National Centers for Environmental Prediction/Climate Prediction Center, 1999.

## Optrodes for Long-Term Remote Monitoring of Environmental Water Quality

Sean M. Christian, Paul Grems Duncan and John A. Schroeder
Airak, Inc, 9058 Euclid Avenue, Manassas, Virginia

E-mail: pduncan@airak.com

### ABSTRACT

Recent innovations pertaining to optrodes (optical sensors) and systems for monitoring environmental water quality will ultimately result in 1) improved mapping of geophysical fields, 2) a substantial reduction in the direct (labor) cost associated with conventional monitoring technologies, 3) robust data products for enhanced modeling capabilities, and 4) an enabling technology for protecting our Nation's resources. Airak, Inc. has been exploring the ability of specific optrodes to gather environmental water quality data in harsh environments using a fluorescent phase-based measurement system that is currently under development with support from the National Science Foundation. This exploration has yielded significant and positive results, including clear advantages over traditional intensity-based field deployable systems. Results from this system will be presented, advantages over conventional techniques will be discussed and the commercial implications of the technology will be addressed.

**Keywords:** Optrodes, chemical analysis, dissolved oxygen, dissolved carbon dioxide, acidity, pH, optical, sensor, antifouling, remote monitoring.

### INTRODUCTION

For many decades scientists have struggled to manage aquatic resources and ecosystems. This is due, in part, to the lack of an economically viable, rugged, field deployable, sensory system amenable to long-term remote monitoring of multiple water quality parameters and capable of real-time data telemetry.

In direct response to this lack of technology, the research team at Airak, Inc has been working on a rugged, economical, real-time monitoring technology for remote in-situ measurement of physical, chemical, and biological water quality parameters. The goal is an optical based system that will prove to be an enabling technology for protecting our Nation's resources. The combined sensory system will consist of a field-deployable support system in addition to the sensor suite that possesses: 1) a cost-effective platform with low power consumption, 2) energy scavenging technology, 3) surface electromagnetic links for control and data telemetry, and 4) dynamic network management software for timely event notification and adaptive system and/or software remote reconfiguration, to include uplink and data transmission rescheduling, alarm notification re-leveling, etc.

#### *Why Optrodes?*

Optical based systems are attractive for remote deployment due to their low hydrodynamic profile, fast response time, high sensitivity (low cross-sensitivity), and long- term stability. In order to highlight the innovative nature of this technology, the inherent advantages of optical based sensors are expanded upon below:

1. Amenability to miniaturization (optrodes the width of a human hair are possible),

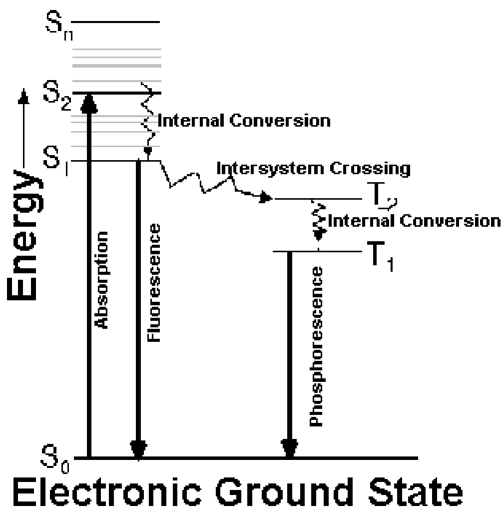



*Figure 1. The power of lifetime phase based sensing can be illustrated with a simplified Joblonski diagram. Represented is the absorption of a high-energy photon by an electron and promotion of that electron to a singlet ($S_n$) excited state. In most cases, this electron losses vibrational energy (Internal Conversion) and rapidly relaxes back to ground state (fluorescence). With Airak's $pO_2$ , $pCO_2$ ,and pH coatings, the excited electron is allowed to undergo intersystem crossing into a long-lived triplet ($T_n$) excited state. The average lifetime of this long-lived excited state is measurable and is dependent upon analyte concentration. More important, however, the lifetime of the emission (phosphorescence) is insensitive to the problems which effect intensity based measurements (i.e. photobleaching, innerfilter effects, background auto-fluorescence, etc.)*

2. Optical based sensors, unlike conventional electrode based sensors, are non-depleting. This means that they are able to sense the analyte of interest without consumption of the analyte and therefore, they can be used under "zero flow" conditions,
3. High sensitivities in conjunction with large dynamic ranges are derived from the high signal-to-noise ratios obtainable with optical-based sensors (optrodes possess enormous bandwidth), and
4. With respect to power consumption, optical based sensors generally posses low power budgets. This is an important consideration in designing instrumentation for field and/or remote deployment.

### *Optrode Design*
### *Phase- vs. Intensity-Based Optrodes*

The principle of fluorescent based sensing is as follows: a fluorescent molecule (molecular probe) is illuminated with an excitation source. Upon illumination, the molecular probe absorbs optical energy from the source and uses this energy to promote an electron into a high-energy (unstable) state. In time, the electron loses its excess energy and returns to its initial energy state (ground-state). In doing this, a photon is emitted. The timescale of the emission, the energy of the photons (i.e. wavelength) and the number of photons (intensity) all depend on the microenvironment around the molecular probe. Thus, by monitoring the intensity, the wavelength, or the lifetime of the probe's emission, one can sense a variety of analytes. This phenomenon is well documented (Demas and DeGraff, 1997).

One commonality that exists between all chemical probes is that they exhibit phenomena such as temperature sensitivities, cross-sensitivities to other analytes (i.e. pH) and support interactions. Support interactions are the interactions that develop upon embedding luminescent probes within a support matrix. The technical term used to describe these interactions is solvatochromism (a shift in the emission spectra due to a change in the permittivity of a luminescent probes local environment). Please note that, if the goal is to develop optical sensors for field applications, a support matrix is necessary to isolate the molecular probe at the tip of the optrode. Numerous supports have been examined by a number of researchers and all exhibit signs of interactions. These interactions can be either beneficial or detrimental to sensor performance. For example, rigid supports typically extend the excited state lifetimes of molecular probes and thus enhance the probe's ability to interact with the analyte of interest. This results in a sensor with increased resolution. At the same time, however, many rigid supports are only slightly permeable to the analyte of interest. This results in a sensor with decreased resolution. This means that the choice of support is paramount to the successful design of an optrode and is often a compromise.

Shahriari and MacCraith are two researchers who have had success with sol gel technology for luminescent probe supports in the analysis of $O_2$, CO and $H_2S$ ( see Krihak and Shahriari, 1995; Krihak et al., 1996; Murtagh et al., 1996; Shahriari et al., 1994; McEvoy et al., 1995). Although a commercial product exist based on this support (Ocean Optics, Inc.), these sensors have a number of limitations: 1) long-term stability is a concern due to leaching of the molecular probe from the matrix, 2) sensor fragility from internal stress buildup is a concern due to cure shrinkage of the support (ca. 50% by volume), and 3) residual catalyst (acid) concentrations have been shown to lead to accelerated degradation of the molecular probe.

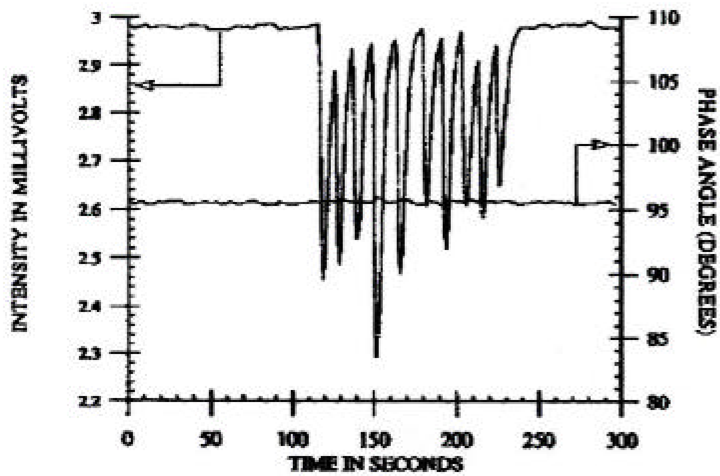



*Figure 2. This illustrates the superiority of the lifetime phase-based system over the intensity-based measurement system. The data is from Dr. Govind Rao of the University of Maryland. The experiment has been referred to as the hand waving experiment, for a graduate student simply waved his hand back-and-forth in front of the optrode to generate the signal response shown. Notice that the intensity-based measurement is extremely sensitive to the disturbance, yet the lifetime phase-based measurement exhibits no response to the parasitic optical signal (the field analog of this is an insensitivity to diurnal effects and/or background auto-fluorescence of the water column).*

To circumvent these problems, a number of researchers have employed the use of polymers as supports with great success. Walt (Walt, 1997; Ferguson et al., 1997; Healy and Walt, 1995; Walt et al., 1995), Weigl, et al. (1992), Morisawa, et al. (1996) and Campagna (1993) have published results that detail different chemical methodologies for sensor fabrication. Using polymers, Airak, Inc. has successfully developed methods to maximize permeability, combat fragility, prevent leaching of the molecular probe from the support, and remove sensor drift due to temporal changes (i.e. physical aging) of the support for a dissolved oxygen ($pO_2$) optrode and is currently addressing these issues for both dissolved carbon dioxide ($pCO_2$) and acidity (pH) optrodes.

Airak realized early in its research efforts that intensity-based systems possess serious limitations with respect to long-term field deployment.

Specifically:

1. *Parasitic optical signals due to backscatter, ambient light fluctuations (i.e. diurnal effects) and background autofluorescence,*
2. *Photobleaching of the molecular probe, and*
3. *Self absorption and innerfilter effects of the coating.*

The result is long-term sensor drift and, consequently, the need for frequent recalibration. In an attempt to mitigate the effects of parasitic optical signals, some organizations have developed dual wavelength (i.e. ratio-metric) techniques (e.g. YSI, Inc.) and opaque optical coatings (e.g. Ocean Optics). However, to Airak's knowledge, no techniques exist to guard against source decay, photobleaching and innerfilter effects, thus, intensity-based systems will always require frequent recalibration.

***This is a serious problem if the desire is to leave the unit at a remote location for extended periods.*** In order to circumvent these limitations, Airak has developed a non-intensity, phase-based, field deployable beta system, which represents the first, commercially viable, long-term, field deployable sensor system for remote, long-term monitoring of water quality parameters. ***Lifetime phase-based measurement techniques are insensitive to the problems that restrict intensity-based techniques to laboratory and/or controlled environments***. While some researchers have developed phase techniques as biomedical diagnostic tools, to Airak's knowledge, Airak is the first to explore the use of phase-based measurement for environmental monitoring.

Figure 1 is a graphic that illustrates the principle and the power of lifetime phase based measurement***.***
***Lifetime phase based measurement techniques are insensitive to all of the problems that restrict intensity-based techniques to laboratory and/or controlled environments***. Note that due to the photophysics of phosphorescence, lifetime phase-based measurements are insensitive to photobleaching, innerfilter effects, and background autofluorescence ***(i.e. these systems are ideal for field deployment).*** A second example that clearly illustrates the superiority of lifetime phase-based measurement is shown in Figure 2. This graphic shows that phase measurements are insensitive to parasitic optical signal.

**RESULTS**

Results for Airak's sensors' sensitivity analysis are shown in Figures 3-7.

***Airak's prototype phase-based beta system (left figure) and run-time data (right figure)***

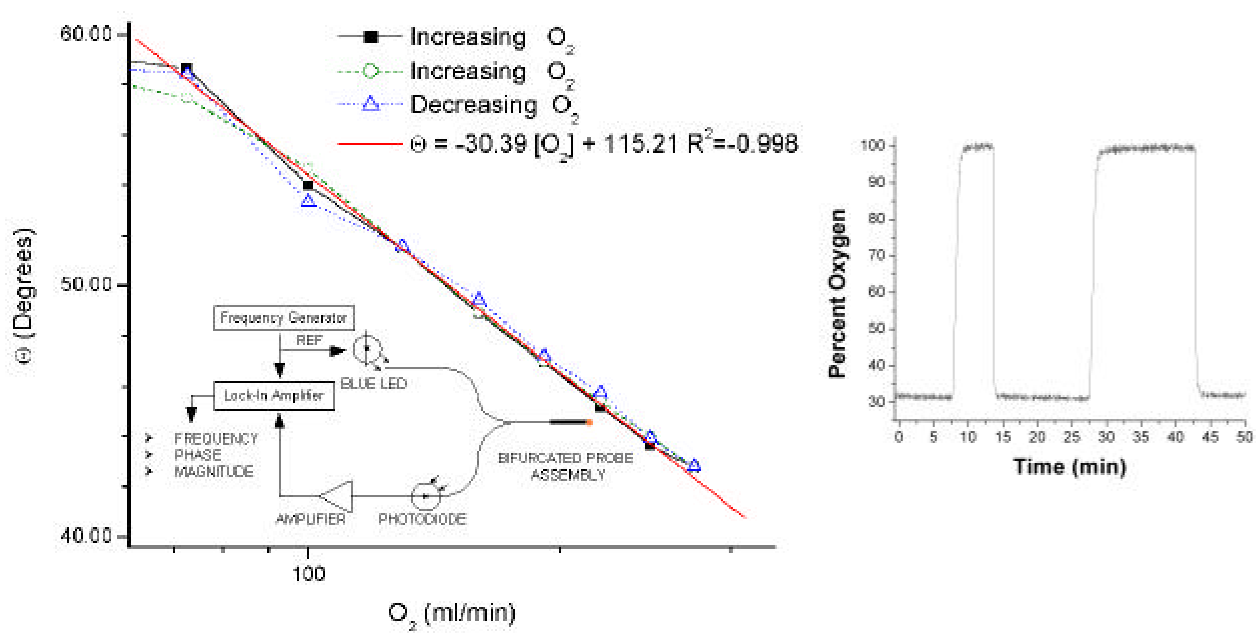



*Figure 3. The graphic on the left is representative data from Airak's prototype phase-based beta system. The resolution of the system was 0.007 ppm at low $pO_2$ concentrations and 0.2 ppm near saturation. This evolving resolution is due to the non-linear response of optrodes (i.e.* ***the lower the $pO_2$ concentration, the higher the optrode's resolution****). Embedded within the graphic is a system schematic. As shown in the schematic, the phase base system can be constructed from low cost components. The system constructed by Airak consists of a low cost light emitting diode (LED) or laser diode (LD), a low cost filtered photodiode (FPD), the optrode and the signal conditioning electronics. Note that this system is applicable to the detection of all three analytes ($pO_2$, $pCO_2$ and pH). Therefore, this system is capable of resolving all three analytes via a single analyzer. Employing an intensity-based technique to monitor all three analytes would require the use of either complex chemometrics or a separate analyzer for each analyte of interest. The graphic on the right is run-time data showing 100% $pO_2$ at the upper limit and 30% $pO_2$ at the lower limit. Notice the rapid response time.*

## $pCO_2$ via Lifetime Phase-based Measurement

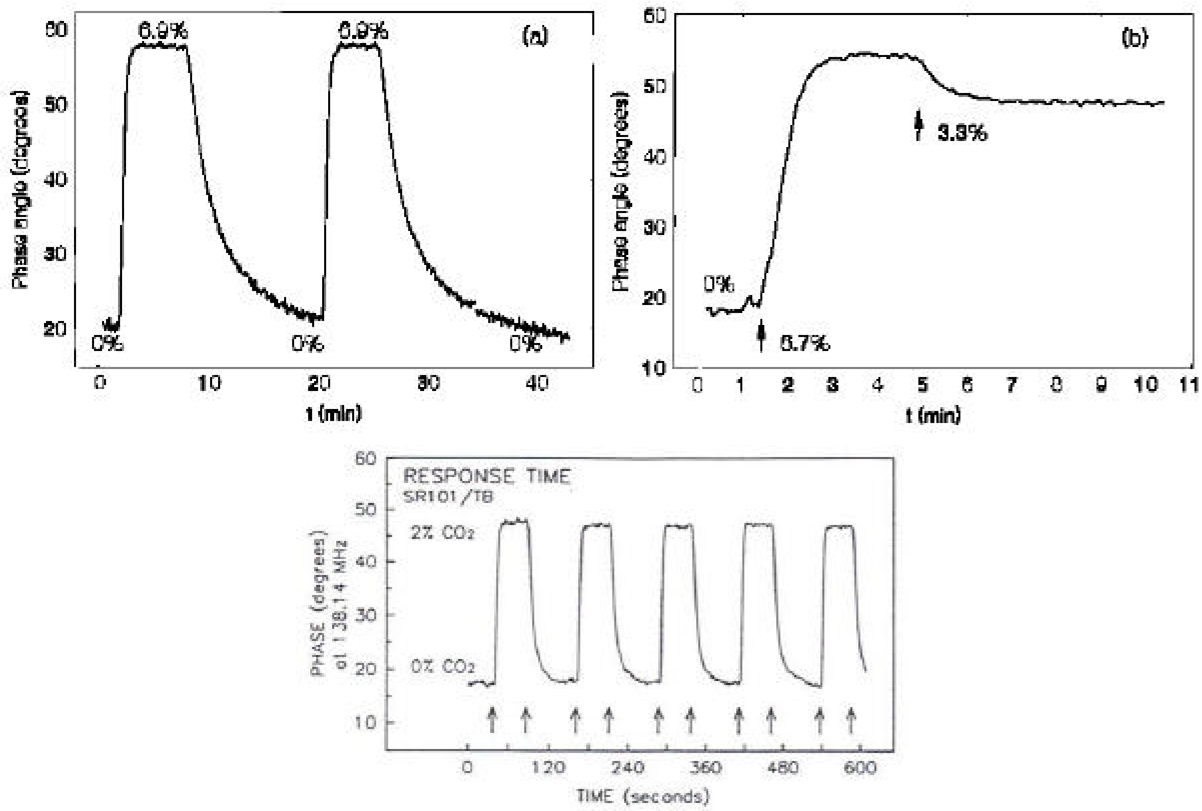



*Figure 5. The above graphics represent $pCO_2$ data taken by a steam-sterilizable phase-based $pCO_2$ optrode. The gases were introduced via (a) bubbling and (b) liquid-to-liquid mixing. The data illustrates that the optrode possesses a **resolution of ~0.1 ppm** at low $pCO_2$ concentrations, such as those found in natural waters. The lower graphic is illustrative of the optrode's rapid response time.*

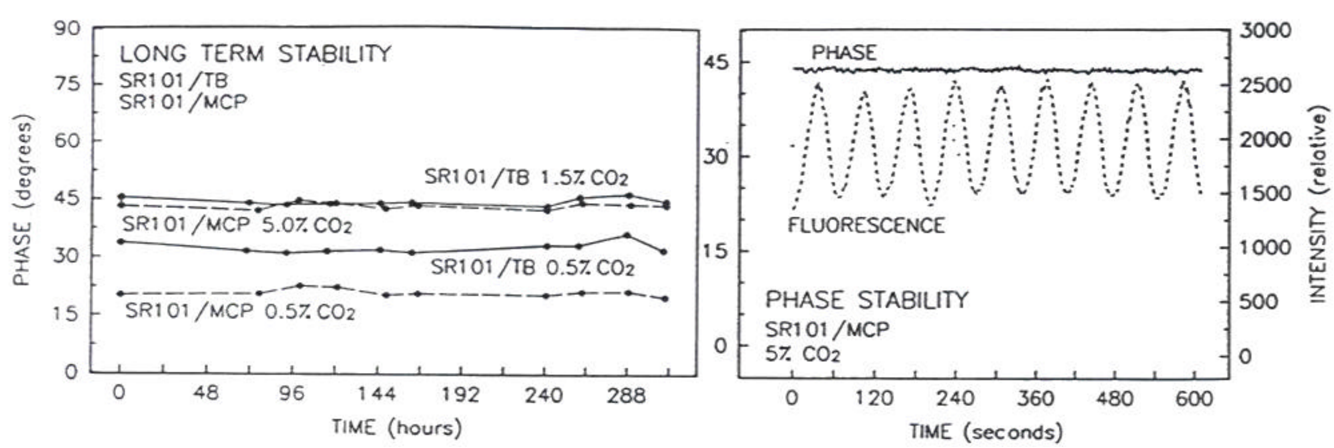



*Figure 4. The left graphic illustrates the stability of the $pCO_2$ optrode with respect to time. The right graphic shows that the phase remains constant with respect to fluctuating excitation intensity. In both cases, the $pCO_2$ concentration was held constant.*

## *pH via Lifetime Phase-based Measurement*

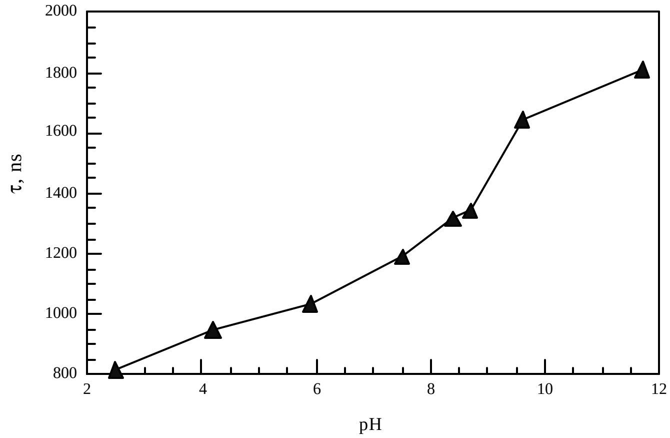



*Figure 6. The above graphic is representative data from Airak's pH optrode taken by Drs. Jim Demas of the University of Virginia and Benjamin Degraff of James Madison University. The resolution of the optrode was ~0.09 pH units and the dynamic range exceeded 10 pH units.*

### *Antifouling Technology*

Early field trials led Airak to aggressively pursue the development of an antifouling technology. Airak was able to develop, through collaboration with Virginia Tech's Department of Plant Pathology, Physiology, & Weed Science (PPPWS), an antifouling technology capable of extending the field deployable lifetime of its probe technologies. No matter how advanced the sensor technology, it must be protected from biofouling (i.e. drift due to biological growth on the sensor). Airak has acquired a vast knowledge base pertaining to the problem of biofouling of both optrodes and conventional sensor technologies in aquatic environments. More important however, Airak, has discovered a non-mechanical, non-toxic (i.e. non-metallic) methodology for protecting optrodes from biofouling.

To Airak's knowledge, no companies' market antifouling methodologies other than those that require user intervention, mechanical action or pose a serious treat to the surrounding environment (i.e. metallic based chemical methodologies).

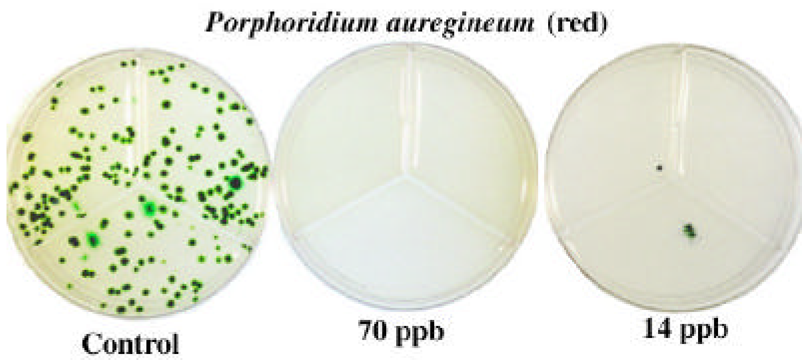



*Figure 7. The photograph above depicts the effects of Airak's organic (non-metallic) antifouling technology on red (Porphoridium auregineum) alga. At 70 ppb the biocide was also effective against green (Klebsormidium flaccidum) and on blue-green (Phormidium faveolarum). Note that the concentration level of biocide is ppb (parts-per-billion).*

## APPLICATIONS

The fully developed and commercialized sensing system will provide tremendous benefits in areas where constant, unattended monitoring of aquaculture and/or water quality is deemed necessary. Specific applications include the following:

1. Aquaculture Production Facilities. Abnormal water quality parameters could potentially devastate this >$800M commercial industry.

2. Endangered Species Habitats. The United States waterways are home to many endangered species that share their environment with farmers, licensed treatment facilities, and other industrial concerns. Establishing monitoring points within the habitat, as well as above and below industrial discharge points, could provide a much-needed early warning system for potentially devastating conditions.

3. Ground Water Monitoring. As previously stated, groundwater-quality concerns within a given watershed can be enormous.

4. Mining Concerns. Several areas within the United States are populated with mines, some of which are abandoned. Many of these mines are in key watershed areas and adversely affect these resources. Due to the rugged terrain, the proposed sensor technology is ideal for this type of application. The U.S. Department of the Interior, U.S. Fish & Wildlife Service has been contacted concerning the system and is extremely interested in deployment.

5. Remote Monitoring. Field biologists must now take field trips to remote areas to sample water quality. Due to staffing constraints, budgetary limitations, and other diminished resources, monitoring of water quality generally occurs only after a catastrophic event has occurred. Airak's system will allow the field biologist to install the system at a remote location, verify proper operation, and leave the area. Upon return to the office, the biologist will be able to connect to a centralized data collection system that receives data from this and other systems, and presents the data in an easy-to-interpret format. The only field trips that will be required will be those to service the individual systems or those that are in response to an alarm event.

6. Wastewater/Effluent Monitoring. This market generated $283M in revenues in 1997 and is forecasted to grow to $341 million by the year 2001. Additionally, wastewater analytical instrumentation generated $175M in revenues in 1997 and is forecasted to grow to $205M by 2001. Most treatment facilities are staffed 24/7, and require periodic sampling and chemical analysis to determine the concentration of analytes under consideration. Again, Airak's system will alleviate much of the need for periodic sampling by providing facilities with online, real-time data products capable of preventing contamination of municipal water supplies through early warning mechanisms.

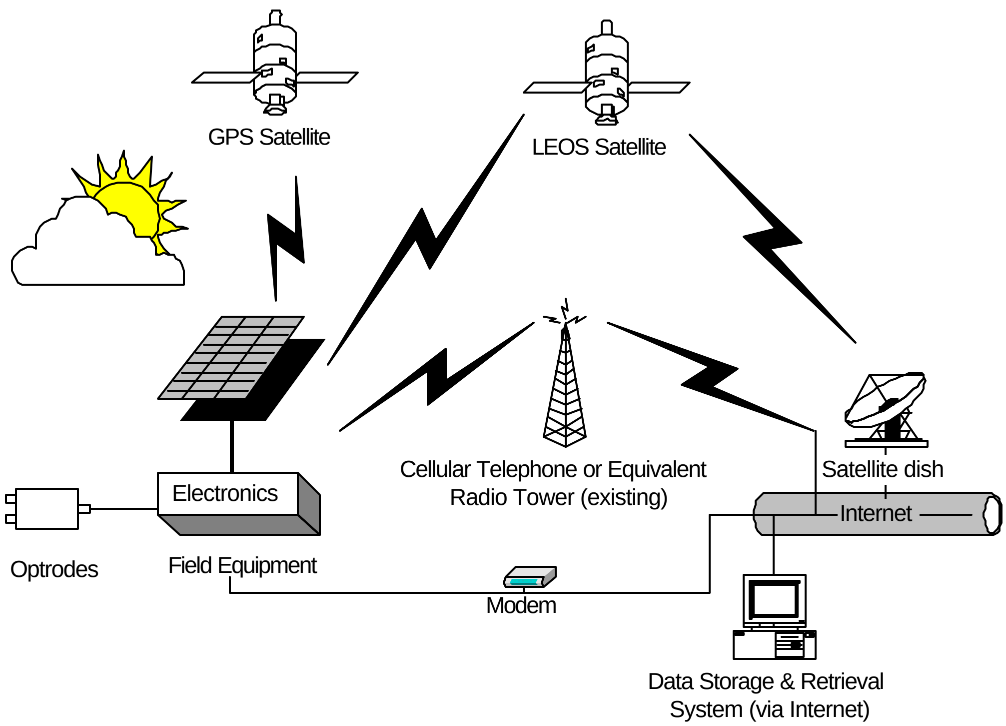



*Figure 8. Above is a concept diagram of Airak's proposed field deployable system that has been adequately titled AFFIRMER (Advanced Field-Functional Instrumentation for Remote Monitoring and Environmental Reconnaissance). As shown, multiple communication options are available. For field units inaccessible to a landline modem connection, a state-of-the-art RF modem option developed by Motorola will be available.*

## CONCLUSIONS

Airak, Inc. develops, manufactures, markets, and services optical sensors and their support systems for users that require the unique advantages of optical sensors. The use of patent-pending technologies, combined with wireless communication and network configurations, will enable Airak to address the needs of diverse user groups.

The primary mission of the research Airak was to develop advanced optical sensor technologies amenable to long-term, remote, unattended monitoring. Conventional sensor technologies for environmental water quality data suffer from the need for frequent recalibration, which limits their field deployable lifetimes. The major impediment with respect to conventional sensor technologies and their inability to be placed in the field for any length of time is a phenomenon known as biofouling. The team at Airak has developed fiber optic sensors and sensor coatings that retard or inhibit sensors from fouling, hence allowing them to remain in operation for extended periods with little to no maintenance.

## REFERENCES


Campagna, S., A. Bartolotta, and G. Di Marco. 1993. Photophysics of Ru (II)- and Os (II) – polypyridine complexes in poly(ethyleneoxide) matrices. Chemical Physics Letters, 206(1,2,3,4): 30-36.

Demas, J.N. and B.A. DeGraff. 1997. Applications of luminescent transition metal complexes to sensor technology and molecular probes. Journal of Chemical Education, 74: 690-695.

Ferguson, J.A., B.G. Healey, K.S. Bronk, S.M. Barnard, and D.R. Walt. 1997. Simultaneous of pH, carbon dioxide and oxygen using an optical imaging fiber. Analytica Chimica Acta, 340: 123-131.

Healey, B.G. and D.R. Walt. 1995. Improved fiber-optic chemical sensors for penicillin. Analytical Chemistry, 67(24): 4471-4476.

Krihak, M. and M.R. Shahriari. 1995. Fiber optic chemical/environmental sensors on active coatings. SPIE, 2508: 358-367.

Krihak, M., M. Murtagh, and M.R. Shahriari. 1996. Fiber optic oxygen sensors based on the sol-gel coating technique. SPIE, 2836: 105-115.

McEvoy, A.K., C.M. McDonagh, and B.D. MacCraith. 1995. Development of a fiber-optic dissolved oxygen sensor based on quenching of a ruthenium complex entrapped in a porous sol-gel film. SPIE, 2508: 190-198.

Morisawa, M. K. Uchiyama, T. Hosaka, H. Inoue, S. Muto. 1996. New cladding polymer for optical oxygen sensor using fluorescent plastic fiber. SPIE, 2836: 336-340.

Murtagh, M., M. Krihak, H.C. Kwon, and M.R. Shahriari. 1996. Luminescence decay analysis of doped sol-gel films. SPIE, 2836: 87-97.

Shahriari, M.R., J.Y. Ding, J. Tong, and G.H. Sigel, Jr. 1994. Sol-gel coating based fiber optic DO sensors. SPIE, 2068: 224-238.

Walt, D.R. 1997. Fiber Optic Sensors. Accounts of Chemical Research 31(5): 267-278.

Walt, D.R., T.A. Dickinson, B.G. Healey, J.S. Kauer, and J. White. 1995. Fiber optic array sensors as an architecture for an artificial nose. SPIE, 2508: 111-116.

Weigl, B.H., A. Holobar, W. Trettnak, I. Klimant, H. Kraus, P. O'Leary, and O.S. Wolbeis. 1992. An optical triple sensor for measuring pH, oxygen and carbon dioxide in bioreactors. SPIE, 1796: 287-297.

# Can the New River be Represented Among Global Waters? Radford Meander Loop of the New River, Radford, Virginia.

Brendan T. Cox and Ernst H. Kastning
Environmental & Engineering Geoscience, Radford University
Radford, Virginia

Stewart J. Sweeney
Nova Environmental Geoscience Services
Nova Scotia, Canada

E-mail: bcox@runet.edu

## ABSTRACT

The purpose of this study was to analyze water samples from the Radford Meander Loop of the New River in Radford, Virginia. From the analytical results, the New River chemical species can be compared to global means for river waters. Also, based on these analyses, the techniques used to determine the controls on the chemical composition of rivers can themselves be assessed.

**Keywords:** New River, water chemistry, discharge quality, global surface-waters

## INTRODUCTION

The New River is a complex system and many of its parameters have not been investigated. At Radford, Virginia, the New River experiences numerous environmental and engineering issues. The New River flows completely across the Valley and Ridge Physiographic Province, something that no other stream system does. This anomalous situation and its geologic explanations have been the source of controversy for years.


**Figure 1.** Radford Meander Loop of the New River, Radford, Virginia

The headwaters of the New River are located near Blowing Rock, North Carolina, and the river flows downstream to the northeast. This direction of flow is parallel to the Valley and Ridge Province. In the vicinity of Radford (Figure 1), the New experiences a change in direction, a downstream-course to the northwest. The monitoring site is located downstream of Memorial Bridge (from USGS 1:62,500 of Radford, Virginia).

Based on geologic evidence downstream from McCoy Falls (approximately 10 to 15 miles from the City of Radford), to Eggleston indicates that the stream system existed before the Alleghanian Orogeny of the Appalachian Mountains. The New River has bisected four large ranges (Figure 2), first Cloyds Mountain (southwest) and Brush Mountain (northeast), second, Walker Mountain (southwest) and Gap Mountain (northeast), and third, Buckeye Mountain (southwest) and Spruce Run Mountain (northeast). Finally at the town of Narrows, the New River separates the last ridge, East River Mountain (southwest) and Peters Mountain (northeast).

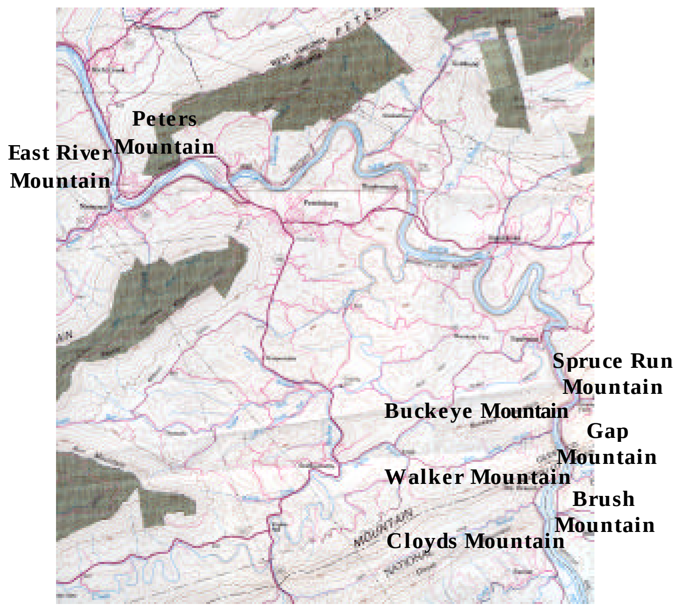



**Figure 2.** New River bisecting four ridges of the Valley and Ridge Physiographic Province (from USGS 1:100,000 of Radford, Virginia-West Virginia).

Techniques described by Gibbs (1970) will allow for an assessment of the New River with respect to global waters. This type of analysis could lead to clues as to why the New River changes it's downstream-course from the northeast to the northwest. These techniques have never been utilized for the New River, including the Radford Meander Loop.

## OBJECTIVE

The purpose of this study is to analyze water samples from the Radford Meander Loop of the New River in Radford, Virginia. From the analytical results, the New River chemical species can be compared to global means for river waters as described by Livingstone (1963). Also, based on these analyses, the techniques used to determine the controls on the chemical composition of rivers can themselves be assessed.

## METHODOLOGY

Gibbs (1970) indicates that surface-water chemistry is determined by either 1) characteristics of rainfall (precipitation-dominated), 2) rock-weathering reactions (rock- dominated), 3) evaporation-precipitation of ions, or by combinations of these parameters. Figures 3 and 4 summarize Gibbs' ideas in plots of total dissolved solids (TDS) versus the $[Na^+/(Na^+ + Ca^{2+})]$ ratio and TDS versus relative mole fractions of $Ca(HCO_3)_2$ versus NaCl in the water. Because the composition of world rainfall (unpolluted) is largely determined by the NaCl content of sea salt, the chemical composition of rainfall plots in the lower right-hand corner of each figure at low TDS, as do streams with chemical composition dominated by rainfall such as the Negro River, a tributary of the Amazon River. Evapotranspiration from drainage basins in arid climates and streams such as the Colorado, Pecos, and Jordan rivers, which receive soil runoff and irrigation return waters, further increase the $Na^+$ and TDS content of streams. Precipitation of $CaCO_3$ further shifts the prevalent chemical character of such streams back toward NaCl and the chemistry of seawater (Langmuir, 1997).

Using data published by Gibbs (1970), the New River, at Radford, Virginia, can be compared with global waters. Weathering of continental rocks increases (TDS) in stream-water as well as concentrations of calcium and bicarbonate relative to sodium and chloride. The composition of streams so affected plot in the left part of both diagrams (Figures 3 & 4) and include the Columbia, Mississippi, Yukon, and Thames rivers, among others.

Three water samples were obtained from the New River (Figure 1) on 15 November 1999. These samples were collected approximately 30 to 50 meters downstream of Memorial Bridge (U.S. Route 11), Radford. One sample was obtained from the middle of the river, and the other two approximately 25 to 30 meters from each respective bank. The samples were analyzed for pH, conductivity, TDS, chloride, calcium, magnesium, sodium, sulfate, iron, and bicarbonate. TDS and sodium/calcium data were used in plotting the New River with respect to global waters. The other parameters that were analyzed were compared to values of global mean-river-waters.

## RESULTS AND DISCUSSION

The data obtained from the New River samples were averaged, so a comparison with global mean-river-waters could be accomplished. Table 1 displays this comparison.

Most of the concentrations of species that were considered are relatively similar between the New River data and the global mean-river-waters (Table 1). $Ca^{2+}$ is significantly lower for the New River, compared with the global mean value. $Mg^{2+}$ is similar between the two waters. $Na^+$ is lower then that for the global mean value possibly due to the geological materials that the New River encounters within the Radford Meander Loop. The bicarbonate numbers are approximately the same, with the New River being slightly lower. This is interesting because carbonate-units are the principle rocks that the New River encounters as it flows through the Valley and Ridge Physiographic Province. Concentration of the sulfate ion in the New River is about half that as of the global mean value. The concentration of elemental iron is significantly less in the New River than the global mean value. Total dissolved solids are also significantly lower in the New River as compared to the global mean value. This could be an effect of either Claytor Dam (owned by Appalachian Electric Power), Little River Dam (owned by the City of Radford), or both. Both dams are approximately 5 to 6 miles upstream of the sampling site within the Radford Meander Loop.

**Table 1.** Comparison between the chemical species of the New River and Global Mean-River Waters (mean-river-water data from Livingstone 1963).

| Species | New River | Global Mean-River-Water |
|---|---|---|
| $Ca^{2+}$ (mg/L) | 10.19 | 15.0 |
| $Mg^{2+}$ (mg/L) | 4.9 | 4.1 |
| $Na^{+}$ (mg/L) | 4.34 | 6.3 |
| $HCO^{3-}$ (mg/L) | 55.0 | 60.0 |
| $Cl^{-}$ (mg/L) | 8.67 | 7.8 |
| $(SO_4)^{2-}$ (mg/L) | 5.39 | 11.0 |
| Fe (mg/L) | 0.034 | 0.67 |
| TDS | 75.33 | 120.0 |

Using Gibbs (1970) techniques, the analyzed samples were averaged and plotted with respect to the world's surface waters. Figure 3 displays the general idea of Gibbs' work. Figure 4 incorporates global surface-waters with respect to the New River at Radford, Virginia.

The position of the New River analysis is towards the rock-dominance part of Gibbs' diagram. This is likely the result of the New River encountering the Elbrook Formation throughout the Radford Meander Loop. The Elbrook Formation is exposed throughout the City of Radford and is well known throughout southwestern Virginia. The Elbrook varies in thickness from 1500 to 2900 feet (variation is the result of karstic terrane). The Elbrook generally consists of thin- to thick-bedded medium gray, fine-grained dolostone that is ribbon-banded and locally shaly. The dominance of dolostone (a carbonate rock) throughout the area results in the New River plot within the rock-dominance part of Figure 3. The TDS data is time-dependent and can fluctuate because, as noted earlier, there are two large dams present not far upstream from the meander loop. The data display convincing results about the general controls on the chemical composition of rivers. Can this data explain why the New River changes its downstream-course from a northeast to a northwest direction within the Radford Meander Loop? This is speculation at this time and may continue to be in the future.

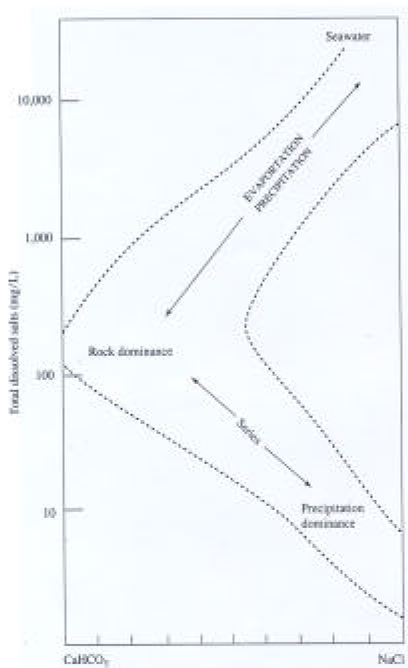



**Figure 3.** Description of processes controlling the chemistry of world surface-waters. From R.J. Gibbs (1970) and Langmuir (1997).

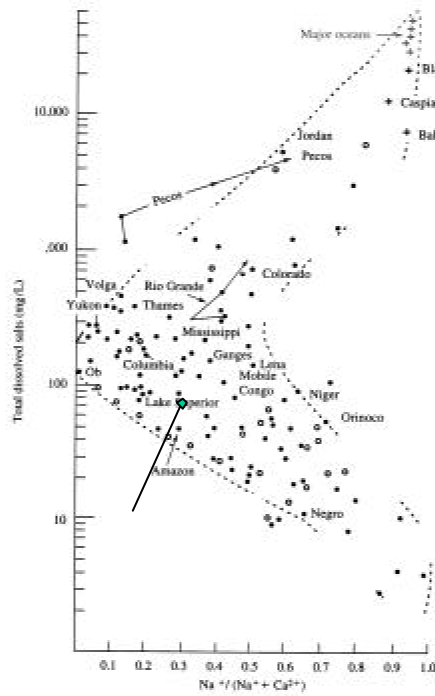



**Figure 4.** New River at the Radford Meander Loop with respect to world surface-waters. From R.J. Gibbs (1970) and Langmuir (1997).

## CONCLUSIONS

This initial water-quality investigation has placed the New River into a global perspective and allowed a comparison with world mean-river-water data. The New River at Radford, Virginia has made the city what is today. There are numerous environmental and engineering issues that are present throughout the meander loop. The environmental issues involve two major areas: agricultural and industrial. The engineering challenges that the New River encounters are numerous. Structures that are present within the Radford Meander Loop include Claytor Dam, Little River Dam, Little River Road Bridge, abandoned Radford Limestone Quarry, abandoned Radford Limestone Quarry Bridge and secondary bridge abutments, Interstate-81 Bridge, Operable train bridge and historic bridge abutments, Memorial Bridge (soon to be replaced, downstream of the existing bridge), and the non-operable train bridge (downstream boundary of Radford University's Dedmon Center Property). Future work in this study includes a review of the monitoring data performed by Virginia's Department of Environmental Quality, investigation of engineering structures within the meander loop for stability-issues, and an environmental-impact-assessment (EIA) to be performed for the Radford Meander Loop because of these various concerns.

## ACKNOWLEDGMENTS

I would first like to thank Radford University for the support that I have received throughout my undergraduate and graduate years. Charlotte Burnett of Virginia's Department of Conservation & Recreation and George Devlin of Virginia's Department of Environmental Quality provided insight to the Radford Meander Loop of the New River. I am grateful to American Heritage of Rivers (AHR) and New River Community of Partners (NRCP), for these organizations nationally recognize the New River watershed. Finally, I would like to acknowledge Ben Borda (New River Navigator, United States Army Corps of Engineers) for his dedication to the New River communities.

## REFERENCES

Gibbs, R.J., 1970. Mechanisms Controlling the Chemistry of World Surface-waters. *Science*, 170:1088-90.
Langmuir, D., 1997. Aqueous Environmental Geochemistry: Prentice Hall, NJ; 340p.

Livingstone, D.A., 1963. Chemical Composition of Rivers and Lakes. U.S. Geological Survey Professional Paper, Report: P 0440-G, pp. G1-G64.

**Lessons Learned from a New Environmental Monitoring Approach in the Mid-Atlantic Region**

Tom DeMoss, EPA Region III; Tom Pheiffer and Patricia Bradley
EPA Office of Research and Development;
Kent Thornton, FTN Associates, Ltd.

E-mail: kwt@ftn-assoc.com

**EXTENDED ABSTRACT**

In 1995, the EPA Office of Research and Development (ORD) formed a partnership with EPA Region 3 to implement a research, monitoring, and assessment initiative in the mid-Atlantic region—the Mid-Atlantic Integrated Assessment (MAIA). The MAIA Mission is to inject scientific knowledge into the decision-making process for the Mid-Atlantic Region of the United States.

MAIA is an interagency, multi-disciplinary program that is integrating and assessing research and monitoring information to provide answers to policy and management questions. The MAIA project is moving from information overload to providing managers and decision-makers with the relevant and necessary scientific information on which to base sound environmental decisions. Scientific projects conducted in MAIA are focused on policy and management issues of critical importance to resource managers and environmental decision makers.

Since its establishment in 1995, MAIA has forged alliances with other federal and state agencies, producing an array of useful products on the ecological condition of estuaries, streams, groundwater, and landscapes. In this next century MAIA will build on this foundation and create an arena in which innovative approaches for environmental assessment and management can be proven, implemented, refined, and subsequently, communicated and transferred not only to MAIA partners and alliances, but also to other EPA Regions, federal, state, and local agencies, academics, Non-Governmental Organizations (NGOs), the public, and other stakeholders. MAIA recently took a retrospective look at its activities and findings over the past 10 years and formulated a set of "Lessons Learned" from this initiative that is presented in this paper.

## Perspective

Based on our historical approaches for monitoring human health and physical and chemical pollutants in the environment, we've made a lot of progress. The State of the Environment in the Mid-Atlantic Region has improved over the past three decades. The quality of municipal and industrial effluents in the Mid-Atlantic Region has improved significantly over the past 30 years. Best management practices have been, and are being, implemented throughout the region. But, if the goal is a safe and sustainable environment for humans and other living organisms, we have not yet achieved that goal!

Over the past 10 years, a new way of monitoring has been taking place through the US Environmental Protection Agency Region 3 MAIA geographic initiative. Instead of just measuring physical and chemical indicators in hand-picked locations in lakes, streams, and estuaries, the condition of living organisms and physical and chemical indicators have been measured in a way that can be related to the condition of the environment for the entire region. In addition, new approaches have been developed for using and evaluating satellite pictures to assess environmental condition for the entire region.

### Lessons Learned

Some of the lessons that have been learned over the past 10 years from these new ways of monitoring are that:

***Lesson 1:*** Living organisms—fish, birds, insects, and trees—are stressed throughout the region. Estuarine bottom organisms, stream fishes, and bird communities all show signs of being stressed (Figure 1). It doesn't matter if we look at the region as a whole, on a watershed by watershed basis, or look at individual states, the condition is the same—living organisms are stressed!

***Lesson 2:*** Birds, ecological condition and land use and land cover are all linked. Good to excellent bird communities are associated with forested land in the area. When forests are cut down and the land converted into other land uses such as agriculture or urban development, the condition of the bird community declines.

***Lesson 3:*** Living organisms integrate chemical, physical habitat, pathogenic and other effects around them and provide a cumulative or longer-term record of what has been going on in the environment. Chemical spills, storm-water discharges of pollutants, or other short-term events can be missed if only chemical or physical indicators are measured. Living organisms provide a more complete picture of the condition of the place in which they live.

***Lesson 4:*** Chemical and physical indicators do not provide a complete picture of environmental condition, which is the corollary to the statement above. Yet, many monitoring programs only measure chemical and physical indicators. We need a better, more complete picture.

***Lesson 5:*** Habitat loss and degradation is a major problem throughout the Mid-Atlantic region (Figure 2). In the eastern half of the region, urban sprawl is contributing to this loss and degradation. In the western half of the region, resource extraction—from timber harvesting to mining—contributes to this loss and degradation. Forest fragmentation—cutting swaths and patches out of the forest—contributes to habitat degradation.

***Lesson 6:*** Forest fragmentation is wide-spread throughout the region. The forests in the Mid-Atlantic region are a world-renowned resource (Ritters, et al., 2000). There is only one other place in the world that has as much continuous or interior mid-latitude type forest as the Mid-Atlantic. It is rapidly being fragmented from large, continuous stands to smaller pieces that do not provide the same kind of habitat for living organisms. Many living organisms, from migratory birds to black bears, require large blocks of continuous forest to sustain their populations. The Mid-Atlantic currently has about 70% of its area in forest, 25% in agriculture, 2% in urban area, and about 3% in other land uses.

***Lesson 7:*** Non-native and exotic species have invaded the Mid-Atlantic region and are a major problem. These non-native/exotic species range from pathogens to plants to fish to birds. These non-native species typically out-compete native species because their natural enemies are not present in the Mid-Atlantic region. Combining habitat loss with non-native species introductions results in the loss, in many cases - permanent loss - of native species forever. Loss of biological integrity is a major problem throughout the Mid-Atlantic region.

**What Is Stressing The Living Organisms In the Mid-Atlantic?**
There are many factors that are contributing to the stress observed in living organisms. But it isn't just one thing! If it were, we could find a "magic bullet" and fix it. Most of the factors shown in Figure 2 can, and do, dominate in various ecosystems, but it is the cumulative effects of these factors that contribute to poor ecological condition. In addition, these effects are occurring throughout the hydrologic cycle, including not only watershed runoff and surface water, but also groundwater quantity and quality and atmospheric deposition. These stressors are also linked and can have effects not only in one ecosystem (e.g., forests), but also in multiple ecosystems, such as streams, lakes, and estuaries that receive runoff from terrestrial systems.

**So, What Can We Do To Fix These Problems?**
First, we can review our existing environmental management programs to make sure they are protecting all living organisms and providing a sustainable environment. If we have compliance with environmental permits, yet, living organisms are still stressed, something isn't right. Next, we need to determine the cause-effect relationships between those factors causing the stresses and the biological or ecological endpoints that relate to what we want as a society. To effectively manage and control these stresses, we need to know with some reasonable certainty that we are managing the right ones.

We also need to manage the environment in a more integrated way. It is the cumulative effect of many factors that contributes to environmental problems. Managing on a pollutant by pollutant, or media by media (e.g., water, air, solid waste) approach is not effective enough anymore. We need to manage to control multiple pollutants in multiple media. For a fish, acid rain coming out of the sky is as deadly as acid mine drainage running into the stream from the mine site.

Finally, we need to present this information in ways that are clear and understandable to the public and our decision makers. Decisions need to be made on the effects of urban sprawl, where to put transportation corridors, how to protect large, continuous stands of forests from

fragmentation, and other similar issues. We need to learn how to integrate socioeconomic considerations and indicators into these measures.

**The Goal**

The ultimate goal is to manage for a safe and sustainable environment for humans and other living organisms. The Lessons Learned through the EPA Region 3 Mid-Atlantic Integrated Assessment geographic initiative will help move us toward that goal.

# REFERENCES

EPA. 1999. From the Mountains to the Sea. The State of Maryland's Freshwater Streams. U.S. Environmental Protection Agency, Office of Research and Development. EPA/903/R-99/023. Washington, D.C.

Ritters et al. 2000. Personnel Communication (report *in press*).

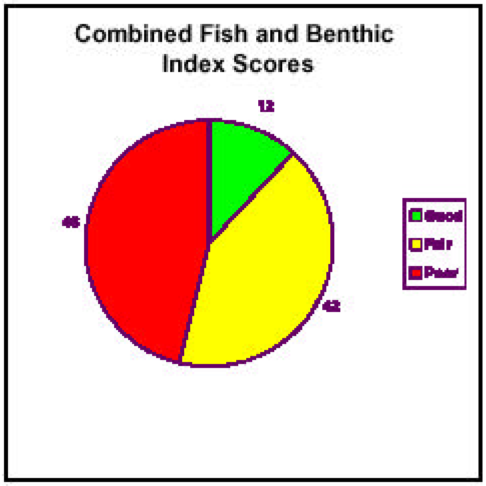



**Figure 1**. Based on the MD Biological Stream Survey data, almost 50% of the stream miles were in poor condition in MD (EPA 1999).

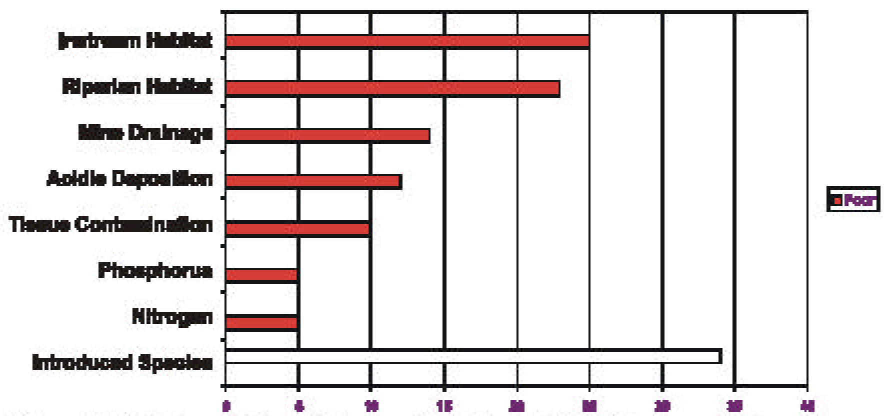



**Figure 2**. Relative ranking of stressors based on Highland stream miles in poor condition. Habitat degradation is the highest ranked stressor. Introduced species include sport fish, which many people do not consider a stressor.

**Evolution of Monitoring Program Strategies for Water Quality Management in the Occoquan Watershed**

Randy Dymond, Thomas J. Grizzard, Adil N. Godrej, and Newland Agbenowosi
Department of Civil and Environmental Engineering, Virginia Tech
E-mail: dymond@vt.edu

## ABSTRACT

Population growth and development in the Northern Virginia/Washington DC region has resulted in increased demands on surface water resources with regards to both water quantity and quality. The Occoquan Reservoir is the terminal surface water impoundment in the Occoquan watershed, and serves as a major source of drinking water for residents of Northern Virginia. In order to protect this vital drinking water source, the Virginia State Water Control Board established the Occoquan Watershed Monitoring Program (OWMP) in 1972 to monitor the water quality throughout the basin.

Since its founding, the OWMP has been successful in building a hydrologic and water quality data acquisition and analysis system that has formed the basis of regional watershed management decision-making for nearly 30 years. This paper describes the evolution of the monitoring activities and an ongoing project for creating a web-enabled GIS and data query system for the Occoquan Watershed. The project integrates a number of the newest GIS, data communications, water quality monitoring, and Internet technologies into a state of the art watershed monitoring and planning system for the Occoquan Basin.

The current project efforts are focused on the development of the relational database, integration with GIS, and a demonstration of the web-enabled GIS environment using ArcIMS. With these tools, it will be possible for governmental agencies, planners, and researchers to not only view intelligent maps of the watershed, but to use the linked databases and maps to ask questions of the database and receive mapped, graphical answers, or to see trends that were not previously apparent.

**Keywords:** watershed monitoring, GIS, data transfer, web-enabled

## INTRODUCTION

In the last decades of the 20th Century, population growth and development in large metropolitan areas of the eastern United States, such as those found in the Northern Virginia/Washington DC region, has resulted in increased demands on surface water resources with regards to both water quantity and quality. Nowhere has this been more apparent than in the Occoquan Watershed, a 600 square mile drainage basin - surface water impoundment system located in the Virginia suburbs to the southwest of the U.S. national capital.

Originally impounded in the late 1950's, the Occoquan Reservoir currently is an integral part of a water supply system serving over 1 million residents of northern Virginia. In the decade of the 1960's, new urban development within the water supply watershed resulted in water quality degradation from increased flows of poorly treated wastewater. In the early 1970's, the Virginia Water Control Board adopted a management strategy, which has come to be known as the Occoquan Policy, and that provided for two key elements: (1) an advanced wastewater reclamation plant to provide state-of-the-art wastewater treatment that would be consistent with a planned indirect re-use to supplement public water supply, and (2) an independent agency to monitor reservoir and watershed water quality, and to advise regulatory agencies, local governments, and public service authorities on the means of best preserving the water supply for its intended use(s). Since 1972, the monitoring program has been operated by the Charles E. Via, Jr. Department of Civil and Environmental Engineering of the Virginia Tech College of Engineering. The Department has established a permanent research facility in Manassas, Virginia, in which the monitoring program and its staff are housed.

Since its founding, the Occoquan Watershed Monitoring Program (OWMP) has been successful in building a hydrologic and water quality data acquisition and analysis system that has formed the basis of regional watershed management decision-making for nearly 30 years. The system has made it possible for the local governments of northern Virginia to successfully deal with the competing uses of urban development (and the attendant wastewater discharges and urban runoff) and public water supply in a critical watershed-impoundment system.

In recent years, however, it has become apparent that new information technologies that may have direct application to watershed management are emerging with the maturing of the Internet, new automated data collection devices, and microcomputer-based Geographical Information Systems (GIS). The potential now exists to offer managers in local government, planning and regulatory agencies, and public utilities with more efficient tools for managing and protecting limited water resources while continuing to meet growing demands for water yield and quality improvement.

A carefully developed GIS can provide the basis for an improved, and better integrated, system for data collection, organization, analysis, and presentation. This paper describes an ongoing project for creating such a web-enabled GIS and data query system for the Occoquan Watershed. The project integrates a number of the newest GIS, data communications, water quality monitoring, and Internet technologies into a state of the art watershed monitoring and planning system for the Occoquan Basin. The overall aim of this project is to develop a centralized GIS for the purpose of enhancing data collection, analysis, and exchange for the Occoquan watershed. The project objectives can be summarized as follows: (1) the transformation of existing water quality and hydrology databases into a relational database system for incorporation into the GIS; (2) development of web-enabled GIS for secure user data mining and visualization functions; (3) development of capabilities for direct data input from hand held devices, wireless remote devices, and other data measuring equipment not currently part of the data collection effort; and (4) incorporation of water quality and hydrology models to provide scenario analysis.

## The Occoquan Watershed

The Occoquan Watershed, which is shown on the map in **Error! Reference source not found.**, is located in northern Virginia and is situated on the southwestern periphery of the Virginia suburbs of the City of Washington, D.C. The basin encompasses six political subdivisions of the Commonwealth of Virginia, including portions of four counties, and the entire land area of two independent cities, as follows: Fairfax County, Fauquier County, Loudoun County, Prince William County, City of Manassas, and the City of Manassas Park.

The watershed lies to the south and west of the U.S. National Capital, Washington, D.C. It is bounded by the Potomac Estuary to the east and Bull Run Mountain to the west. The northern and southern boundaries lie in the Counties of Fairfax and Fauquier-Prince William, respectively. At the location of the Occoquan High Dam, the watershed drains 570 square miles (mi.$^2$).

The major drainages tributary to the Occoquan Reservoir may be divided into two principal sub-basins: Bull Run and Occoquan Creek. Bull Run lies in the northern portion of the basin, and constitutes the principal drainage bounded by Bull Run Mountain on the west, Dulles Airport on the north, and the Manassas urban area on the south. To the east lies the confluence with Occoquan Creek in the upper reaches of the Occoquan Reservoir. The aggregate drainage of Bull Run and its tributaries above the Reservoir is 185 $mi^2$.

Occoquan Creek is formed by the confluence of Broad Run and Cedar Run. Broad Run, as noted previously, drains the western extreme of the basin. Lake Manassas, which is the principal water supply of the City of Manassas, and is also an artificial impoundment, lies within the Broad Run drainage. Below the confluence of Cedar Run and Broad Run lies Lake Jackson, which is an impoundment of Occoquan Creek. Lake Jackson was originally constructed in the 1930's as a hydroelectric power production facility. At the present time, however, the lake is the centerpiece of a residential area lying to the south of the City of Manassas, and is maintained for recreational purposes only. It should be noted that substantial pollutant loading reductions to the Occoquan Reservoir result from the presence of these upper-basin impoundments. Below Lake Jackson, Occoquan Creek flows directly into the tail waters of the Occoquan Reservoir, and drains an aggregate of 343 $mi^2$.

The remaining direct drainage to the Occoquan Reservoir originates in small streams in both the Counties of Fairfax and Prince William. The total drainage distributed among these small tributaries is 42 $mi^2$., which is slightly over seven percent of the total drainage area. In Fairfax County, these small tributaries include Pope's Head Creek, Wolf Run, and Sandy Run. Similarly, in Prince William County, one may identify Hooes Run, which enters the Reservoir directly upstream of the high dam.

**Water Quality Considerations**

Early in the decade of the 1960's, the urban growth began to reach into the upper Occoquan Watershed in unprecedented (and unanticipated) proportions. Coincident with the onset of accelerated population growth, a number of wastewater treatment plants were constructed and/or expanded in western Fairfax County and central Prince William County, resulting in substantial increases in the discharge of domestic wastes to the receiving waters of the basin. By the latter part of the decade of the 1960's, eleven (11) publicly owned treatment works (POTW's) of conventional secondary design were discharging an average of nearly three (3) million gallons per day (MGD) of treated wastewater to the basin. The quality of the plant effluents was quite variable, and no provisions were made for the removal of plant nutrients from the discharges. In addition, the percentage of basin area devoted to urban land uses began to increase substantially, raising the input of urban stormwater into the system. Increased conventional agricultural activity in the western basin, along with the application of chemical fertilizers, resulted in greater soil erosion, and the accompanying loss of nutrients in surface runoff.

Because the Occoquan Reservoir had become an irreplaceable resource for the citizens of northern Virginia, it was apparent that steps would be required to insure the long-term viability of the reservoir as a public water supply. In 1968, the Virginia State Water Control Board (SWCB) commissioned a study of the Reservoir and its tributary streams by the consulting engineering firm of Metcalf and Eddy (1969), with the goal of developing a management plan for the surface waters of the basin. That study, completed in 1970, stated that the reservoir was "highly eutrophic...", and further, that "the sewage plant effluents are mainly responsible for the advanced stage of eutrophication occurring..." Metcalf and Eddy study concluded with the recommendation that three alternatives be considered for future management of water quality in the reservoir:

-Wastewaters from the basin be exported to another watershed.

-Advanced wastewater treatment practices be adopted; treated waters be exported for re-use, and basin population be limited.

-Advance wastewater treatment practices be adopted with effluents remaining in the watershed, and basin population be limited.

## The Occoquan Policy

In July of 1971, after considering the recommendations of the Metcalf and Eddy report, the SWCB adopted A Policy for Waste Treatment and Water Quality Management in the Occoquan Watershed (VSWCB, 1971). Recognizing the practical limitations imposed by both the inter-basin transport of wastewaters and the imposition of population limitations, the Occoquan Policy, as it has come to be known, was based on a modification of the third option shown above. A milestone in water quality management in the Commonwealth of Virginia, the Policy included an implicit recognition that an indirect re-use of treated wastewater would become the operational norm in the Occoquan Watershed. It also recognized that extraordinary measures would be required to protect the public health in a situation where a water body was to be subjected to the competing uses of wastewater disposal and public water supply. In addressing this, the document not only specified the type of waste treatment practice to be adopted on a basin-wide scale, but it provided for an ongoing program of water quality surveillance to quantify the success of the water quality protection effort.

## The Occoquan Watershed Monitoring Program

The Occoquan Policy, in addition to mandating the adoption of regional advanced wastewater treatment practices at all new regional wastewater treatment plants in the Basin, went so far as to establish an innovative requirement for the establishment of an independent entity for the purpose of water quality surveillance and evaluation, and "to insure that performance levels are maintained at the ... plant, and that the effects of discharges and urban run-off (sic) are known." The entity charged with the creation and governance of the monitoring program was the Occoquan Watershed Monitoring Subcommittee (OWMS).

The OWMS was given the authority to create an independent facility to conduct the required monitoring program, using funds contributed by the wastewater generators and the finished water purveyor. In practical terms, this means that funds were contributed by the counties and cities in the Basin, and by the Fairfax County Water Authority. The resulting facility, the Occoquan Watershed Monitoring Laboratory (OWML), was established by the Virginia Tech Department of Civil Engineering. The laboratory began its on-site operations in 1972, and has conducted comprehensive studies of receiving water quality, and effects of the AWT discharges to the present time.

In the course of its studies, OWML has developed a comprehensive database of water quality in the Occoquan Basin, and has been instrumental in making determinations in a number of areas which have proven to be critical to the ongoing management of water quality:

Determining the suitability of AWT effluent for indirect discharge into a public water supply;

Providing information required for consideration of alternative treatment practices at the

AWT plant;
Providing receiving water data for use in contemplating AWT plant expansions;
Providing information on water quality effects and cost-effective control of nonpoint sources of pollution.

## DEVELOPMENT OF A GIS AND DECISION SUPPORT SYSTEM

Over the years, since its founding, the OWML has been at the forefront in utilizing new and state of the art tools to enhance its monitoring activities in the watershed. During this time period, significant advances have been made in other fields that, collectively, enable OWML to bring the monitoring data into a cohesive geographic information system (GIS) that can present mapped objects, such as reservoirs, outfalls, or measuring devices, together with linked monitoring data.

Developments in computer hardware and software, networking, aerial and satellite imagery, wireless communication, hand-held computers, and the Internet are revolutionizing many disciplines, including watershed monitoring and management. It is possible for governmental agencies and planners to not only view intelligent maps of the watershed, but to use the linked databases and maps to ask questions of the database and receive mapped, graphical answers, or to see trends that were previously hidden. The OWML and the GIS/CAD research group within the Civil and Environmental Engineering Department of Virginia Tech are currently developing system to integrate a number of the newest GIS, data communications, water quality monitoring, and Internet technologies into a state of the art watershed monitoring and planning system for the Occoquan Basin.

One of the major goals of this system is to provide a web-based watershed map navigation tool with the ability to mine down through layers of physical, chemical, and biological data for the watershed. As a longer-term goal, we will develop on-line access to real-time data from some key stations, including flow, temperature, DO, pH, turbidity, etc. Furthermore, we see the development of GIS linkages to water quantity and quality models as critical steps in developing a set of usable decision support system tools for watershed managers.

The primary goals of the system are oriented towards the integration of the latest GIS, computing, and communications tools into the existing OWML database and work flow. Stated another way, the overall aim of this project is to develop a centralized GIS for the purpose of enhancing data collection, data analysis, and data exchange for the Occoquan watershed. Individual goals oriented toward this objective are summarized below:

1. Demonstrate a rudimentary web-enabled GIS for OWML data.
2. Re-design the current database structure and convert the existing data into a relational table structure with affiliated testing by OWML staff, form and report development, data conversion, and staff training.
3. Acquire and develop a series of initial base GIS data layers.
4. Insure that the computing system (hardware, software, and networking) will support a centralized GIS.
5. Web-enable the GIS for secure user data mining and visualization functions.
6. Develop capabilities for direct data input from hand held devices, wireless remote devices, and other data measuring equipment not currently part of the data collection effort.
7. Establish programming linkages from the GIS to hydrologic, ecological, and water quality models.
8. Maintain important ongoing activities of GIS data acquisition and customization.

The demonstration of a rudimentary web-enabled GIS for OWML data has been accomplished. However, due to the current database structure and the lack of adequate base GIS layers, the web page only shows a minimal amount of data.

## GIS and Data Query System Design

The structure of the GIS system consists primarily of four modules. These are the Display, Input, Database, and Analysis modules. **Error! Reference source not found.** shows an illustration and schematic of how the modules are related. Discussion of these primary modules follows.

### *Input Module*

This module, which is not yet fully functional, will primarily gather data from different sources. The instruments/gauges sub-module will provide for connection and downloading data from automated instruments. This data may be fed live into the database or manually uploaded periodically. Current efforts are being focused on the installation of a modem equipped data logger at one of the sampling station to enable real time data acquisition and transmission to the OWML facility in Manassas, Virginia through a phone line. Subsequently the transmitted data will be retransmitted over the internet to the database server in Blacksburg, Virginia for display on the internet. The field data sub-module will be designed to permit data entry by OWML field personnel after periodic visits to sampling stations. The lab sub-module will provide an interface for OWML lab personnel to enter lab analysis results. The GIS input sub-module will comprise layout and metadata standards for GIS data layers entered into the system.

### *Database Module*

The database module constitutes the core of the GIS system. It contains a relational database structure linked to the spatial GIS layers to provide spatial reference for hydrologic, meteorological, and water quality data. The current relational database system being used is Microsoft Access. There are plans to migrate to another database system such as Oracle or Microsoft SQL Server once the system is fully functional.

### *Analysis Module*

The analysis module will allow for data interfacing to various computational models used for watershed decision support. Typical water quality and hydrology models such as WASP, Qual2E, HECHMS, and HECRAS could be incorporated in this module. Integrated decision support and visualization modules are also intended for this module. Ongoing research is vital to the updating the policies in place to protect the watershed and this portion of the GIS system will provide the types of tools necessary to affect decision-making processes.

### *Display Module*

This module primarily provides a window to the users of the system, enabling them to interact and acquire information from the system. The intended audience includes three classes of users: agencies, scientists/researchers, and the general public, each with programmable levels of access and security. Agencies would use the system to obtain up to date information and graphic displays of pollutant activity or meteorological data in the watershed. Scientists and researchers with appropriate permissions will be able to readily access data for research on the watershed, including the ability to link spatial data to models. Public stakeholders can be involved in maintaining the good health of the watershed by illustrating the various water demands and pollutant levels in the watershed. Public education programs can also be linked to this interface to promote education general knowledge about issues like storm sewer drains and their impact on surface water quality, or how improper disposal of household pollutants can affect the water quality.

Technically, user interface issues can be very difficult. In this instance, primary issues include, 1) design of various levels of user interface with appropriate security, 2) providing downloadable documents, perhaps in the Adobe PDF file format, and 3) dealing with raster image compression and display with internet browsing programs.

### *Current State of the GIS and Data Query System Design*

In its present state, the web-enabled system allows users to perform queries on the water quality and flow database. Figure 3 illustrates the flow of information starting from the user's request and ending with the returned results from the server. The user is able to select the desired station by using an interactive map that is being served using ArcIMS 3.0. The HTML viewer version of the ArcIMS is the one currently used to serve the interactive map (see **Error! Reference source not found.**). After selecting the desired station, the user is taken to a second screen (see **Error! Reference source not found.**) where the desired time period for the query and the parameters can be selected. At this stage, one of the three output formats (HTML, chart, or excel file) must also be selected. The results from the query are reported in three available formats: HTML table, chart, and downloadable excel file.

The html output prints the results in a simple html format. The excel file option provides the results in a downloadable excel file. This option is particularly intended for users intending to download the queried data for use in their own local analysis. The graph or chart option returns the queried data in the form of an excel type chart. Two options are being used for generating the chart. The first option shown in **Error! Reference source not found.** utilizes the chart object in available in Microsoft Office Web Components (OWC)®. The Office Web Components are a set of COM controls that provide a vehicle for interactive spreadsheet modeling, data analysis and data reporting. The library consists of four main components: a Spreadsheet component, a Data Source component, a Chart component and a PivotTable component. Though the use of the OWC provides a faster response to generating spreadsheets and charts, it lacks the flexibility for formatting the charts. It also provides fewer options for selecting chart types. An alternative means of developing the chart was therefore developed. This alternative involves the use of a separate program (called a ChartServer) that runs outside the web server (see **Error! Reference source not found.**). The external chart server receives the results for the queried data and uses the information to generate the chart that is then passed to the web server as a gif image.

### **Current System Configuration**

Existing computing infrastructure within the Virginia Tech Civil and Environmental Engineering Department is being used as a basis for this project. A Dell PowerEdge 4300 Server configured with 512 MB RAM and 54 GB of RAID5 storage is serving as the processing and storage center for the project. The server is running on a Microsoft Windows 2000 Server operating system. Also, the Microsoft Internet Information Server 5.0 (IIS) is being used as the web server for the project. The IIS provides both a web server and FTP server that is useful for periodic remote update of the water quality and flow database from the OWML site in Manassas, Virginia.

The interactive map is being served using ArcIMS 3.0. Most of the data query process is designed using active server pages (ASP). The ASP is used to integrate the Internet map server (ArcIMS) and the water quality database. Other software being used are Arcview GIS 3.2 for preparation of GIS data layers for the ArcIMS software; AutoCAD Map 2000 which is used to edit the spatial data; and Microsoft Access 2000 which is used to manage the water quality and flow database.

### Future Activities

*Develop capabilities for direct data input and display from new hand held devices, wireless remote devices, and other data measuring equipment not currently part of the data collection effort.*

The quality of decisions depends on the amount, quality, and timeliness of pertinent information. Therefore, as new technology is developed in the data collection arena, efforts to utilize the products in the Occoquan GIS system must be made. These products include, but are not limited to; GPS enhanced devices, hand-held data loggers, wireless real-time transmitters, video cameras, and new airborne sensors. As a large watershed in a very important region of the United States, the Occoquan basin is an ideal partner for companies wishing to research or showcase new products of this type. Discussions with a major manufacturer of data collection devices (Sutron Corporation) have already been initiated. The Sutron Corporation provides real-time data collection, telemetry, and technical expertise to monitor, control, manage, model, and forecast activities in the areas of hydrology, meteorology, and water management. Incorporating these types of data sources into the GIS system will be a continuing effort.

*Establish programming linkages from the GIS to hydrologic, ecological, and water quality models*

Linking models to GIS based information is of great benefit to watershed managers and political decision makers. Modeling current and future conditions can clearly illustrate how certain development decisions can impact conditions within a watershed. Floods, erosion, water quality, ecological health, economic and many other issues can be modeled more accurately with geographically specific information contained in a GIS. Implementation of this goal will depend upon the priorities determined by OWML constituencies. Success in early models may lead to an increased interest in continuation of these efforts.

*Maintain important ongoing activities of GIS data acquisition and customization*

There are a number of ongoing activities foreseen for this project and the extent to which they will be pursued depends upon the priorities of the OWML constituencies. GIS spatial data layers are changing daily as the world changes. New data layer acquisition will be necessary throughout the lifetime of this project. Each month brings changes to the watershed (e.g., land use, water demand, roadway networks, nutrient loadings, etc.) Weather systems are dynamic and today's drought may change into tomorrow's flood. Updating and modifying the GIS system to accommodate current needs will most certainly be a continuing effort aimed at providing the best information for decision makers in the watershed.

## REFERENCES


Metcalf and Eddy, Inc. 1970. *The Occoquan Reservoir Study*. Boston, Massachusetts.

VSWCB. 1971. *A Policy for Waste Treatment and Water Quality Management in the Occoquan Watershed*. Virginia State Water Control Board, Richmond, Virginia.

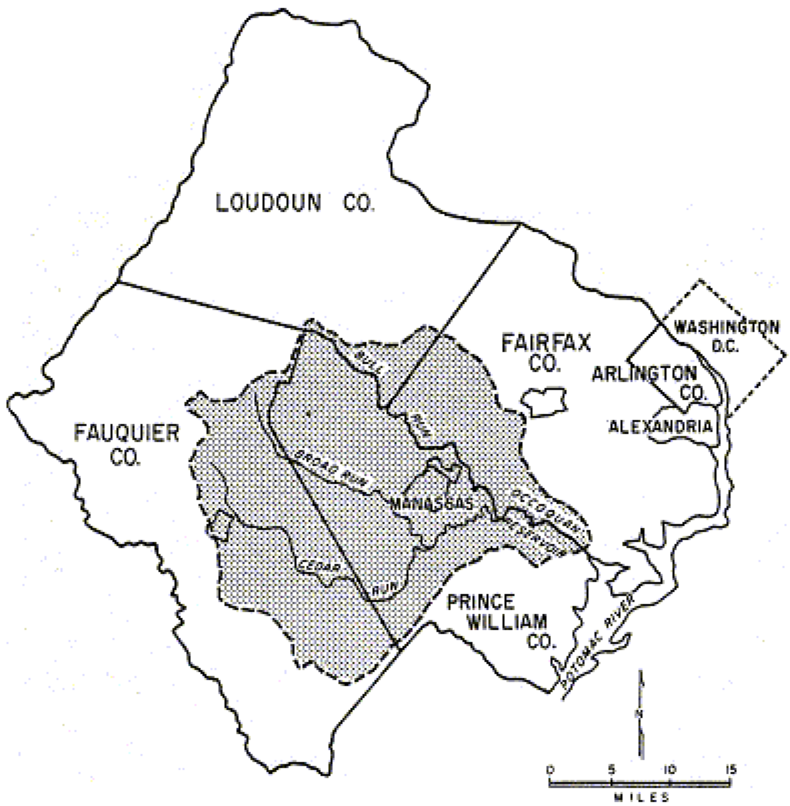



**Figure 1. The Occoquan Watershed**

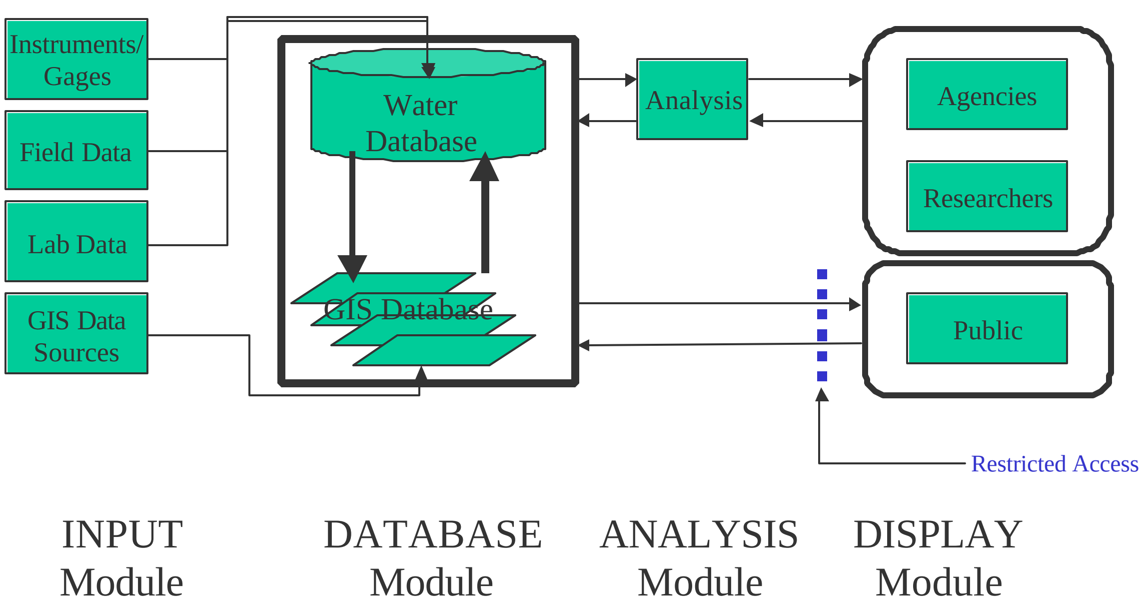



**Figure 2. Configuration of the web-enabled GIS**

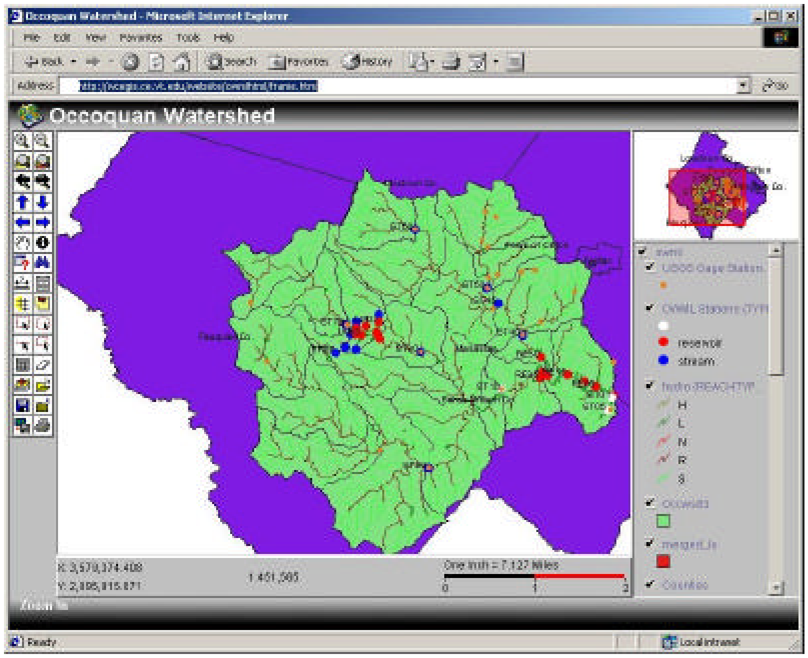



**Figure 3. Interactive Map for Station Visualization**

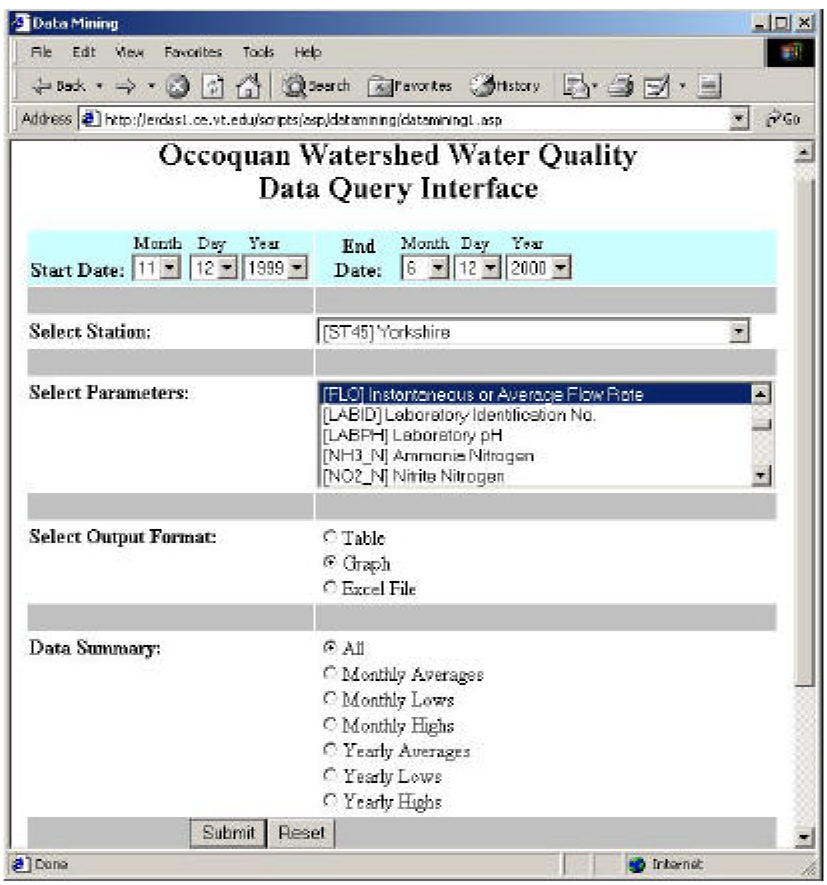


**Figure 4. Input Screen For Data Query**

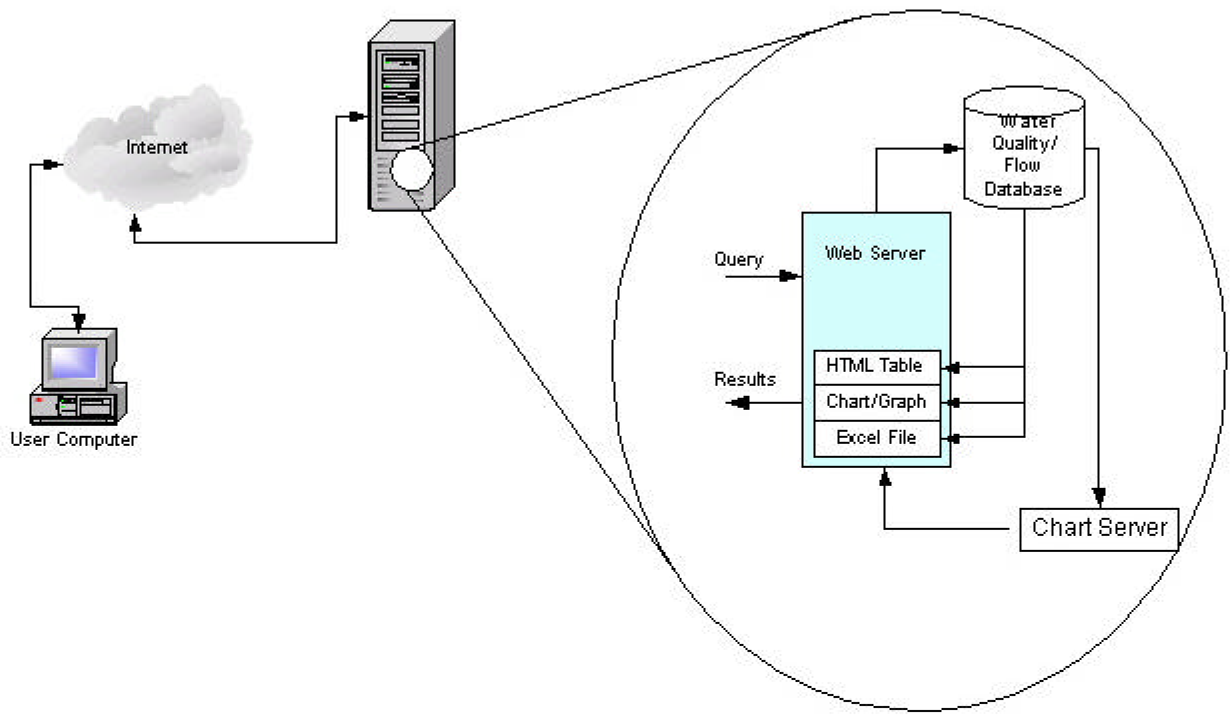



**Figure 5. Flow of Information**

# Use of a Hydrogeologic Framework in Relating Surficial Hydrogeology to Shallow Ground-Water Quality in the Mid-Atlantic Coastal Plain

Tracy C. Hancock, USGS, Richmond, Virginia
Ator, S.W., USGS, Baltimore, Maryland
Denver, J.M., USGS, Dover, Delaware

E-mail: thancock@usgs.gov

## EXTENDED ABSTRACT

Local topography and the texture and chemical composition of surficial and near-surface geologic materials affect the movement of water and the transport and reactivity of chemical constituents, which in turn affect the quality of ground-water resources. In order to evaluate the effects of variable surficial hydrogeology on shallow ground-water quality, the USGS analyzed water-quality data collected from unconfined aquifers of the Mid-Atlantic Coastal Plain (New Jersey through North Carolina) in the context of a newly developed surficial hydrogeologic framework. The results illustrate the importance of considering surficial hydrogeology in the design of water-monitoring programs and the interpretation of results.

A surficial hydrogeologic framework was developed based on the variable topography, texture, and depositional history of surficial sediments as delineated by newly available regionally consistent maps of the Mid-Atlantic Coastal Plain. The framework was designed to define subregions of the Coastal Plain within which the natural physical factors controlling the occurrence and movement of chemicals into shallow ground water and small streams are relatively consistent. Seven hydrogeologic subregions were distinguished by physiography and the bulk texture of surficial sediments.

All seven subregions represent areas of primarily unconsolidated siliclastic sediments along a continuum of texture and drainage characteristics, from sand and gravel that provide well-oxygenated conditions to fine-grained sediments providing poor drainage and reducing conditions. The *Coastal Lowlands* (subregion 1) is primarily flat, low-lying, and poorly drained with numerous wetlands. Sediments are primarily fine-grained estuarine or near-shore deposits, which typically contain organic-rich matter. The Middle Coastal Plain (MCP) was subdivided into four subregions which reflect differences in sediment texture and stream dissection. The *MCP-Mixed* (subregion 2) is moderately dissected with laterally and vertically variable sediments, including coarse sands associated with shorelines and estuarine and lagoonal silts and clays. The *MCP-Fine* (subregion 3) is more heavily dissected with predominantly fine-grained sediments at the surface. The *MCP-Sands with Overlying Gravels* (subregions 4 and 5) contain predominantly well-drained, permeable, weathered, coarse-grained deposits, but subregion 4 is moderately dissected and subregion 5 is completely incised in most places. The *Inner Coastal Plain* (subregion 6) includes an outcrop belt of deeply weathered sediments. The *Alluvial and Estuarine Valleys* (subregion 7) contains a mixed sequence, with coarse alluvial deposits at depth and finer, estuarine sediments near the surface.

We compiled data on major ions, nutrients, and pesticides in water samples collected from 1987 through 1997 by the U.S. Geological Survey and other Federal and State agencies from 533 wells in unconfined aquifers in the Mid-Atlantic Coastal Plain. The data were screened to eliminate bias toward sites at the same location or sites sampled multiple times. The data were grouped by subregion and evaluated to distinguish differences in the quality of water between subregions. Geographical data for the Mid-Atlantic Coastal Plain were compiled for comparison to water

quality, both within each hydrogeologic subregion and independent of subregion. For this study, Multi-Resolution Landscape Characterization (MRLC) land covers were combined to five major classes: agriculture, forest, urban land, wetland, and barren land. Data analyses were designed to evaluate the relation of both hydrogeologic subregion and land use to measured ground-water chemistry.

Concentrations of most major ions and nutrients are similar among many of these subregions, reflecting similar geology. However, water-quality differences were observed in subregions with distinctly different sediment texture, topography, geochemistry, and land use. For instance, concentrations of most major ions and reduced nutrients are higher in subregion 1, which is a poorly-drained area of abundant organic matter and little dissolved oxygen. Dissolved oxygen and nitrate concentrations are the highest in subregion 4, which has similar land use to subregion 1 but coarser, more weathered sediments and better drainage.

Pesticide concentrations are related to both hydrogeology and land use. Because pesticides are applied to the land surface, land use within individual subregions is a source of bias not eliminated by the initial screening of data. Detection frequencies of commonly used agricultural pesticides are greater in subregion 6 than in subregion 1. In subregion 1, fine grained, organic-rich soils most likely retard the transport of pesticides from the land surface to ground water. Subcropping geologic units in subregion 6 are deeply weathered where exposed, and can be quite permeable, promoting the transport of pesticides from the land surface to the ground water. The soil and hydrogeologic conditions in subregion 4 are also particularly favorable for the transport of pesticides. However, detection frequencies were typically lower in subregion 4 than subregion 6, probably because a larger proportion of data from subregion 6 was collected in agricultural areas, where there is a greater likelihood that pesticides are applied.

## REFERENCE

Ator, S.W., J.M. Denver, and T.C. Hancock. 2000. Relating Shallow Ground-Water Quality to Surficial Hydrogeology in the Mid-Atlantic Coastal Plain. Proceedings of the National Water Quality Monitoring Conference, April 25-27, 2000, Austin, TX. p. 409-423.

# Pollution Source Identification Strategies for Coastal New Hampshire

Jones, Stephen H., University of New Hampshire, Durham
Landry, Natalie, New Hampshire Department of Environmental Services, Concord

E-mail: shj@hopper.unh.edu

## EXTENDED ABSTRACT

The Seacoast area of New Hampshire has experienced rapid population growth and accompanying land development during the past few decades. Shellfish-growing and recreational waters are highly valued resources that are presently threatened by a variety of contaminants. Over the past ten years, much progress has been made in improving water quality, opening shellfish beds, and building more effective state programs to maintain environmental quality in the Seacoast. The State of New Hampshire and the University of New Hampshire have worked together to provide information on the status, trends and sources of microbial, nutrient and toxic contaminants in coastal waters. The work here summarizes the progress made from 1990 to 2000 in identifying sources of microbial and toxic chemical contamination in coastal New Hampshire in the context of an overall pollution source identification strategy for the area. Data compiled over this ten-year period are used to illustrate the context and progression of specific steps taken to identify sources of residual microbial contaminants in surface waters that currently limit their use.

### Pollution Source Identification

A great deal of scrutiny of the state's shellfish program occurred as a response to the closure of the Hampton Harbor clam flats in 1988. In 1990, the recreational harvest program was run as small, relatively insignificant parts of three state agencies. However, the public outcry from the complete closure of the clam flats in Hampton Harbor, which left only a small amount of oyster beds open for harvesting in the whole state, caused Seacoast legislators to investigate how to remedy the situation.

The most obvious first step was to eliminate significant point sources by improving the efficiency of municipal wastewater treatment facilities (WWTFs). The Seacoast area had recently experienced a population and development boom, and many of the WWTFs had not been upgraded for years. By 1996, ten of the seventeen coastal municipalities significantly improved wastewater treatment at their WWTFs. Some of the most important improvements included: upgrading of the treatment and disinfection systems and eliminating all but one Combined Sewer Overflow (CSO) in Exeter in 1990: the upgrading to secondary treatment and adding UV disinfection at the Dover WWTF: elimination of 10 CSOs and upgrading the WWTF to advanced primary treatment with dechlorination in Portsmouth: and the completion of a secondary WWTF in Seabrook which eliminated all septic systems in the town.

Monitoring by the State shellfish program and by UNH showed decreasing fecal coliform concentrations in estuarine waters in some areas most influenced by point sources. For example, fecal coliform concentrations in the Squamscott River downstream of Exeter decreased significantly after the upgrade of the Exeter WWTF . However, residual use-limiting concentrations remained in many shellfishing areas of Great Bay. At the same time during the early 1990s, many of the municipal and industrial point sources of toxic chemical contaminants had been significantly reduced. Even so, monitoring for bioexposure in blue mussels through the

Gulf of Maine Gulfwatch program showed residual contamination well above background at most coastal sites in New Hampshire. With the decreasing significance of point sources of contamination, residual nonpoint sources became the focus of continued pollution source identification.

In 1992, UNH and the New Hampshire Coastal Program (NHCP) teamed up to begin a strategy for identifying nonpoint sources of pollution in specific coastal watersheds, complimenting a similar watershed approach initiated by NH Department of Environmental Services (NHDES). The Oyster River watershed was chosen as it had characteristics of a typical NH watershed, including a tidal dam, an urban area centered around the dam, a WWTF discharging into the tidal river, and a rapidly developing suburban area in the sub-watersheds bordering the tidal river. Water quality analyses revealed elevated fecal coliform concentrations in small tributaries up to the tidal river and increasing concentrations along a transect of the river from the mouth to the tidal dam. The study documented many different types of sources of microbial contamination, including septic systems, livestock, municipal sewer lines and, occasionally, the WWTF itself. The approach developed for the Oyster River watershed was applied to the Squamscott River watershed, and the similar results for both studies formed the basis for more detailed pollution source identification studies throughout coastal NH by NHDES.

**Major Findings**

Numerous studies of bacterial water quality in coastal NH waters strongly suggested significant loading of microbial contamination occurred during storm events. A graphic example is the time series off enterococci concentrations in Great Bay during the summer of 1991, before and after Hurricane Bob. Enterococci concentrations in the bay had been extremely low (<1/100 ml) during the hot, dry summer until skyrocketing to well above 1000/100 ml following the hurricane, during which ~6.8 inches of rain fell in two days. A three-year cooperative study of storm event influences on the water quality in the six major tributaries to the Great Bay Estuary was initiated in 1993 by UNH, NHDES, NHCP, and the NH Fish and Game Department. Water samples were collected during eight storm events and eight dry weather days at paired sites just above and below tidal dams at the six tributaries. The adverse effects of storm events on bacterial water quality was obvious at every tidal and non-tidal site, with significantly greater concentrations of fecal coliforms, *Escherichia coli* and enterococci, at all sites following storm events compared to dry weather. Simultaneous studies by NHDES and NHCP showed detectable and occasional elevated concentrations of zinc, copper, lead and aluminum in water from freshwater and estuarine sites along the Exeter, Squamscott, Lamprey and Oyster rivers. Some of the results suggested increased loading during storm events. These results provided convincing evidence for concerted efforts to determine the mechanism and identify significant sources of stormwater-borne microbial contaminants.

The next steps in the quest for identifying pollution sources in Seacoast NH involved detailed studies of specific types of potential sources. Taking advantage of the impending completion of a new WWTF in Seabrook, a three-year study of the impacts of existing septic systems on water quality was conducted by UNH. Analysis of soil samples revealed significant subsurface transport of fecal contamination away from septic systems toward the estuarine waters. However, bacterial contamination of groundwater was more difficult to document. The removal of septic tanks and drainage beds following tie-in to the WWTF caused elimination of bacterial contamination in groundwater and a dramatic shift from ammonium to nitrate in groundwater below a residential septic system.

The most significant source of contamination in coastal NH was suspected to be linked to stormwater runoff. In 1996, NHDES began assessments of dry weather sources of bacteria in stormwater drainage systems in the urbanized areas of the Great Bay Estuary. Surprisingly, many illicit sources were found in all of the areas studied. UNH followed these screening studies with more detailed studies in Exeter, Portsmouth and Dover. Dramatic levels of bacterial loading were observed in many storm pipes, and NHDES began identifying sources and eliminating discharges. A drastic decrease in bacterial loading was observed at one pipe after NHDES had eliminated a source. Further investigations are continuing by prioritizing the storm drains that are contributing the largest bacterial loads in an effort to facilitate remediation efforts.

The Gulfwatch mussel monitoring program was expanded in New Hampshire to help determine whether suspected sources of toxic chemical pollutants should be a concern for estuarine biota. This more detailed program was successful in determining that a number of suspected sources of toxic contaminants were not problems, while other sources were still a concern. The continuation of the program for the past nine years also provided valuable temporal trend information that has helped to document effects of remediation activities in NH.

Residual bacterial contamination still limits shellfishing and other uses in some coastal areas. Traditional pollution source identification studies are laborious and sometimes cannot be directly linked to improvements in water quality. The state of New Hampshire is developing a microbial source tracking method to identify sources of residual microbial contaminants in surface waters that currently limit their use. Studies to test the applicability of the new ribotyping approach will be focused in two areas. One is a small sub-watershed, Varney Brook that has shown wide variations in bacterial concentrations. The variation has been observed spatially and temporally, as well as in relation to storm events. The other area is Hampton Harbor, where unknown events cause relatively frequent elevated concentrations of bacteria, even though overall geometric mean concentrations of fecal coliform are below the 14 MPN/100 ml standard. Both sites have a mixture of potential sources of contaminants. The goal of testing the ribotyping approach is to determine what type of source is responsible for the contamination at both sites to aid efforts to identify specific controllable sources for elimination.

The water quality of coastal New Hampshire has greatly improved during the 1990s as a result of the various studies, source identification, and elimination activities. Many more acres of shellfish beds are currently open for recreational harvesting, and the more detailed, overall assessment of water quality provides a more sound basis for managing the different uses of coastal waters. Promising investigatory tools are under development with the goal of precisely identifying bacterial sources. The partnerships between the University of New Hampshire and the various state agencies has resulted in improved water quality and increased use of the waters by the public in coastal New Hampshire.

# Bacterial Source Tracking: A Tool for Total Maximum Daily Load Development

James Kern, Byron Petrauskas, Phillip McClellan, Vernon Shanholtz

MapTech, Inc.
Blacksburg, VA 24060

Email: jkern@maptech-inc.com

Charles Hagedorn, Professor
Crop Soil Environmental Sciences Department
Virginia Tech

## ABSTRACT

In the United States, pathogen contamination (e.g. fecal coliform, bacteria, and *E. coli*) is the second most frequent cause given for surface waters being placed on their states 1998 303(d) list of impairments, with 5,281 occurrences. Accurate data and modeling techniques are required for the development of total maximum daily loads (TMDLs) for these waters. Model accuracy, along with source inventory, have been questioned with reference to TMDLs developed to date. The need to accurately model all sources of fecal coliforms is critical, as TMDL implementation will shift greater responsibility to those sources that have been modeled. Various approaches have been taken in the development of TMDLs. For wildlife sources, these approaches have ranged from considering one species and one delivery mechanism to considering multiple species and multiple delivery mechanisms. MapTech, Inc. is in the process of developing calibration techniques for models simulating fecal coliform fate and transport by incorporating the results of bacterial source tracking (BST). This paper demonstrates how BST can be used in the TMDL process. A minimal amount of BST data can be used to verify the presence or absence of sources in a watershed, which allows for a qualitative analysis of the source inventory and modeling effort. With adequate data, the initial calibration of the model can be improved to more accurately reflect the contribution of different sources. In the future, these calibration procedures will be available to enhance confidence (modeler and public) in model accuracy. A model calibrated in such a fashion and linked to measured data should be more widely accepted by the stakeholders in a TMDL setting.

**Keywords:** BST, TMDL, fecal coliform, fecal streptococci

## INTRODUCTION

All states have been required to establish Total Maximum Daily Loads (TMDLs) of pollutants in areas where surface waters are considered impaired based on federal and state water quality standards. The TMDL (for a specific pollutant, in a given drainage area) is the largest loading of the pollutant that will allow the impairment to be corrected. Upon establishing TMDLs for impaired waters, all states are required to implement procedures for restricting total daily loadings of pollutants to meet water quality standards. States use various indicators of fecal contamination in setting water quality standards (e.g. fecal coliforms and *E. coli*). These indicators can originate from wildlife, human (e.g. failing septic systems and straight-pipes), and livestock sources.

Currently, no state monitors for contamination from distinct sources, as Bacterial Source Tracking (BST) is an emerging field and the cost of analysis has been prohibitive. Bacterial Source Tracking has been used to identify the presence or absence of sources in seven TMDLs developed to date (VADEQ/VADCR, 2000a, 2000c-h). However, determination of the sources of fecal contamination is typically achieved through analysis of land use, animal populations, and demographics. While models are calibrated with regard to the overall concentrations of the fecal indicators (e.g. fecal coliforms and *E. coli*), no TMDL model to date has been calibrated with data indicating the relative contributions from specific sources (i.e. livestock, human, and wildlife). Determination of fecal contamination sources, loads, and transport mechanisms is thus left largely to professional judgment during the modeling process. When the lack of adequate data leads to a source (e.g. wildlife) being underrepresented in the model, other sources (e.g. livestock and human) can face larger load reductions to compensate for the underrepresented source, and the load reductions specified are not likely to result in water quality goals being met during implementation.

To composite data from state lists of impaired waters, the USEPA groups all fecal-indicator causes as "pathogens." Pathogens were the second most frequently recorded cause of impairment in the 1998 303(d) lists of impaired waters approved by the USEPA, with 5,281 impairments listed due to pathogens. This number does not include impairments listed more broadly as fish consumption advisories, many of which are due to fecal contaminants. State agencies and the public at large have an interest in improved methods for determining the sources of fecal contamination, as improved source identification can increase confidence in fair and effective solutions.

## BACTERIAL SOURCE TRACKING

Bacterial Source Tracking methods can be subdivided into three basic groups: Molecular, Biochemical, and Chemical. Molecular (genotype) are typically referred to as "DNA fingerprinting" and are based on the unique genetic makeup of different strains, or subspecies, of fecal bacteria. Biochemical (phenotype) methods are based on an effect of an organism's genes that actively produce a biochemical substance. The type and quantity of these substances produced is what is actually measured. Chemical methods are based on finding chemical compounds that are associated with human wastewaters, and would be restricted to determining if sources of pollution were human or not. All of these procedures are under development and few studies have been completed that compare the methods against each other. Since TMDL development requires differentiation between wildlife, livestock, and human sources, chemical methods would not be appropriate. All monitored data presented in this paper were measured using a biochemical-profiling method (i.e. antibiotic resistance analysis). Antibiotic resistance analysis was selected because it has been demonstrated to be a reliable procedure for confirming the presence or absence of human, livestock and wildlife sources. Compared to DNA fingerprinting, biochemical profiling is much quicker, typically analyzes many more isolates (e.g. hundreds per week vs. a few dozen per week for DNA analysis), is less expensive (e.g. $10 to $20 per isolate vs. $30. to $100. per isolate for DNA analysis), has survived limited court testing, and has undergone rigorous peer review from the academic community. Additionally, observation of an increased number of isolates allows for an estimate of the relative proportions of the fecal indicator (e.g. fecal streptococci) originating from different sources.

The antibiotic resistance analysis method used in this study separated fecal sources based on patterns of antibiotic resistance in the fecal streptococci (including the enterococci). The premise is that human fecal bacteria will have the greatest resistance to antibiotics and that livestock and wildlife fecal bacteria will have significantly less resistance (but still different) to the battery of antibiotics and concentrations used. The analyses performed for this study tested each isolate on 30 different combinations of antibiotic type and concentration. Fecal bacteria are grown in wells in microtiter plates and then replica-plated onto a series of agar plates, each containing one specific antibiotic concentration. After incubation, each isolate is scored for growth or no growth on each plate and a resistance pattern emerges that can be used in source differentiation. Confidence in BST techniques is measured by the level of separation of isolates from known sources, represented as the percentage of isolates that were accurately separated into human, livestock and wildlife sources. Analyses conducted by the Laboratory for Soil Microbiology, Crop and Soil Environmental Sciences Department, Virginia Tech, on samples collected in the Blackwater River Watershed by MapTech, and used in this study, showed levels of correct separation of known source isolates ranging from 85 to 95 percent, demonstrating a high degree of confidence in the analyses.

## SOURCE IDENTIFICATION

Given the nature of the TMDL development process as it pertains to fecal coliform contamination, identification of sources and delivery mechanisms is a particularly critical step. As can be seen in Table 1, source identification and approaches to modeling sources vary widely among the TMDLs that have been developed to date for fecal coliform contamination. The number of nonpoint sources identified ranges from 5 in the Cottonwood Creek study (USEPA, 2000a) to 18 in the Blackwater River studies (VADEQ/VADCR, 2000c-h). Additionally, the delivery mechanisms modeled for any particular source vary from study to study. In the case of private residential sewage treatment (e.g. septic systems), some studies modeled only one mechanism for delivery to surface waters (i.e. direct loading to the stream from failed septic systems), while others recognized multiple transport mechanisms (e.g. direct loading as well as indirect loading to the stream through fecal coliform build-up on land above failed septic systems and wash-off to the stream during runoff events). There were also various methods of calculating septic failure rates, with some using a straight percentage while others considered age of systems and/or proximity to the stream.

No effort is made in this paper to promote one methodology over another, as specific methods can and should vary from one situation to another and professional judgment does play a role in the modeling process. However, it is clear from Table 1 that the identification of sources and delivery mechanisms is directly related to the load reduction allocations determined. For instance, the only studies that identified a need to reduce loadings from direct deposition of wildlife in streams were those that identified direct deposition by wildlife sources as a potential source and delivery mechanism. This is not to imply that any of the TMDLs developed to date are inaccurate, but that the model is sensitive to this input. This relationship illustrates the need to accurately and comprehensively identify sources in any TMDL development, but particularly in development of fecal coliform TMDLs. Bacterial Source Tracking is an invaluable tool for validating the presence or absence of sources and potentially for calibrating models used to determine load allocations.

## BST APPLICATION TO TMDL MODELING

In using BST as a tool to improve source identification, MapTech has identified three levels of refinement, based on the amount of BST data available. First, BST data has been used for determining the presence or absence of sources in a drainage area. Second, with additional BST data, water quality models, calibrated for total fecal coliform, can be adjusted to more accurately reflect the measured source loads, without altering the original calibration. And third, if sufficient BST data is available, the model can be calibrated based on measured concentrations of the fecal indicator originating from each source.

Using BST Data to Confirm the Presence or Absence of Sources

The most basic application of BST data to the TMDL modeling process is identification of the presence or absence of sources. This level of use has been practiced by both Gene Yagow, Research Scientist in the Biological Systems Engineering Department at Virginia Tech, in developing a TMDL for Mountain Run in Virginia, and by MapTech in developing six TMDLs in the Blackwater River Watershed in Virginia. Yagow collected water samples that were subsequently analyzed by the Virginia Tech Biology Department using a DNA fingerprinting technique to identify the presence of *E.coli* from specific species in each of the subwatersheds feeding Mountain Run. Species identified included: human, cow, horse, dog, deer, otter, muskrat, raccoon, goose, and duck. No attempt was made to quantify the relative contribution from these sources. MapTech collected water samples, which were processed by the Laboratory for Soil Microbiology in the Crop and Soil Environmental Science Department at Virginia Tech under the supervision of Professor Charles Hagedorn. Hagedorn used the antibiotic resistance technique described earlier to analyze the water samples and establish the percentage of fecal streptococci isolates that originated from wildlife, livestock, and human sources (Figure 1). MapTech used this data to confirm the presence of all three sources in the watershed. Additionally, MapTech was able to validate the modeling procedures used based on the load reductions necessary to bring the impairment within the state standard. Specifically, as can be seen in Figure 1, the predominant sources in the watershed were livestock and wildlife with a less prominent but significant contribution from human sources. The load reductions reflected this relationship, requiring an elimination of straight pipes and direct deposition by livestock, as well as varying reductions in direct deposition by wildlife. Anthropogenic sources (i.e. straight pipes and direct deposition by livestock) were addressed first, however, violations of the standard continued to occur during low flow conditions, indicating a need to reduce the load from direct deposition by wildlife. The amount of BST data available for this process was limited to an average of six samples per impairment, sampled from two different stations in each impairment. At any given station 3 to 4 samples were collected, separated in time by 1 to 2 months. All of the samples were taken during low flow conditions. Because of these circumstances, the collected data neither reflect seasonal fluctuations nor the full range of hydrologic conditions in the watershed. As such, further analysis was not deemed feasible for the Blackwater River TMDL development process.

Using BST Data to Adjust Loads in a Traditionally Calibrated Model

Since the development of the Blackwater River TMDLs, MapTech has explored various methods of using BST data to improve model calibration. As explained earlier, the BST data available in the Blackwater River Watershed was insufficient to use for a thorough calibration. However, for this study, the BST data was used to demonstrate a technique for improving the initial calibration that was performed using standard fecal coliform data. The model used in developing the Blackwater River TMDLs was the United States Geological Survey (USGS) model, Hydrologic

Simulation Program - Fortran (HSPF). The HSPF model is a continuous simulation model that can incorporate nonpoint (NPS) pollutants in runoff, as well as pollutants entering from point sources. The three main parameters available for adjusting the mathematical approximation of fate and transport of land-based pollutants are the accumulation rate, the maximum accumulation, and the runoff rate necessary to remove 90% of the pollutant. These parameters account for all non-hydrologic environmental factors leading up to the delivery of land-based pollutants to the water body. In modeling fecal coliform, the maximum accumulation is typically viewed as a "die-off" parameter and set at some multiple of the accumulation rate. The accumulation rate and maximum accumulation can be varied monthly over a year. Direct NPS loads to the water body (e.g. direct deposition in a stream by livestock) can be modeled as point sources, with the rate variable at any multiple of the time-step chosen for the model. One additional parameter is available for modeling the pollutant, a coefficient for defining the exponential decay (die-off) of the pollutant in the water. In order to refine the calibration using BST data, the model must be run as though there were three different pollutants being addressed (i.e. human fecal coliform, livestock fecal coliform, and wildlife fecal coliform). To illustrate this process, the Blackwater River model was run with fecal coliforms separated out as three different pollutants (Figure 2).

Next the results of the modeling were converted to display the percentage of each source represented in the total fecal coliform concentration and plotted against the total fecal coliform concentration (Figure 3). This figure can be graphically compared to monitored results collected in the upper Blackwater River impairments. Although calibration and validation based on standard total fecal coliform data showed a close match of modeled to monitored data, it appears that the livestock source has been generally overestimated and the wildlife source has been generally underestimated. It should be restated here that the monitored data was collected at four points in time, at multiple locations in the watershed, and during low flow conditions. This being said, an attempt was made to improve the calibration so that modeled output would reflect the distribution of fecal sources monitored. Since all of the monitored data was collected during low flow conditions, improvement of the calibration focused on adjusting the distribution of loadings being directly deposited to the stream. Specifically, loadings that were reduced from the livestock direct inputs were transferred to the wildlife direct inputs, so that the total direct load to the stream at any point in time did not change, and consequently the original calibration remained valid. Results of this adjustment are shown in Figures 4 and 5. The adjusted model produces output that is more in agreement with the monitored data. Given monitored data that represented concentrations during runoff events, a similar adjustment could be made to land based loads. To determine necessary load reductions for the TMDL, the model would be run again with the adjusted loads, but with fecal coliform modeled as a single pollutant. The adjusted model would be no different from the original in terms of matching monitored levels of fecal coliform, however the resulting allocations may be different due to the changes made in the sources of the load.

Using BST Data to Calibrate the HSPF Model

A third level of analysis could be pursued if considerably more data were available. If enough BST data were available to reflect seasonal fluctuations and the full range of hydrologic conditions, then the water quality model being used could be calibrated based on the contribution of fecal coliform from individual sources. In calibrating the model to individual sources of fecal coliform, the model would be run with fecal coliform represented by three pollutants as described above. The three pollutants would each be calibrated based on measured values of fecal coliform concentration and percentage of isolates identified from that particular source. For instance, if a particular observation showed a fecal coliform concentration of 100 cfu/100 ml, and 50% isolates classified as originating from wildlife, then, based on that observation, 50 cfu/100 ml would be used as a data point in calibrating the wildlife fecal coliform concentrations. Finally, the results of the calibrated model would be summed across pollutant sources to determine the total fecal coliform concentrations. This is a fundamentally different approach to modeling fecal coliforms using HSPF as compared to the methods described thus far. With the method described above for improving calibration, fecal coliforms were modeled as three separate pollutants in order to determine load adjustments, however any TMDL allocations would be made based on running the model with fecal coliforms represented as a single pollutant, as compared to summing results of modeling three different pollutants.

Since the model does not use strictly linear relationships, dividing the pollutant in this way and summing the resulting output to determine the total fecal coliform concentrations does not yield the same results as modeling all fecal coliforms together as one pollutant, assuming that no parameters are changed except those related to loadings. Figure 6 shows residuals based on modeling using these two techniques and calculated by subtracting concentrations determined by treating fecal coliforms as a single pollutant from the sum of concentrations determined by treating fecal coliforms as three separate pollutants. Occasionally a large negative value is displayed in the plot of these residual values. These large negative values can be explained based on the relationship between the maximum accumulation parameter and the accumulation rate. As described earlier the maximum accumulation parameter is typically set as a multiple of the accumulation rate, and was set accordingly in this case. Suppose there is a dry period with no runoff, eventually the pool of fecal coliform available for runoff will reach the maximum accumulation and stabilize. Now assume that the accumulation rate for fecal coliforms from livestock increases because of herd management changes (e.g. animals spending more time in pasture and less in confinement). If fecal coliforms are being modeled as a single pollutant, then this increase is treated as an increase in total fecal coliform accumulation. The maximum accumulation is increased accordingly, and all sources of fecal coliforms contribute to the pool of fecal coliforms available for runoff until the new maximum accumulation rate is achieved. On the other hand, suppose that fecal coliforms are being modeled as three separate pollutants. If the accumulation rate for livestock fecal coliform is increased, and the associated maximum accumulation parameter is increased accordingly, only loadings of fecal coliform from livestock will increase the pool of fecal coliform available for runoff, since the human and wildlife pools remain at their limit. Because of this relationship, it will take longer for the fecal coliform pool to re-stabilize at its maximum value if fecal coliforms are modeled as 3 separate pollutants. If there is a runoff producing event during the time frame when these pools are stabilizing, there will be more fecal coliforms available from the model where fecal coliform was treated as a single pollutant and a negative residual will result. To further complicate matters, runoff-producing events will further delay the stabilization process creating more opportunities for negative residuals.

In spite of this difference between these modeling techniques, arguments can be made for modeling fecal coliforms as three separate pollutants. First, it makes "physical" sense that, if the maximum accumulation of fecal coliforms from a given source is increased because the accumulation rate from that source has increased, then only that source should contribute to increasing the overall fecal coliform pool. Second, disregarding any physical basis for the model parameters, the available parameters should be adequate to get an acceptable fit of modeled to monitored data in either scenario. And third, the benefit of being able to validate the representation of sources in the model outweighs any slight deviation in modeling a pollutant that demonstrates a high degree of variability such as fecal coliform.

Sufficient BST data for calibration of the Blackwater River model using this method were not available. As to the amount of data that is sufficient, this is largely a matter of professional judgment, however, it is clear that the three to four data points available at any given station in the Blackwater River Watershed were not adequate to perform any meaningful calibration. However, MapTech is continuing to monitor water quality in the Blackwater River Watershed as part of the TMDL implementation plan development and will be reviewing the potential for calibration using the described method once sufficient data is available.

## CONCLUSIONS AND RECOMMENDATIONS

- Objective methods for determining fecal coliform sources should be incorporated into the TMDL development process.
- Bacterial Source Tracking (BST) using antibiotic resistance analysis is a viable tool for verifying the presence or absence of fecal sources.
- Bacterial Source Tracking data can be used to improve the calibration of the HSPF model without altering previous calibration results.
- Fecal coliforms can be modeled as three different pollutants (i.e. human fecal coliform, livestock fecal coliform, and wildlife fecal coliform) using HSPF in order to calibrate using BST data.
- Calibration of water quality models using BST data should be pursued, given sufficient data to represent seasonality and both wet and dry climatic conditions.

## ACKNOWLEDGEMENTS

MapTech, Inc. of Blacksburg, Virginia expresses their sincere appreciation to: Virginia Department of Conservation and Recreation who supported the water quality monitoring performed in the Blackwater River Watershed through funding (Contract C199:99-592); Amy Bowman, Graduate Research Assistant, in the Laboratory for Soil Microbiology, Department of Crop and Soil Environmental Science, Virginia Tech, who performed bacteria source tracking analyses on samples collected in the Blackwater River Watershed; and Land owners who provided access through their property.

# REFERENCES


USEPA. 1999. Fecal Coliform TMDL Development for Muddy Creek, Virginia. Prepared by The Muddy Creek TMDL Establishment Workgroup for U.S. Environmental Protection Agency, Region III, 841 Chestnut Street, Philadelphia, PA .

USEPA. 2000a. Fecal Coliform TMDL Modeling Report Cottonwood Creek Watershed Idaho County, Idaho. EPA Office of Water, Office of Science and Technology, Standards and Applied Science Division, Exposure Assessment Branch, Washington D.C.

USEPA. 2000b. Proposed Total Maximum Daily Load for Fecal Coliform in Cash Hollow Creek, Watauga River Watershed, Tennessee - DRAFT. Prepared by Tennessee Department of Environment and Conservation for U.S. Environmental Protection Agency, Region IV, Atlanta Federal Building, 61 Forsyth Street SW, Atlanta, GA.

VADEQ/VADCR. 2000a. DRAFT Fecal Coliform TMDL Mountain Run Watershed Culpeper County, Virginia. Prepared by Gene Yagow, Department of Biological Systems Engineering, Virginia Tech for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000b. Fecal Coliform TMDL Development for Cedar, Hall, Byers, and Hutton Creeks, Virginia. Prepared by CH2M Hill in association with Aqua Terra Consultants for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000c. Fecal Coliform TMDL Development for Lower Blackwater River, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000d. Fecal Coliform TMDL Development for Maggodee Creek, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000e. Fecal Coliform TMDL Development for Middle Blackwater River, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000f. Fecal Coliform TMDL Development for North Fork of the Blackwater River, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000g. Fecal Coliform TMDL Development for South Fork of the Blackwater River, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000h. Fecal Coliform TMDL Development for Upper Blackwater River, Virginia. Prepared by MapTech Inc., Blacksburg, VA for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000i. Fecal Coliform TMDL Plan for Big Otter River including Sheep Creek, Elk Creek, Machine Creek, Little Otter River and Lower Big Otter River, Bedford County, Virginia. Prepared by Virginia Tech Department of Biological Systems Engineering and Department of Biology for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000j. Fecal Coliform TMDL Plan for Lower Dry River, Rockingham County, Virginia. Prepared by Virginia Tech Department of Biological Systems Engineering and Department of Biology for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000k. Fecal Coliform TMDL Plan for Mill Creek, Rockingham County, Virginia. Prepared by Virginia Tech Department of Biological Systems Engineering and Department of Biology for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

VADEQ/VADCR. 2000l. Fecal Coliform TMDL Plan for Pleasant Run, Rockingham County, Virginia. Prepared by Virginia Tech Department of Biological Systems Engineering and Department of Biology for Virginia Department of Environmental Quality and Virginia Department of Conservation and Recreation, Richmond, VA.

WVDEP/USEPA. 1997. Fecal Coliform TMDL Development for South Branch Potomac [River] including Lunice Creek, Mill Creek, and North Fork, West Virginia. State of West Virginia, Division of Environmental Protection 1201 Greenbrier St., Charleston, WV and U.S. Environmental Protection Agency, Region III 841 Chestnut Street, Philadelphia, PA .

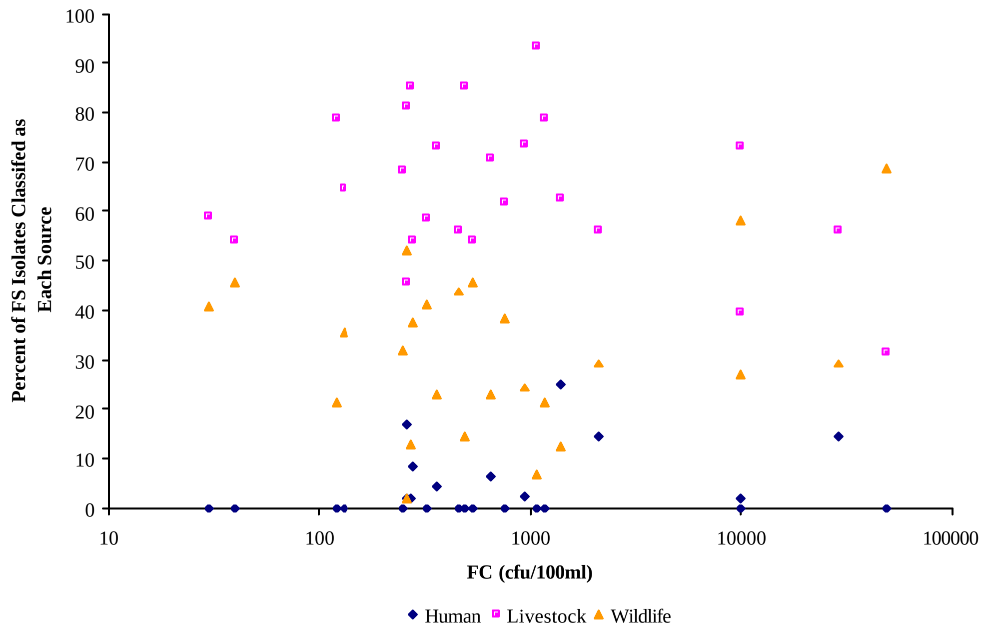



**Figure 1.** Results of MapTech's in-stream monitoring for fecal coliform concentrations and fecal sources. Each sample is represented by three points representing the percentage of fecal streptococci (FS) isolates in that sample classified as originating from each source.

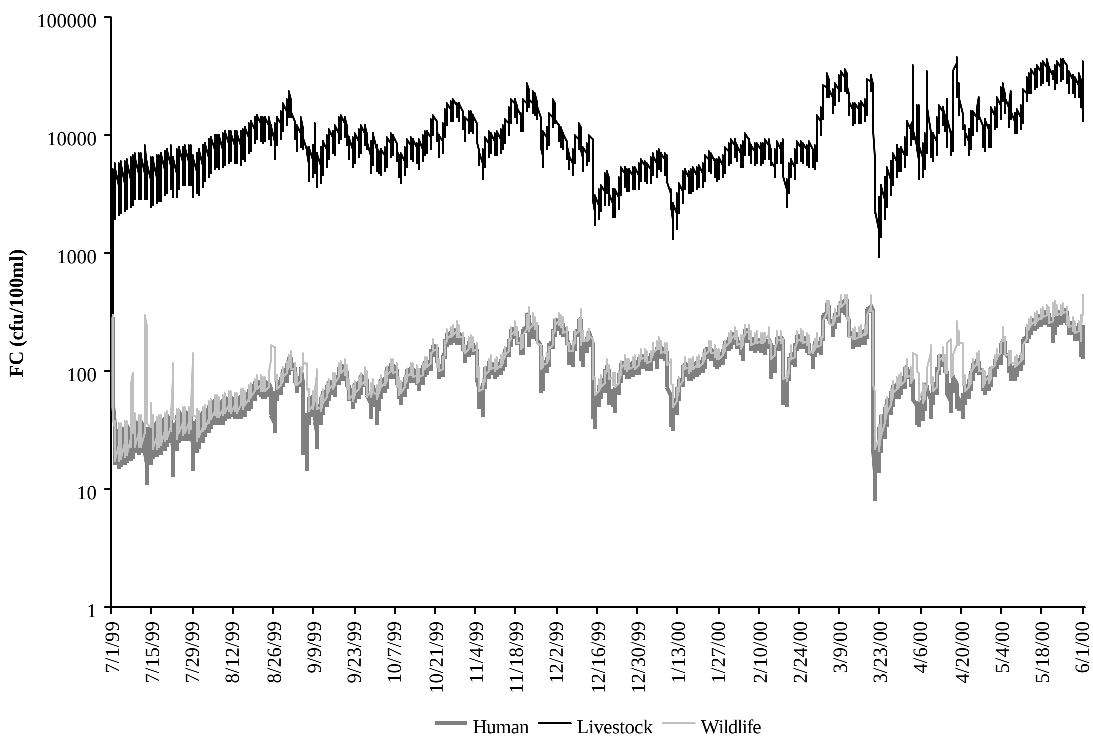



**Figure 2.** Results of modeling fecal coliform (FC) concentrations as three different pollutants (i.e. human FC, livestock FC, and wildlife FC) at the outlet of the Upper Blackwater River Watershed.

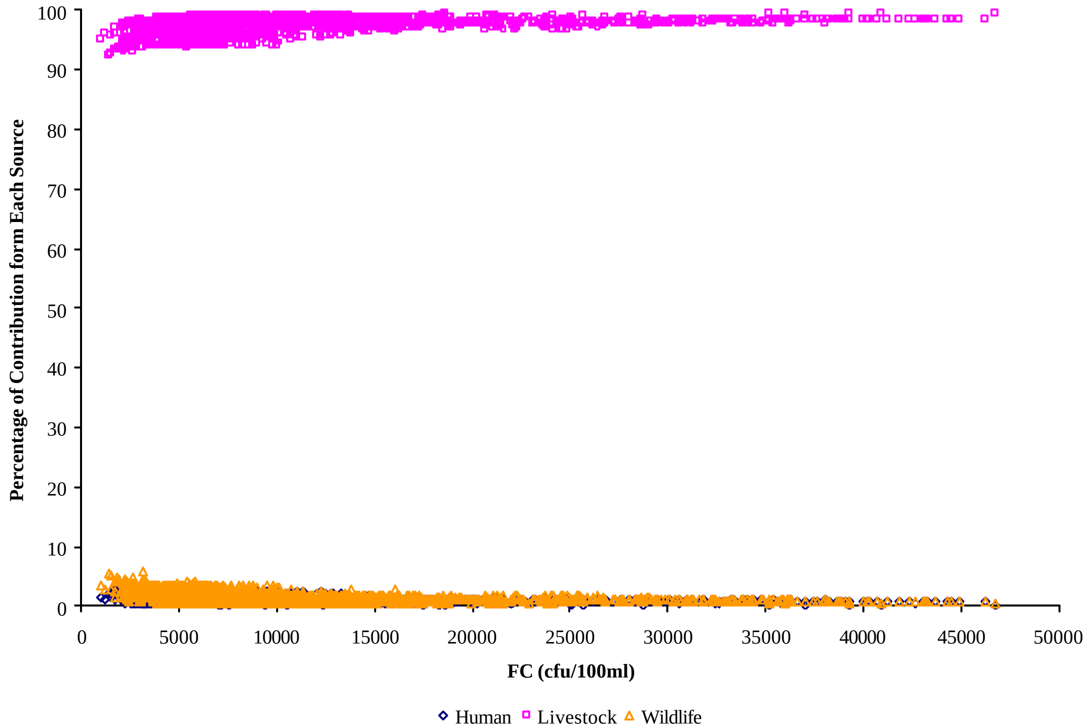



**Figure 3.** Percentages of modeled fecal coliform concentrations at the outlet of the Upper Blackwater River Watershed originating from each source (i.e. human, livestock, and wildlife) before refinement of calibration using BST data.

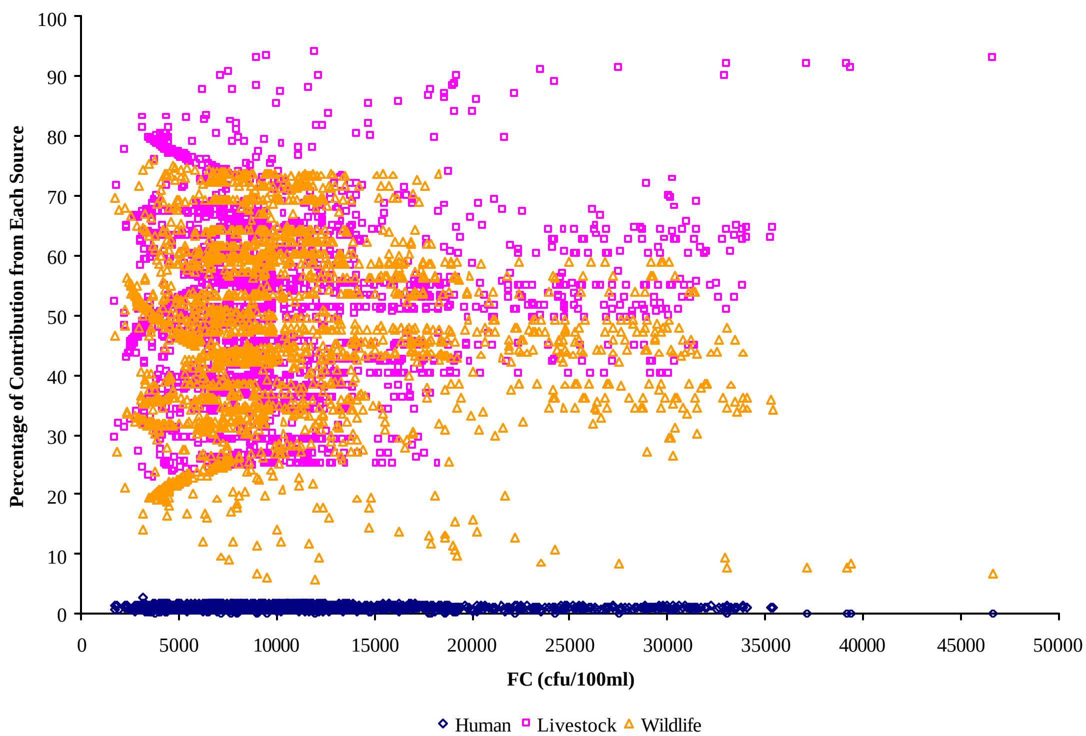



**Figure 4.** Percentages of modeled fecal coliform concentrations at the outlet of the Upper Blackwater River Watershed originating from each source (i.e. human, livestock, and wildlife), after improvement of calibration using BST data.

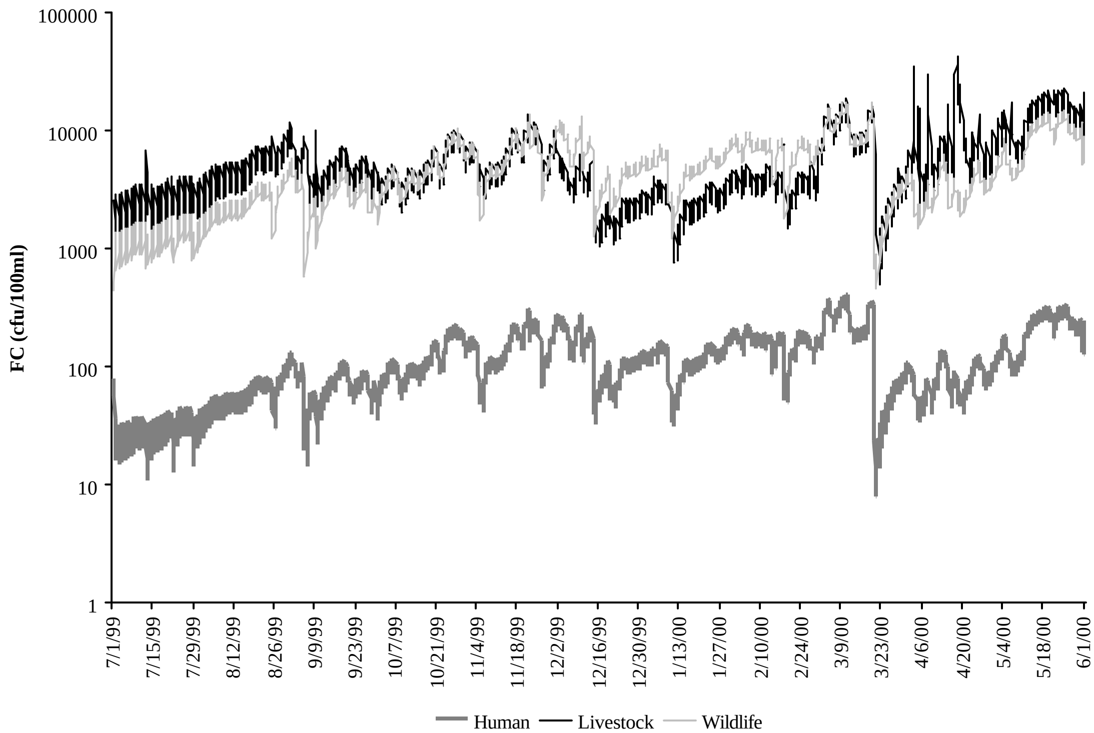



**Figure 5.** Modeled fecal coliform (FC) concentrations from three sources (i.e. human FC, livestock FC, and wildlife FC) at the outlet of the Upper Blackwater River Watershed, after adjustment.

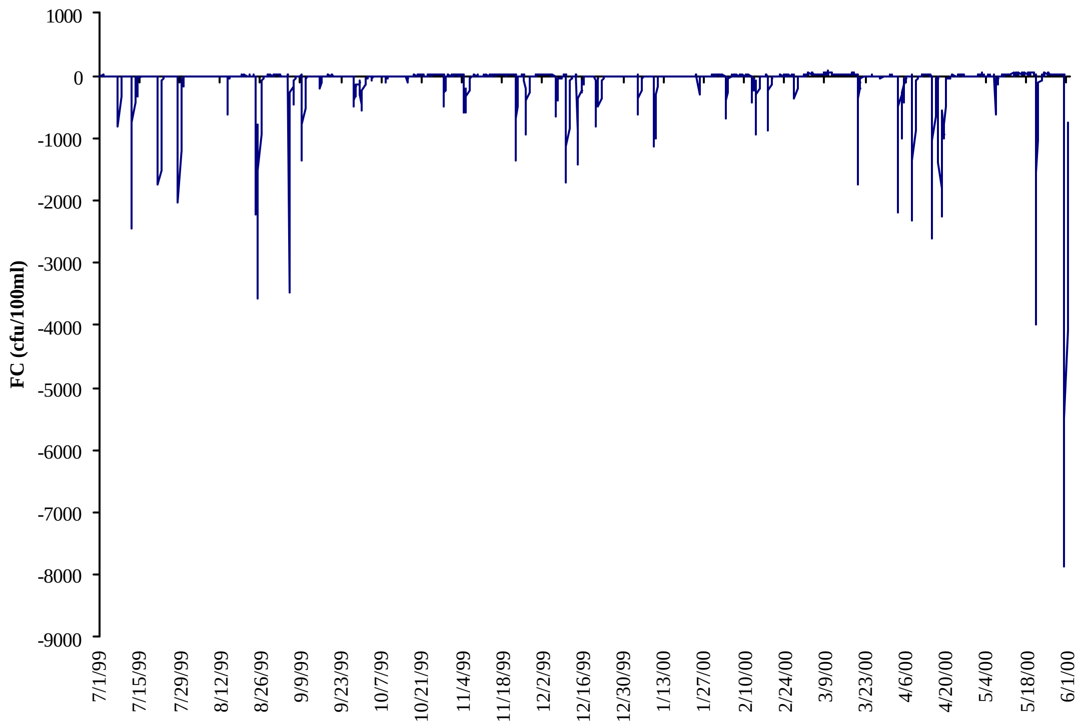



**Figure 6.** Residuals of total fecal coliform (FC) concentrations modeled as one pollutant and summed after modeling as three pollutants (i.e. human FC, livestock FC, and wildlife FC).

*Residual = Single Pollutant FC Concentrations - Sum of Three FC Pollutant Concentrations*

**Table 1.** Summary of Nonpoint Fecal Coliform Sources Identified and Associated Load Reductions from Fecal Coliform TMDLs Developed to Date. (Part 1 of 3)

| Impairment Area | Nonpoint Fecal Sources Identified | | NPS Load Reductions | | Reference |
|---|---|---|---|---|---|
| South Branch Potomac River, WV | Human | | | | WVDEP/ USEPA, 1997 |
| | Septic Failure | (Direct) | | | |
| | Livestock | | | | |
| | Poultry | (Indirect) | 30-50% | Ag. & Pasture Land | |
| | Cattle | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Indirect) | | | |
| | Geese | (Indirect) | | | |
| | Ducks | (Indirect) | | | |
| Muddy Creek, VA | Human | | | | USEPA, 1999 |
| | Septic Failure | (Direct) | 100% | Septic Failure | |
| | Straight Pipes | (Direct) | 100% | Straight Pipes | |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 99.3% | Cattle - Direct | |
| | Dairy | (Direct & Indirect) | 57.4% | Ag. & Pasture Land | |
| | Poultry | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Indirect) | | | |
| | Background | (Direct) | | | |
| Mountain Run, VA | Human | | | | VADEQ/ |
| | Septic Failure | (Direct & Indirect) | | | VADCR, 2000a |
| | Straight Pipes | (Direct & Indirect) | 100% | Straight Pipes | |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 90-95% | Cattle - Direct | |
| | Dairy | (Direct & Indirect) | | | |
| | Horse | (Direct & Indirect) | | | |
| | Swine | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Indirect) | | | |
| | Muskrat | (Indirect) | | | |
| | Raccoon | (Indirect) | | | |
| | Geese | (Indirect) | | | |
| | Duck | (Indirect) | | | |
| | Pet | | 0-97% | Urban (Pet) | |
| | Dog Equivalent | (Indirect) | | | |
| Middle Fork Holston River, VA | Human | | | | VADEQ/ |
| | Septic Failure | (Direct) | 98-100% | Septic Failure | VADCR, 2000b |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 98-100% | Cattle - Direct | |
| | Dairy | (Direct & Indirect) | 0-10% | Pasture/Hayfield | |
| | Horse | (Indirect) | | | |
| | Swine | (Indirect) | | | |
| | Sheep | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Indirect) | | | |
| | Canada Geese | (Indirect) | | | |
| | Pet | | | | |
| | Dog | (Indirect) | | | |

**Table 1.** Summary of Nonpoint Fecal Coliform Sources Identified and Associated Load Reductions from Fecal Coliform TMDLs Developed to Date. (Part 2 of 3)

| Impairment Area | Nonpoint Fecal Sources Identified | | NPS Load Reductions | | Reference |
|---|---|---|---|---|---|
| Cottonwood Creek, ID | Human | | | | USEPA, 2000a |
| | Septic Failure | (Direct) | 80-90% | Septic Failure | |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 80-100% | Cattle - Direct | |
| | Dairy | (Indirect) | | | |
| | Swine | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Indirect) | | | |
| | Elk | (Indirect) | | | |
| Blackwater River, VA | Human | | | | VADEQ/ VADCR, 2000c |
| | Septic within 50' of stream | (Direct) | | | |
| | Septic Failure | (Indirect) | | | VADEQ/ VADCR, 2000d |
| | Straight Pipes | (Direct) | 100% | Straight Pipes | |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 89-100% | Livestock - Direct | VADEQ/ VADCR, 2000e |
| | Dairy | (Direct & Indirect) | | | |
| | Horse | (Indirect) | | | |
| | Donkey | (Indirect) | | | VADEQ/ VADCR, 2000f |
| | Sheep | (Indirect) | | | |
| | Goat | (Indirect) | | | |
| | Wildlife | | 0-85% | Wildlife - Direct | VADEQ/ VADCR, 2000g |
| | Deer | (Direct & Indirect) | | | |
| | Raccoon | (Direct & Indirect) | | | |
| | Muskrat | (Direct & Indirect) | | | VADEQ/ VADCR, 2000h |
| | Beaver | (Direct & Indirect) | | | |
| | Turkey | (Direct & Indirect) | | | |
| | Geese | (Direct & Indirect) | | | |
| | Duck | (Direct & Indirect) | | | |
| | Other | (Direct & Indirect) | | | |
| | Pet | | | | |
| | Dog | (Indirect) | | | |
| | Cat | (Indirect) | | | |
| North River, VA | Human | | | | VADEQ/ VADCR, 2000j |
| | Septic Failure | (Indirect) | 0-25% | Indirect Sources | |
| | Straight Pipes | (Direct) | | | |
| | Livestock | | | | VADEQ/ VADCR, 2000k |
| | Milking parlor wash-off | (Direct) | 0-100% | Milking parlor wash-off | |
| | Beef | (Direct & Indirect) | 84-100% | Cattle - Direct | VADEQ/ VADCR, 2000l |
| | Dairy | (Direct & Indirect) | | | |
| | Poultry | (Indirect) | | | |
| | Swine | (Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Direct & Indirect) | 0-60% | Wildlife - Direct | |
| | Raccoon | (Direct & Indirect) | | | |
| | Muskrat | (Direct & Indirect) | | | |
| | Geese | (Direct & Indirect) | | | |
| | Duck | (Direct & Indirect) | | | |
| | Pet | | | | |
| | Generic | (Indirect) | | | |

**Table 1.** Summary of Nonpoint Fecal Coliform Sources Identified and Associated Load Reductions from Fecal Coliform TMDLs Developed to Date. (Part 3 of 3)

| Impairment Area | Nonpoint Fecal Sources Identified | | NPS Load Reductions | | Reference |
|---|---|---|---|---|---|
| Big Otter River, VA | Human | | | | VADEQ/ |
| | Septic Failure | (Indirect) | 50-60% | Indirect Sources | VADCR, 2000i |
| | Straight Pipes | (Direct) | 0-100% | Straight Pipes | |
| | Livestock | | | | |
| | Beef | (Direct & Indirect) | 97-100% | Livestock - Direct | |
| | Dairy | (Direct & Indirect) | | | |
| | Wildlife | | | | |
| | Deer | (Direct & Indirect) | 50-80% | Wildlife - Direct | |
| | Raccoon | (Direct & Indirect) | | | |
| | Muskrat | (Direct & Indirect) | | | |
| | Geese | (Direct & Indirect) | | | |
| | Duck | (Direct & Indirect) | | | |
| | Pet | | | | |
| | Generic | (Indirect) | | | |
| Cash Hollow Creek, | Human | | | | USEPA, 2000b |
| TN | Septic Failure | (Direct) | 50-90% | Septic Failure | |
| | Urban Runoff | (Indirect) | 90% | Urban Runoff | |
| | Urban Unknown | (Direct) | | | |
| | Livestock | | 95.1-98.4% | Other Direct | |
| | Beef | (Direct & Indirect) | | Sources | |
| | Dairy | (Direct & Indirect) | | (Urban & | |
| | Swine | (Indirect) | | Livestock) | |
| | Wildlife | | | | |
| | Deer Equivalent | (Indirect) | | | |

# Stream Temperature Model Input Parameters: How Close is Close Enough?

C.W. Krause and T.J. Newcomb
Department of Fisheries and Wildlife Sciences
Virginia Tech
149 Cheatham Hall
Blacksburg, VA 24061
cokrause@vt.edu

## ABSTRACT

Stream temperature prediction models can be used to predict thermal regimes under the influence of hydrologic changes, watershed land-use changes, and/or remediation actions to assist management of aquatic biota. The input-parameters, air temperature and relative humidity, typically have the largest influence on model output and they can be collected at the stream with data loggers or from a weather station. Whether the increased time and costs of onsite weather data collection are warranted in terms of model predictive ability is not known. We used the Stream Segment Temperature model (SSTEMP) to evaluate the effect of weather data input parameters on model predictions. Parameters were collected at increasing distance and elevation difference from Back Creek in Roanoke County, Virginia. We found that the weather data was statistically different between that measured onsite and that obtained from four Virginia weather stations for January and July 2000. This difference did not result in statistically different model predictions. Using the sensitivity analysis tool in the SSTEMP model the most sensitive parameters during July were air temperature and relative humidity, and during January was temperature of lateral inflow. Therefore, weather data collection at the stream will be more influential during July if weather station conditions differ from those at the stream. However, our research indicates that such differences would have to be larger than we observed to result in unreliable model predictions.

**Keywords:** SSTEMP, stream temperature modeling, predictive ability, thermal water quality

## INTRODUCTION

Anthropogenic activities such as logging, agriculture, hydropower generation, and urbanization can alter flow rates, stream channel shape, or reduce streamside vegetation thereby changing the temperature regime of a stream (Bartholow 1989, Brown 1980, Hewlett and Fortson 1982, Jensen 1987, LeBlanc et al. 1997, Webb and Walling 1993). This change may have a negative effect on fish and other aquatic biota because water temperature strongly influences survival, growth and reproduction (Armour 1991). Predicting the effect of such hydrological and watershed land-use changes, or remediation actions prior to their implementation, is possible with stream temperature prediction models. Information on the use of such models and best parameter collection methodologies are not well documented. One widely used temperature model is the Stream Segment Temperature Model (SSTEMP) developed by the U.S. Fish and Wildlife Service and available through the United States Geological Survey (USGS).

The SSTEMP model is a simplified version of the Stream Network Model (SNTEMP) where the primary difference is that the SSTEMP model only predicts for a single stream segment and single time period (e.g. day, week, month) per run, while SNTEMP allows predictions for a stream network over multiple days. Because of these differences SSTEMP is better suited for scoping exercises where the time period and length of reaches to be modeled are small. Primary assumptions of the SSTEMP model are (1) that water in the system is instantaneously and completely mixed, (2) all geometry, meteorological, and hydrological parameters are 24-hour means, (3) the representative time period is long enough to allow water to flow the full length of the segment, and (4) flow is essentially steady over the averaging period (Bartholow 2000). Because SSTEMP is a mechanistic one-dimensional heat transport model, its water temperature predictions are based on an energy budget. The energy budget is comprised of factors that either add or remove energy (i.e., heat) from the water in a stream. The primary factors used by the model are direct solar radiation (short-wave), radiation with the sky (long-wave), vegetative and topographic shading, convection with the air, evaporation, conduction with the streambed, and advection from incoming water sources (Bartholow 1989, Bartholow 2000, Sullivan et al. 1990). The model uses these factors to assess the net change in stored energy to a parcel of water (where the upstream temperature and volume of the parcel is known) as it progresses downstream experiencing different heat fluxes. The model predicts the energy budget factors in order to ultimately predict mean water temperature using 26 input parameters that describe the average stream geometry, hydrology, meteorology, stream shading, and time period. The input parameters are collected in the field, obtained from available sources, or estimated from recommended values as described in the SSTEMP model user-manual (Bartholow 1989, Bartholow 2000).

The majority of the parameters when assessed individually are insensitive, thereby having little effect on model predictions. According to the literature the SSTEMP model is most sensitive to air temperature and relative humidity, henceforth humidity, parameters (Bartholow 1989, Bartholow 2000, Sullivan et al. 1990). These parameters can be collected continuously onsite (i.e., at the stream) using battery powered data loggers or obtained offsite (i.e., from a weather station). If parameters are collected at a location other than the stream, the quality and relevance of the data may vary due to the distance of the weather station from the stream, elevation differences, topographical differences, and the posible presence of thermal inversions. For example, the recorded values at a weather station in an urban area may differ from those recorded at a stream surrounded by forested hills. However, whether these differences translate to differences in predictive ability is uncertain. Furthermore, it is uncertain whether these differences would warrant collection of weather data onsite which may be time consuming and costly. Information assessing the trade-off between predictive ability of the model and location of weather data collection would allow aquatic resource managers to determine which option is better.

In order to address these concerns, the objectives of this study were to (1) evaluate differences between weather data measured onsite and obtained offsite 15, 175, and 240 km from the stream, (2) evaluate differences between stream temperature predictions when using onsite versus offsite weather data, and (3) determine the sensitive parameters during January and July 2000 using the automated first-order sensitivity analysis tool in the SSTEMP model.

## METHODS

We used a third order tributary to model stream temperature and collect The study site for onsite weather data collection, water temperature collection, and water temperature predictions, Back Creek, is located in southern Roanoke County, Virginia and is in the Roanoke River drainage basin in southwestern Virginia. Back Creek is 42 km in length, however we limited our study reach to 38.03 km to avoid backwater conditions near the Roanoke River confluence. Back Creek is 9-15 m wide near the mouth and 3-6 m wide in the upper reaches. For 1999, annual mean flow was 0.75 $m^3$/sec. The mean water temperature recorded in the lower portion of Back Creek during January 2000 was 2.57?C and during July 2000 was 23.67?C.

The SSTEMP model was used to evaluate model predictive ability when using weather data, consisting of air temperature and humidity collected onsite versus offsite. This assessment was conducted for one winter month (January 2000) and one summer month (July 2000). Daily mean air temperature and humidity were initially used in the model for January and July. We additionally used minimum air temperature and daily mean humidity for modeling stream temperature in January. The SSTEMP model was chosen for its straightforward, easy to use format, Windows? based operation, likelihood of use by aquatic resource managers, and free availability (available at ww.mesc.usgs.gov/rsm/rsm_download.html #TEMP and model use information at www.mesc.usgs.gov/training/if312.html).

Of the twenty-six input parameters, the majority can be obtained from available sources or recommended values listed in the user-manual (Bartholow 2000), but some must be measured at the study stream. The following are obtainable andrecommended parameters: Flow data from the USGS gage

02056650 (obtained at http://waterdata.usgs.gov) on Back Creek was used to calculate 24 hr mean segment outflow values. We used DeLorme? 3-D TopoQuad software to determine latitude, segment length, segment azimuth, and upstream and downstream elevations. As recommended in the user-manual, we used mean annual air temperature for the inflow, accretion, and ground temperature parameters, as well as a segment inflow value of 0 $m^3$/sec since Back Creek originates at a headwater (Bartholow 2000). Recommended values were also used for manning's n (0.035), thermal gradient (1.65), dust coefficient (9.5 in winter, 6.5 in summer), and ground reflectivity (25.0 for grass covered ground) (Bartholow 2000). We obtained hourly air temperature and humidity values for 3 offsite locations, Roanoke Regional Airport (ROA) (15 km away, 350 m elevation), Charlottesville Albemarle Airport (CHO) (175 km away, 195 m elevation), and Richmond International Airport (RIC) (240km away, 50 m elevation), from the National Climactic Data Center (NCDC) (www.ncdc.noaa.gov) for the months of January and July 2000. Hourly wind speed and percent possible sun values were obtained from NCDC for the closest weather observation station to Back Creek, which is ROA.

The following parameters were measured at Back Creek: Flow data combined with measured stream widths from a field survey conducted in July 2000 allowed approximate values to be determined for the A coefficient and B exponent parameters used in the equation, width=A?flow$^B$, by the model to estimate stream width at varying flows. Water temperatures for comparison against model predictions were measured hourly with an Onset? StowAway XTI temperature logger (measurable range -4°C to +37°C) near the end of the modeled segment in Back Creek. Air temperature and humidity were measured hourly in the riparian zone of Back Creek (340 m elevation) with an Onset? StowAway Tidbit data logger (measures temperatures ranging from -20?C to +50?C with an accuracy of ±0.4°C at +70°C) and an Onset? HOBO H8 Pro data logger (measures humidity ranging from 0% to 100% with an accuracy of ±3%). We measured topographic altitude and vegetation height, crown, offset, and density parameters for both sides of the creek at 168 random locations over 34.6km. Topographic altitude was measured with a clinometer. Vegetation height was calculated by measuring the distance to the tree from the observer's position and multiplying it by the tangent of the angle from the water surface to the top of the vegetation measured with a clinometer. We measured vegetation crown and vegetation offset via visual estimation and periodically verified estimations with a measuring tape. Vegetation density was measured by using a light meter to measure light intensity in foot-candles reflected off a standardized surface (an 18% gray card was used) both in the shade and in the sun, and a percentage calculated. The average vegetation density values measured in July were 44.5% for the West side and 46.5% for East side. These values were reduced to typical winter values, 8% for West Side and 10% for East Side, for the month of January to account for leaf-fall (J. Bartholow, Midcontinent Ecological Science Center, Fort Collins, CO. personal communication).

To evaluate the weather data and model predictions, three tests were conducted. First, Wilcoxon's signed rank test was used to test the absolute difference between weather data (air temperature and humidity) measured onsite with loggers and weather data obtained from offsite weather stations (ROA, CHO, and RIC) for statistical difference. The SSTEMP model was then run using air temperature and humidity values from each weather station (onsite, ROA, CHO, and RIC) for days 1 to 31 of January and July 2000. Daily mean air temperature was used during January and July, and daily minimum air temperature was used during January. Daily mean humidity was used for both months. The absolute difference between predictions using onsite measured weather data and water temperature was calculated. This was considered the benchmark to which the absolute difference between predictions using offsite weather data (ROA, CHO, and RIC) and measured water temperature were assessed. Second, the Wilcoxon's rank sum test for two samples was used to test the benchmark absolute differences to the offsite absolute differences for statistical difference. Statistical tests were considered significant at p-value <0.05 and all tests were conducted with SAS. Third, the automated first-order sensitivity analysis tool in the SSTEMP model was then used to determine the sensitive parameters during the months of January and July 2000.

## RESULTS

Wilcoxon's signed rank test resulted in sufficient evidence to conclude that January and July air temperature and humidity measured at ROA, CHO, and RIC differ from those measured for the same period onsite (p-value <.0001) (Table 1). The January mean difference between mean air temperature measured onsite versus ROA was 0.84?C, versus CHO was 1.35?C, and versus RIC was 2.03?C (Table 1). The January mean difference between minimum air temperature measured onsite versus ROA was 1.41?C, versus CHO was 1.76?C, and versus RIC was 2.82?C (Table 1). The January mean difference between humidity measured onsite versus ROA was 7.22%, versus CHO was 9.39%, and versus RIC was 7.46% (Table 1). The July mean difference between air temperature measured onsite versus ROA was 1.39?C, versus CHO was 1.25?C, and versus RIC was 2.23?C (Table 1). The July mean difference between humidity measured onsite versus ROA was 12.87%, versus CHO was 8.70%, and versus RIC was 9.62% (Table 1).

On average January minimum air temperature was coldest onsite and warmest at RIC (Figure 3). January temperatures also fluctuated greatly from day to day; for example, minimum air temperature dropped over 12?C at RIC from January $4^{th}$ to $5^{th}$ and increased over 11?C at onsite and RIC from January $15^{th}$ to $16^{th}$ (Figure 3). On average July mean air temperature was coldest onsite and warmest at RIC (Figure 4). Day to day fluctuations of July mean air temperature never exceeded 5?C for any of the sites (Figure 4).

On average January humidity was highest onsite and lowest at CHO (Figure 5). On average July humidity was highest onsite and lowest at ROA (Figure 6). Day to day fluctuation of humidity was greater during January, up to 51.6% at CHO from the $29^{th}$ to 30th, where as in July the largest fluctuation was only 18.2% at ROA from the $10^{th}$ to $11^{th}$ (Figure 5 and 6).

Wilcoxon's rank sum test resulted in insufficient evidence to conclude that predicted temperature using January or July air temperature and humidity values from ROA, CHO, and RIC differ from temperature predicted using values measured onsite (p-values ranged 0.25 – 0.92) (Table 2). When mean air temperature was used the January mean difference between predicted temperature using onsite weather data versus measured temperature was 3.04?C, versus ROA was 3.09?C, and versus CHO was 2.76?C (Table 2). When minimum air temperature was used the January mean difference between predicted temperature using onsite weather data versus measured temperature was 1.39?C, versus ROA was 1.60?C, versus CHO was 1.38?C, and versus RIC was 1.66?C (Table 2). The July mean difference between predicted temperature using onsite weather data versus measured temperature was 1.30?C, versus ROA was 1.41?C, versus CHO was 1.71?C, and versus RIC was 1.56?C (Table 2).

Water temperature predicted using onsite weather data predicted more days within 2.0?C of the measured temperature for January and July than predictions using weather data from ROA, CHO, or RIC (Table 3). Model predictions improved when using minimum daily air temperature values rather than mean daily air temperature values during the month of January (Table 3). Model predictions followed the trend of the measured water temperature (Figure 1 and 2), and air temperature as well as humidity were correlated at all sites (Figure 3, 4, 5, and 6). The 13.7?C mean annual air temperature calculated from hourly air temperatures measured at Back Creek corresponded well with the 13.6?C mean annual air temperature calculated from 20 years of air temperature data measured at ROA 15 km away from Back Creek.

Using the automated first-order sensitivity analysis tool in the SSTEMP model with onsite weather data the most sensitive parameters during January and July were determined. During January accretion temperature, A coefficient in the equation width=A?flow$^{B}$, air temperature, segment outflow, humidity, and wind speed were the most sensitive parameters (Figure 8). During July air temperature and humidity were the most sensitive parameters (Figure 8).

## DISCUSSION

Air temperature and humidity were statistically different between all weather stations for both months assessed, but model predictions using these parameters were not. Though model predictions were not statistically different there was a noticeable decline in predictive ability to within 1?C of the measured water temperature when July air temperature and humidity values used were from RIC (Table 3). This can be attributed to air temperature being the most sensitive parameter during July (Figure 8) and RIC experiencing overall warmer air temperatures than the other weather stations (Figure 4). These results suggest that there is a threshold for which weather conditions will differ enough between the stream and offsite weather station to begin reducing predictive ability. It is likely that greater the distance and/or elevation change between the stream and offsite weather station, the larger the difference in weather data. It is likely that predictive ability would have been even worse if other parameters had been obtained from an offsite location, for example wind speed and percent possible sun.

During January, declines in predictive ability to within 1?C were not as evident as distance and elevation change from the stream increased (Table 3). For example, ROA and RIC (15 km and 240km away respectively) predicted equally poorly to within 1?C, while CHO (intermediate in distance at 175 km away) predicted well to within 1?C (Table 3). The maximum difference between model predictions and measured water temperatures were higher than for July (Table 2). Also, January mean differences were slightly higher than July's for all weather stations except CHO (Table 2). This pattern may be due to larger fluctuations in air temperature and humidity for January as compared to July (Figures 3, 4, 5, and 6).

Humidity measured onsite during July was higher than for other weather stations (Figure 6, Table 1). This is most likely due to onsite humidity being measured in the forested riparian zone local to the stream, whereas offsite humidity is typically measured in an open area at an airport. Days predicted to within 1, 1.5, 2, 3, and 4?C of the measured water temperature for onsite and ROA in July were nearly the same (no more than 1 day difference) despite onsite having the overall highest humidity and ROA having the overall lowest humidity (Table 3, Figure 6). Predictions remained similar between onsite and ROA because onsite had lower air temperatures (Figure 4), which offset the effect of higher humidity increasing predicted water temperature. If this compensatory situation were not present such as in a situation where air temperature differed little between onsite and offsite, but humidity did, it may be warranted to collect parameters onsite. In situations where elevation differs greatly between the stream and offsite weather station it may be feasible to use offsite air temperature and humidity successfully by adjusting them for the elevation difference.

Predictions were poor on days when air temperature fluctuated greatly from one day to the next. This is because the SSTEMP model assumes steady conditions over the simulation period (24 hrs). Though the model uses daily averages to help minimize fluctuations, the model is unable to take into account attributes of the previous day(s). This can cause poor predictions for days where, for example, air temperature, humidity, and/or flow change quickly from what occurred the previous day. Such scenarios were apparent in the January and July model predictions. During January the model predicted higher than the measured water temperature on days where air temperature and humidity rose sharply from the previous day such as on the 2nd, 23rd, and 30th or when air temperature and flow rose such as on the 10th (Figures 1, 3, 5, and 7). The model predicted lower than the measured temperature on days where air temperature and humidity fell from the previous day such as on the 4th and 5th (Figures 1, 3, and 5). In July, the model predicted lower than the measured temperature on days when air temperature fell from the previous day such as on the 12th, 13th, 20th, and 23rd or when flows rose (Figures 2, 4, and 7). From these examples it can be inferred that (1) unsteady conditions from one day to the next cause worse predictions, (2) an increase in air temperature and/or humidity causes an increase in water temperature, (3) during the winter an increase in flow causes an increase in water temperature, and (4) during the summer an increase in flow causes a decrease in water temperature. Though variations in these parameters from one day to the next caused predictions to be off a few degrees for any one day, predictions were off less than two degrees overall for the months of January and July (Table 2).

Though the model was designed to use mean air temperature as an input parameter, use of daily minimum air temperature greatly improved predictive ability during January (Table 3). For example, the model only predicted 4 days to within 1?C when mean onsite air temperature was used, but predicted 17 days to within 1?C when minimum onsite air temperature was used (Table 3). A likely reason for this is that ground temperature remains cold with minimal fluctuation, whereas air temperature fluctuates over a wide range (Figure 3). Therefore, the temperature at which water enters the stream from groundwater and runoff has a greater influence on water temperature than does air temperature. This is also implied from the sensitivity analysis where accretion temperature (i.e., the temperature of lateral inflow) becomes the most sensitive parameter in winter (Figure 8). Air temperature remains one of the more sensitive parameters in winter, but not to the extent it is in summer (Figure 8). Air temperature also fluctuated over a larger range during January than in July (Figure 3 and 4). Thus, during winter months the use of minimum air temperature, which still fluctuates as much as mean air temperature but is closer to the ground temperature, helped improve predictions (Table 3).

## CONCLUSIONS

The SSTEMP model has primarily been used to predict water temperature during summer months when high temperatures are most likely to present unfavorable conditions to aquatic biota. We found predictive ability during the winter to be less accurate than in the summer. We found that use of minimum air temperature for winter water temperature predictions, rather than mean air temperature, allowed the model to achieve a winter predictive ability similar to that which it is capable of during summer months. Winter water temperature predictions close to the measured were found more difficult to achieve due to a larger day to day fluctuation in air temperature and humidity. The air temperature and humidity parameters are more sensitive during the summer making the location of their measurement more important. But even though these parameters were significantly different between onsite and offsite, we found that collection of air temperature and humidity from an offsite location within a reasonable distance and elevation difference from the stream of study provided predictions not significantly different from predictions using onsite collected air temperature and humidity. In situations where study stream characteristics are different from those in this study, such as a groundwater driven or high-mountain stream, predictive ability with offsite data may be different.

## ACKNOWLEDGMENTS

This study was funded by the Virginia Water Resources Research Center and supported by the Virginia Tech Department of Fisheries and Wildlife Sciences. We thank Terry Smith for assistance with field-work.

## REFERENCES


Armour, C. 1991. Guidance for evaluating and recommending temperature regimes to protect fish. U.S. Fish and Wildlife Service. Biological Report 90(22). 13 pp.

Bartholow, J. M. 1989. Stream temperature investigations: Field and analytical methods. Instream Flow Information Paper No. 13 US Fish Wildlife Service Biological Report 89(17). 139 pp.

Bartholow, J. M. 2000. The stream segment and stream network temperature models: a self study course. U.S. Geological Survey, Biological Resource Division, Midcontinent Ecological Science Center, Fort Collins, CO. 276 pp.

Brown, G. W. 1980. Forestry and water quality. School of Forestry, Oregon State University, OSU Bookstores Inc. Corvallis, Oregon.

Hewlett, J. D. and J. C. Fortson. 1982. Stream temperature under an inadequate buffer strip in the southern piedmont. Water Resources Bulletin 18(6):983-988.

Jensen, A. J. 1987. Hydropower development of salmon rivers: effects of changes in water temperature on growth of brown trout (Salmo trutta) presmolts. Pages 207-218 *in* J. F. Craig and J. B. Kemper, editors. Regulated Streams: Advances in Ecology, Plenum Press, New York, NY.

LeBlanc, R. T., R. D. Brown, and J. E. FitzGibbon. 1997. Modeling the effects of land use change on the water temperature in unregulated urban streams. Journal of Environmental Management 49(4):445-469.

Sullivan, K., J. Tooley, K. Doughty, J. E. Caldwell, and P. Knudsen. 1990. Evaluation of prediction models and characterization of stream temperature regimes in Washington. Timber/Fish/Wildlife Report No. TFW-WQ3-90-006, Washington Department of Natural Resources, Olympia, WA. 224 pp.

Webb, B. W. and D. E. Walling. 1993. Temporal variability in the impact of river regulation on thermal regime and some biological implications. Freshwater Biology (Oxford) 29:167-182. Reprint No.: 1993/0790

Table 1. Summary statistics (Wilcoxon's signed rank test p-value, mean difference, and standard deviation) for January and July air temperature and relative humidity values compared between at the stream (onsite) and Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) (n = 31 for all tests).

| | onsite vs. ROA | | | onsite vs. CHO | | | onsite vs. RIC | | |
|---|---|---|---|---|---|---|---|---|---|
| | p-val | mean diff | SD | p-val | mean diff | SD | p-val | mean diff | SD |
| Jan mean air temp | <.0001 | 0.84 | 0.59 | <.0001 | 1.35 | 0.9810 | <.0001 | 2.03 | 1.45 |
| Jan min air temp | <.0001 | 1.41 | 0.97 | <.0001 | 1.76 | 1.3518 | <.0001 | 2.82 | 1.84 |
| Jan mean humidity | <.0001 | 7.22 | 4.20 | <.0001 | 9.39 | 6.2965 | <.0001 | 7.46 | 8.60 |
| July mean air temp | <.0001 | 1.39 | 0.60 | <.0001 | 1.25 | 0.6760 | <.0001 | 2.23 | 1.17 |
| July mean humidity | <.0001 | 12.87 | 4.02 | <.0001 | 8.70 | 8.6985 | <.0001 | 9.62 | 6.84 |

Table 2. Summary statistics (Wilcoxon's rank sum test p-value (two sided pr > z), mean difference, maximum difference, and standard deviation) for absolute difference between measured water temperature and SSTEMP model predictions using at the stream (onsite) measured air temperature and relative humidity values during January and July compared to predictions using values measured at Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) (n = 31 for all tests).

| | measured vs. onsite | | | | measured vs. ROA | | | |
|---|---|---|---|---|---|---|---|---|
| | p-val | mean diff | max diff | SD | p-val | mean diff | max diff | SD |
| Jan (mean air temp used) | * | 3.04 | 7.18 | 1.81 | 0.8664 | 3.09 | 6.72 | 1.87 |
| Jan (min air temp used) | * | 1.39 | 6.32 | 1.48 | 0.5172 | 1.60 | 4.86 | 1.44 |
| July | * | 1.30 | 3.30 | 0.97 | 0.6728 | 1.41 | 3.47 | 0.98 |
| | measured vs. CHO | | | | measured vs. RIC | | | |
| Jan (mean air temp used) | 0.6190 | 2.76 | 7.21 | 1.94 | ** | ** | ** | ** |
| Jan (min air temp used) | 0.9215 | 1.38 | 5.45 | 1.52 | 0.5733 | 1.66 | 4.94 | 1.55 |
| July | 0.3278 | 1.71 | 4.18 | 1.32 | 0.2541 | 1.56 | 3.22 | 0.83 |

**  p-values unobtainable for measured vs. onsite because the absolute difference between measured and onsite represents the benchmark to which absolute differences between ROA, CHO, and RIC and measured were assessed, thus we cannot test measured vs. onsite against itself.*
***  model was not run using January mean air temperature from RIC.*

Table 3. Number of days the SSTEMP model predicted within 1.0, 1.5, 2.0, 3.0, and 4.0?C of the measured water temperature for January when using minimum and mean daily air temperature and for July when using mean daily air temperature measured at the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC). Both months used mean daily relative humidity.

| | January using mean air temperature | | | January using minimum air temperature | | | | July | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Degree criteria | onsite | ROA | CHO | onsite | ROA | CHO | RIC | onsite | ROA | CHO | RIC |
| 1.0 | 4 | 5 | 8 | 17 | 14 | 18 | 13 | 15 | 15 | 12 | 8 |
| 1.5 | 6 | 7 | 9 | 19 | 17 | 20 | 19 | 19 | 18 | 20 | 14 |
| 2.0 | 9 | 10 | 9 | 24 | 21 | 22 | 21 | 22 | 21 | 20 | 20 |
| 3.0 | 14 | 15 | 17 | 28 | 25 | 25 | 24 | 29 | 28 | 25 | 29 |
| 4.0 | 23 | 20 | 23 | 29 | 28 | 29 | 26 | 31 | 31 | 28 | 31 |

**FIGURES**

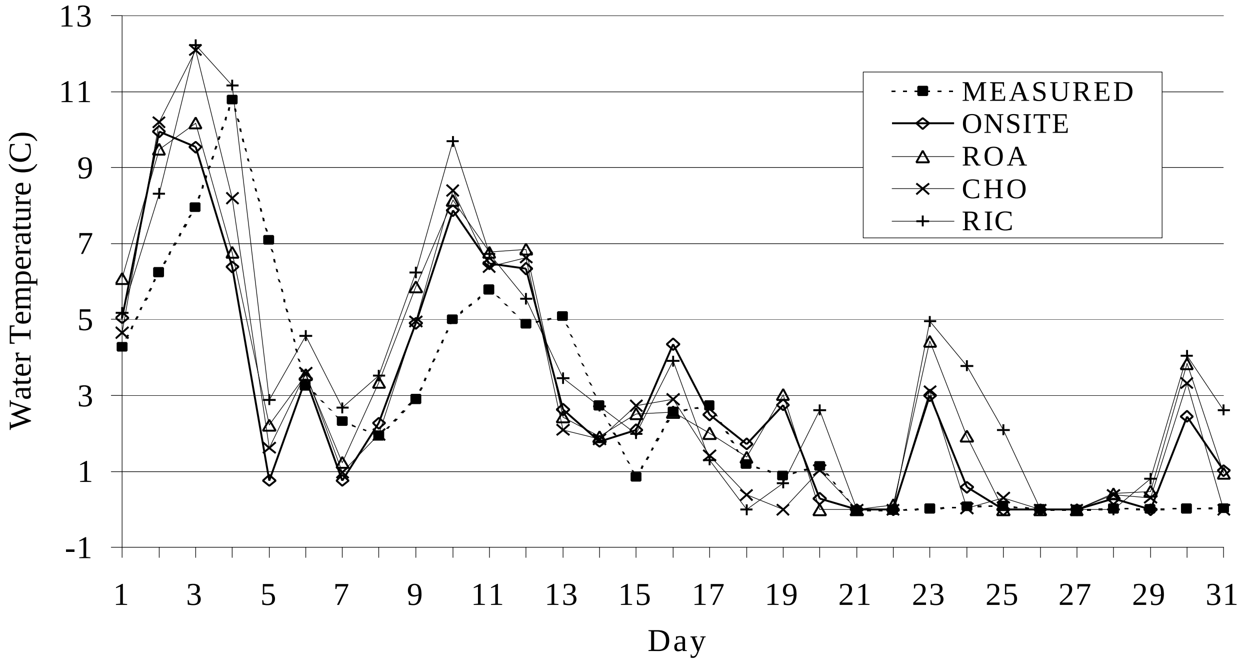



Figure 1. Measured water temperature and SSTEMP model predicted water temperature using air temperature and relative humidity values fromat the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for January.

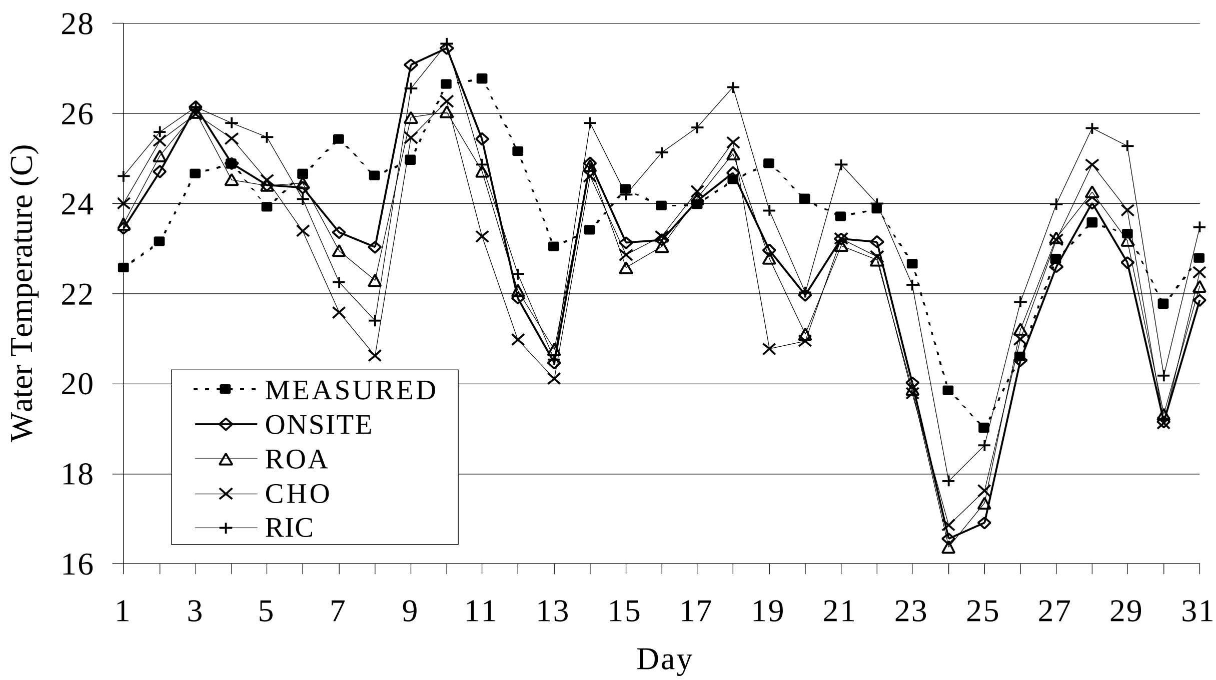



Figure 2. Measured water temperature and SSTEMP model predicted water temperature using air temperature and relative humidity values fromat the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for July.

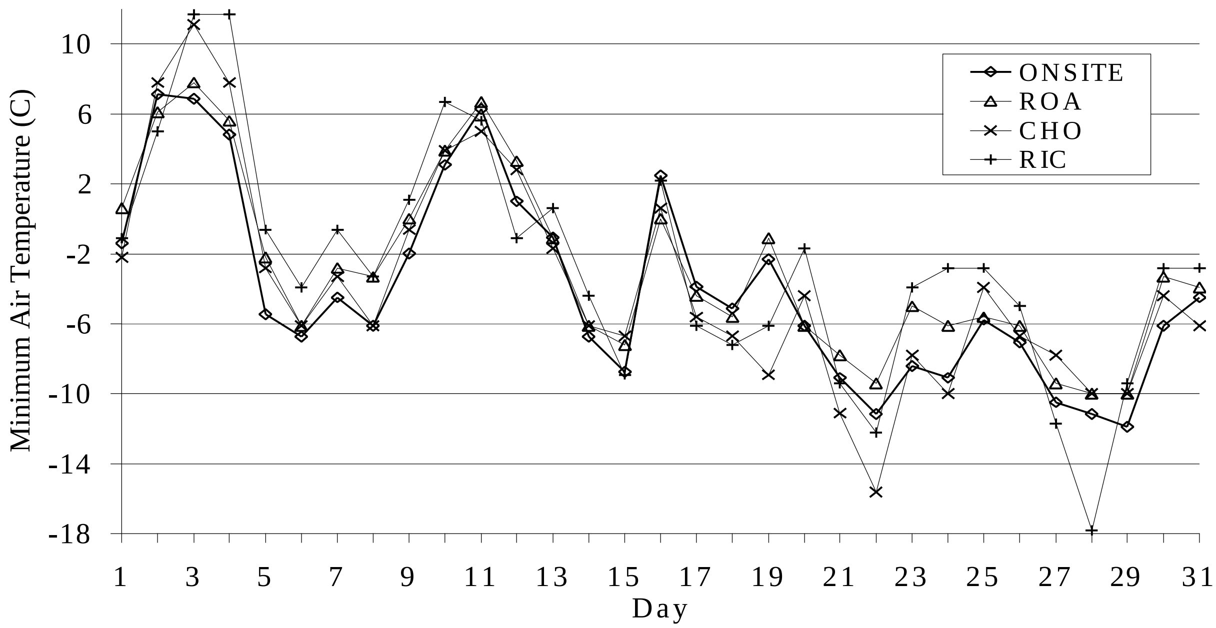



Figure 3. Daily minimum air temperature values from at the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for January.

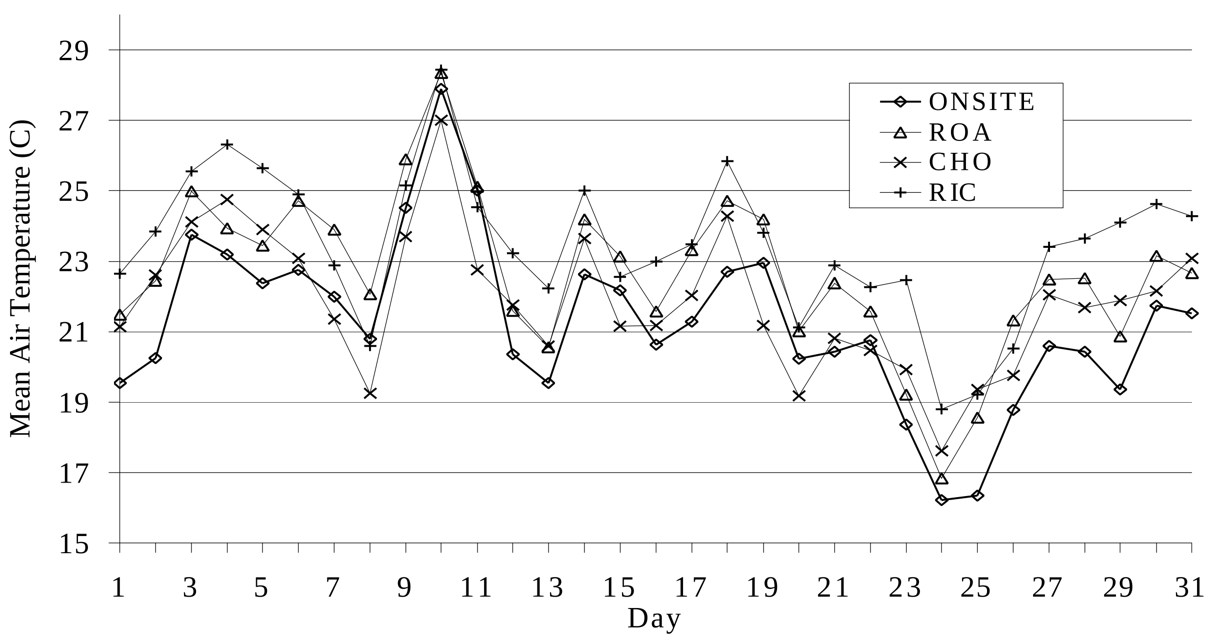



Figure 4. Daily mean air temperature values from at the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for July.

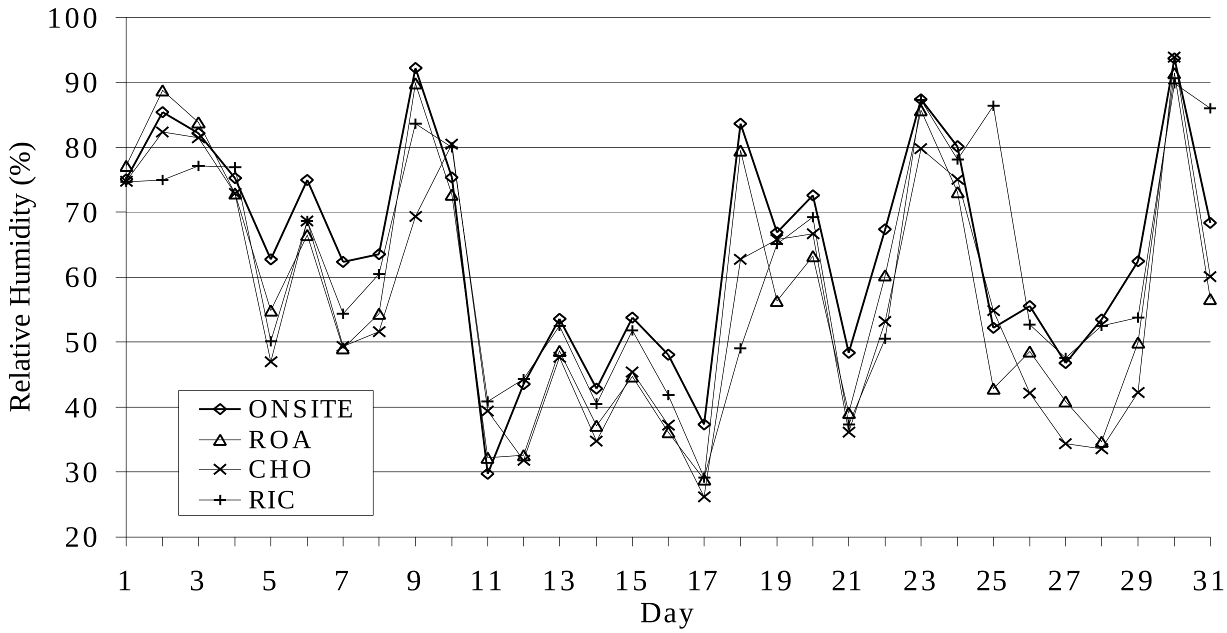



Figure 5. Daily mean relative humidity values from at the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for January.

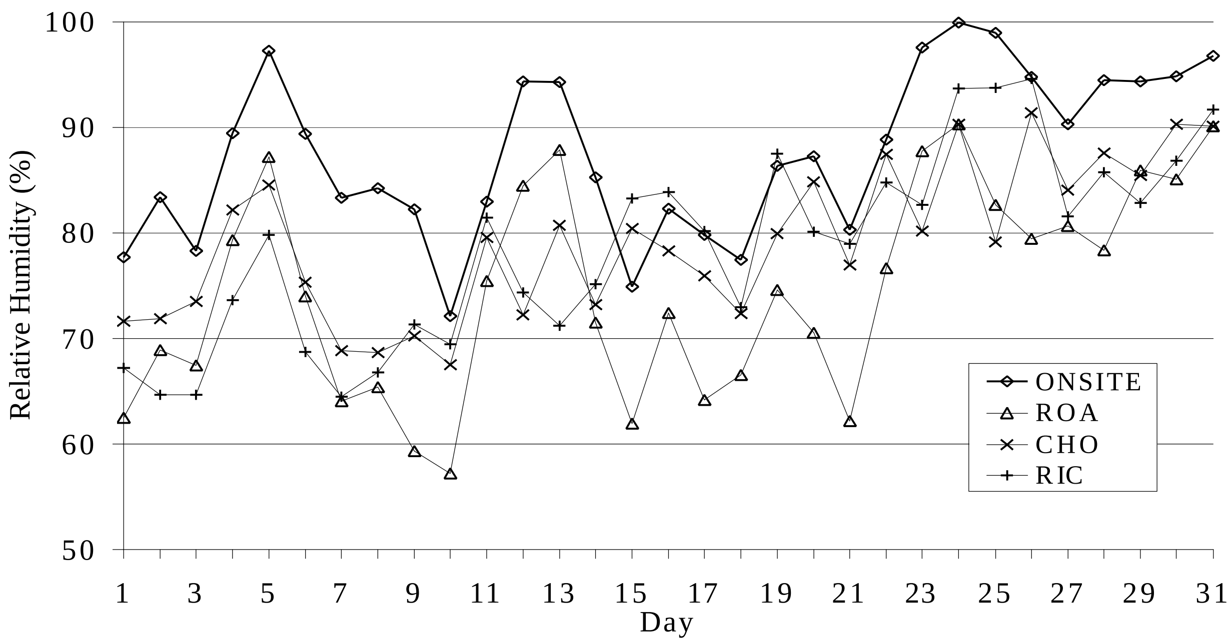



Figure 6. Daily mean relative humidity values from at the stream (onsite), Roanoke Regional Airport (ROA), Charlottesville Albemarle Airport (CHO), and Richmond International Airport (RIC) for July.

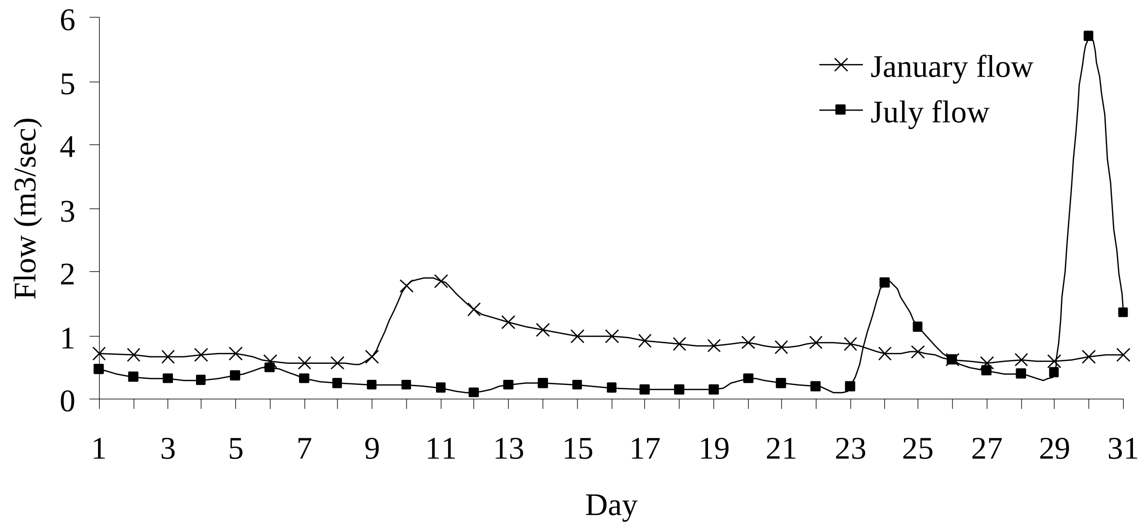



Figure 7. Discharge ($m^3$/sec) in Back Creek during January and July 2000.

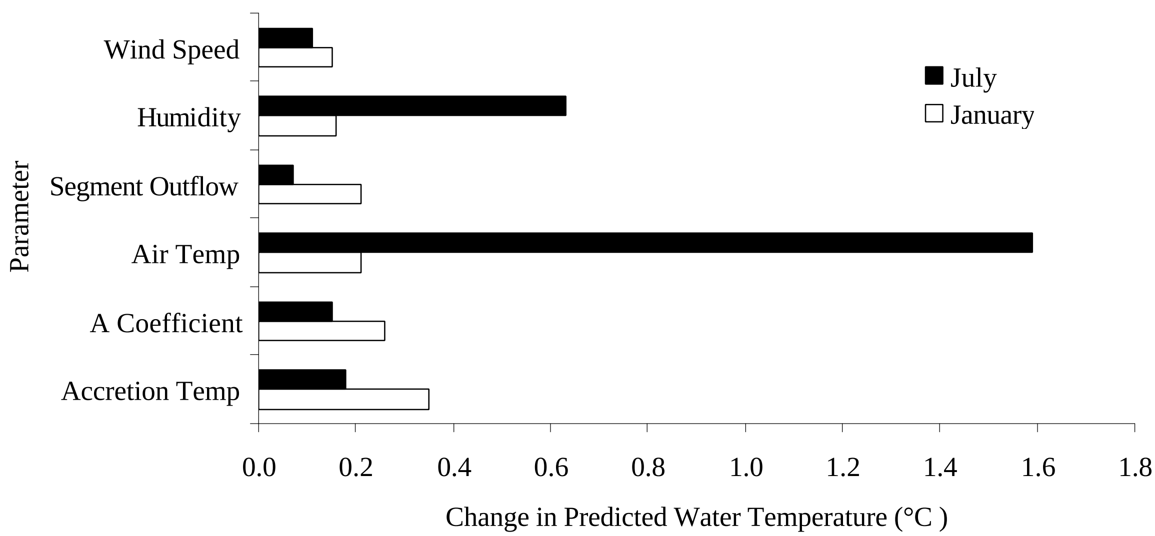



Figure 8. Average change in model predicteddaily mean temperature (?C) for January and July if a single parameter is adjusted by 10%.

**Evaluation of Water Quality Benefits From An Urban Stormwater Wetland System Retrofit**

Dipmani Kumar, Russell Smith, Fred Rose
Stormwater Planning Division
Department of Public Works and Environmental Services
Fairfax County, Virginia

E-mail: dipmani.kumar@co.fairfax.va.us

## ABSTRACT

A regional in-line dry pond, originally constructed in northwest Fairfax County in 1980 for water quantity control, has been identified for retrofitting to provide water quality benefits. The retrofit will provide extended detention of the water quality volume for a period of 24 to 48 hours. An existing wetland community that has developed in the pond will be incorporated into the new extended detention basin. A monitoring program has been initiated to evaluate the performance of the stormwater wetland system before and after retrofitting. The monitoring program includes evaluation of the wetland community in the pond and water quality sampling of the pond inflows and outflow. This paper discusses the preliminary retrofit design, and various design alternatives currently being considered to ensure that the existing wetland vegetation are able to withstand the more frequent and higher levels of inundation expected after retrofitting. The results of a pre-construction wetland vegetation inventory are presented. The water quality monitoring procedures and data analysis methods that will be used to evaluate water quality benefits resulting from the pond retrofit are also discussed.

**Keywords:** Retrofit, Stormwater, Water Quality, Wetlands

## INTRODUCTION

Fairfax County has been involved in storm water management and control for almost 40 years. During the early years, the emphasis was on storm water conveyance and channelization, which included the delineation of flood plains and implementation of flood control projects. Starting in 1972, on-site detention control was required for all new development. In the early 1980's, water quality Best Management Practices (BMPs) were required for new development in the southern areas of the county draining to the Occoquan reservoir, which is the major source of drinking water for the County. BMPs have been required for new development in all areas of the county since 1993.

In the late 1970's, master drainage plans were prepared for all watersheds in Fairfax County. A supplemental Regional Storm Water Management plan was prepared in 1988. In 1989, the County adopted a Regional Storm Water Management Plan, which proposed regional ponds in the most rapidly developing watersheds in the county. The adoption of this plan marked a shift in Fairfax County's approach to implementing storm water management from onsite controls to regional controls, based on the belief that the latter are most effective. It is the county's objective to expand the implementation of regional stormwater management ponds wherever opportunities exist. In addition, opportunities to retrofit existing storm water devices are encouraged. In the context of this paper, retrofitting is taken to mean a process that involves the modification of an existing surface water runoff control structure that was designed to control flooding, so that it will also serve a water quality improvement function.

Reston 913, a 1.8 ha regional in-line dry detention pond, originally constructed in northwest Fairfax County in 1980 for water quantity control, has been identified for retrofitting to provide water quality benefits. The existing stormwater management pond is perceived to have minimal pollutant removal and water quality benefits. The pond retrofit design will provide extended detention of the water quality volume for a period of 24 to 48 hours. Extended detention will be accomplished through construction of a new weir wall outlet control structure. An existing wetland community that has developed in the pond will be incorporated into the new extended detention basin. It is expected that the addition of the weir wall will result in inundation of the existing wetland vegetation more frequently and at higher levels than currently experienced.

As part of a Virginia Department of Conservation and Recreation Water Quality Improvement Fund (DCR WQIF) grant, Fairfax County has initiated a monitoring program to evaluate the performance of the stormwater wetland system before and after construction of the weir wall. The monitoring program includes evaluation of the wetland community in the pond and water quality sampling of the pond inflows and outflow utilizing automated samplers.

## OBJECTIVES

The overall objectives of the monitoring program are to:

- determine the survivability and succession of existing wetland vegetation in a stormwater management facility when the system is retrofitted to provide extended detention.
- evaluate differences in pollutant loadings for selected constituents before and after retrofitting the facility.

## METHODS AND FINDINGS

### Site Overview

Reston 913 is an in-line regional dry detention pond located in the Hunter Mill district of Fairfax County on the south side of Sunset Hills Road near its intersection with Isaac Newton Square. The total area drained by the pond is approximately 126 ha. Of this area, 80 ha drain directly to the pond, while 46 ha drain to a wet pond located southwest of Reston 913. Two unnamed tributary streams draining the pond watershed flow into the facility and confluence within it. After the point of confluence, the stream flows through the facility for about 150 m before exiting through a metal grate outlet structure into a 2400 mm X 1840 mm concrete box culvert (CBC) which drains to the main stem of Colvin Run. One of the unnamed tributary stream flows into the facility at the southern end through a 2100 mm X 1340 mm reinforced concrete pipe (RCP). The second unnamed tributary stream flows into the facility through a 2100 mm RCP pipe at its western end. The western tributary has a total length of about 185 m from the inlet to the confluence while the southern tributary has a total length of about 60 m within the facility. The two streams join at the south-central portion of the facility.

A 750 mm RCP outfall downstream of the western tributary discharges stormwater runoff into the facility from a developed parcel located to the northwest. Stormwater runoff from this parcel is also conveyed to a grassed swale that drains into the facility upstream of the outlet. Land located to the east of the site is currently being developed and additional stormwater discharges to the facility are conveyed through a recently constructed 750 mm RCP outfall on the east side of the pond.

A wetlands study conducted in April last year (WSSI, 1999), identified jurisdictional wetlands associated with the two unnamed tributaries and downstream of their confluence upto the outlet of the stormwater management facility. A sketch of the facility showing all inflow points, the outlet, and the approximate boundaries of jurisdictional wetlands, superimposed on an orthophoto of the site is provided in Figure 1.

## Retrofit Design

The retrofit design for Reston 913 is complicated by the fact that the facility has lost a significant amount of its original storage capacity, and that a wetland system has developed within the pond. A profile and plan sketch of the existing Reston 913 embankment and outfall is shown in Figure 2. The water surface elevations for the 2, 10, 25, and 100 year design floods were obtained from an HEC-1 (McClellan, 1999) analysis. At the current time, only one of the three barrels of a concrete box culvert (CBC) under Sunset Hills road is being used, with the other barrels plugged. There is no emergency spillway, and the HEC-1 analysis indicates the 10-year and higher design floods would overtop the facility.

The proposed retrofit design employs a weir wall to provide extended detention of the water quality volume. To maintain the 10-year and higher design floodwater surface elevations at or below their current levels, it is proposed to utilize a second barrel of the CBC under Sunset Hills Road. A 1650 mm RCP would connect to the second barrel and convey flow to an outfall adjacent to the existing outfall. Construction of the additional outfall will require jacking and boring through the W&OD trail, and open-cutting through the VDOT parking lot. The preliminary construction cost estimate for the retrofit exceeds $280,000.

According to the current Fairfax County Public Facilities Manual (PFM, 1997), extended detention dry ponds are required to have a water quality storage volume of 22 mm per impervious hectare drained, with a minimum 48 hour drawdown of the water quality volume. These standards have to be met to obtain full credit for the 50% Total Phosphorus removal efficiency for dry extended detention facilities. The maximum BMP pool elevation in the preliminary design is slightly higher than the current 2-year design flood elevation, with a 48-hour drawdown time. However, this will clearly inundate existing wetland vegetation more frequently at higher levels and for longer periods than currently experienced. It is unlikely that the current wetland vegetation will be able to withstand the extent of inundation expected with a 48-hour drawdown time.

Schueler (1987) notes that one of the most difficult problems in extended detention design is determining appropriate outlet sizes on the control device so that adequate detention time is achieved for the entire spectrum of storms that are expected at a site. Grizzard et al. (1986) suggest that a detention time of 40 hours for the full water quality volume will result, on average, in a 24 hour detention time for the entire spectrum of storms expected at site.

Based on the need to ensure that the existing wetland vegetation are able to withstand the more frequent and higher levels of inundation expected after retrofitting, the following modifications to the preliminary design are being considered:

- Reduction of the drawdown time to 24-30 hours.

- A two-stage drawdown of the water quality volume to allow rapid (6-8 hour) initial lowering of the maximum BMP pool elevation, followed by a slower (30-40 hour) complete drawdown.

A review of the literature will be conducted to determine how these modifications could impact expected pollutant removal rates. In addition, the county's Maintenance and Stormwater Management Division (MSMD) have recommended the installation of a redundant gate valve and frequent inspections after construction to allow rapid drawdown when necessary to mitigate possible wetland kills.

## Wetland Vegetation Monitoring

An existing wetland vegetation conditions inventory was performed by stratified random sampling of the stormwater management facility in September 2000 to characterize wetland and non-wetland vegetation in the stormwater management facility. Wetland vegetation stratification was primarily evident from the downstream to the upstream end of the channel in the facility, rather than from the channel to the facility boundary. Four wetland zones (strata) were identified within the stormwater facility. The first zone extends from the outlet structure to about 60 m upstream, is relatively flat with channel braiding and sedimentation evident, and is dominated by herbaceous vegetation. The second zone extends from about 60 m upstream to the point of confluence and is also relatively flat, but has a single channel with more woody vegetation. The third and fourth zones extended from the point of confluence to the westerly and southern inlet, respectively, and are characterized by wide shallow channels with relatively dense woody vegetation. Four sampling points were randomly selected within the first and second zones, and two sampling points were randomly selected within the third and fourth zones. The sampling points were monumented and their location recorded using a global positioning system (GPS) unit. Figure 3 shows the wetland zones and sampling points in each zone. Each point was sampled for herbaceous vegetation using a 1 $m^2$ grid, and for woody vegetation using a 3 m diameter plot.

Individual herbaceous and woody species were identified within each sampling area and recorded. The herbaceous vegetation sampling was performed by visually estimating the percentage of the 1 $m^2$ grid under herbaceous cover, and the relative composition of the cover based on different species identified. The woody vegetation sampling was performed by stem counts of the various species identified within the 3 m diameter plot. A summary of the different herbaceous and woody species and the wetland indicator category associated with each species is provided in Table 1. The indicator categories were taken from Reed (1988). Table 2 shows the relative occurrence of herbaceous and woody species within each wetland zone, grouped by probability of occurrence within a wetland. It can be seen that Zone 1 is dominated by herbaceous vegetation with very few woody species. Additionally, Zones 1 and 2 contain predominantly obligate (OBL) wetland species, while zones 3 and 4 contain a mix of obligate and facultative (FACW or FAW) species.

It is expected that following construction of the BMP weir wall, post-construction inventories will be conducted annually at each monumented sampling point in a similar manner beginning August 2002 for a minimum of 2 years.

## Water Quality Monitoring

Pre-construction water quality monitoring activities are expected to commence by December, 2000. At the current time, it has only been possible to establish three automated monitoring stations within the facility because of equipment and personnel limitations. Monitoring stations have been established at the two inlet points for the tributary streams, and the entrance of the facility outlet. Water quality monitoring will consist of manual sampling of baseflow (when observed at a station), and both discrete and composite automated stormflow sampling utilizing Isco 6700 portable samplers. A discrete sample is a grab sample taken within a short period of time, and characterizes the stormwater quality at a given time of discharge. Composite samples are formed by combining discrete samples in some manner, and usually characterize stormwater quality over a longer period of time, such as the entire duration of a storm event. In addition, an attempt will be made to obtain manual grab samples from the two piped stormwater inflows during a selected number of storm events.

Water column depth measurements will be obtained using a steel rule for baseflow samples and stormflow samples that are obtained manually. At the automated stations, an integrated Isco bubbler flow meter will be employed to obtain the stage hydrograph during wet weather events. A recording, tipping-bucket type Isco rain gauge installed near one of the inflow monitoring stations will be used to obtain the storm hyetograph. It is expected that at least 2 baseflow sampling events and 4 wet weather sampling events will take place during pre-construction monitoring and each year of post-construction monitoring. A set of minimum criteria will be established for determining whether a storm is to be monitored, or data from a monitored storm is to be used for analysis. These criteria will include:

- Storm precipitation of at least 5 mm.
- The storm duration is at least 1 hour.
- Storm generated runoff is expected at the outflow monitoring station.

To determine the rise in stage to trigger a sampling event, and also determine an appropriate sampling interval for the automated discrete sampling, the time of concentration for the areas drained by the south and west tributaries as well as the time to peak for various storm events are currently being estimated.

For each wet weather event monitored, a minimum of four discrete samples will be selected for laboratory analysis at each monitoring station. These samples will be selected after observing the stage hydrograph for the event, and will be representative of the first flush, the rising stage, the hydrograph peak, and the recession stage. Depending on the nature of the hydrograph, it may be necessary to select more than four samples to adequately represent these stages. A composite sample will also be obtained at each station. The following constituents will be analyzed in each sample: Total Suspended Solids, Total Phosphorus, Total Kjeldahl Nitrogen, Nitrate plus Nitrite Nitrogen Copper, Lead, Zinc, and 5-day Biochemical Oxygen Demand. In addition, continuous temperature and pH measurements will be obtained using appropriate sensors at the automated stations. Portable pH and and temperature meters will be used for baseflow sampling.

**Data Analysis**

A relational database will be developed to house all data generated from the wetland vegetation and water quality monitoring program. Before conducting any formal statistical tests on the monitoring data, exploratory analysis using summary statistics and graphical representations will be performed. Exploratory data analysis assists in developing a mental picture of the data and can be used to assess the validity of assumptions made in formal statistical tests. For example, the normality assumption used in parametric tests require that the observed data be drawn from a normal (Gaussian) probability distribution. With few exceptions, tests that require the normality assumption perform poorly when the underlying distribution is at least not approximately normal. When the assumption of a normal distribution is difficult to verify, it is generally advantageous to employ nonparametric techniques, which have much less stringent requirements on the underlying distribution

Formal hypothesis tests will be conducted to determine whether differences in pollutant loadings (obtained from composite samples) and peak concentrations (obtained from discrete samples) at the stormwater wetland system outlet before and after construction of the weir wall are statistically significant. If significant differences are indicated, estimators for the difference will be obtained and confidence intervals for the estimator computed. Similar hypothesis tests will be conducted to determine whether significant changes in wetland vegetation characteristics (such as % of obligate wetland species) are indicated. Since the sample size for both pre-construction and post-construction monitoring data will be relatively small and it may be difficult to verify the normality assumption, nonparametric statistical tests will be employed for all hypothesis tests. In addition, since the basic monitoring design is the before and after approach, an important aspect of data analysis will be to take into consideration year-to-year and seasonal climatic variability. This will be done by initially comparing pre- and post-construction sampling data at the inflow points. Since land use in the watershed draining to the inflow points is not expected to change over the monitoring period, the pre- and post-construction inflow loads and peak concentrations should not show significant differences if the storms sampled during the pre- and post-construction periods are similar.

## SUMMARY

Fairfax County emphasizes regional controls in their stormwater management program and encourages opportunities to retrofit existing flood control storm water devices to provide a water quality improvement function. Reston 913, a 1.8 ha regional in-line dry detention pond originally constructed in northwest Fairfax County in 1980 for flood control has been identified for retrofitting.

The pond retrofit design will provide extended detention of the water quality volume for a period of 24 to 48 hours. Extended detention will be accomplished through construction of a weir wall outlet control structure. An existing wetland community that has developed in the pond will be incorporated into the new extended detention basin.

The retrofit design is complicated by the fact that the facility has lost a significant amount of its original storage capacity, and that a wetland system has developed within the pond. To maintain the 10-year and higher design flood water surface elevations at or below their current levels, it will be necessary to utilize a second concrete box culvert barrel under the facility dam embankment that is currently plugged. An additional outfall is then proposed to be connected to the barrel by jacking and boring through an existing embankment and open-cutting through an existing parking lot. Preliminary construction cost estimates for the retrofit exceeds $280,000. Based on the need to ensure that the existing wetland vegetation will be able to withstand the more frequent and higher levels of inundation expected after retrofitting, various design alternatives that allow a shorter than 48 hour draw-down time are being considered.

A monitoring program has been initiated to evaluate the performance of the stormwater wetland system before and after retrofitting. The monitoring program includes evaluation of the wetland community in the pond and water quality sampling of the pond inflows and outflow. A pre-construction wetland vegetation inventory was performed in September 2000 using a stratified random sampling approach. Water quality monitoring, which is expected to commence by December 2000 will consist of manual baseflow sampling, and both discrete and composite automated stormflow sampling at two inflow points and the facility outlet.

Data generated from the monitoring program will be used to determine whether differences in pollutant loadings and peak concentrations before and after construction of the weir wall are statistically significant. Similar hypothesis tests will be conducted to determine whether significant changes in wetland vegetation characteristics are indicated. Since the basic monitoring design is the before-and-after approach, an important aspect of data analysis will be to take into consideration year-to-year and seasonal variability.

## ACKNOWLEDGEMENTS

The authors would like to acknowledge assistance provided by Matt Meyers, Matt Handy, and Amy Maher in wetland plant identification. Rose Barrie provided information on the preliminary retrofit design.

## REFERENCES


Grizzard, T. J, C. W. Randall, B. L. Weand, and K. L. Ellis. 1986. Effectiveness of Extended Detention Ponds. In: "*Urban Runoff Quality-Impacts and Quality Enhancement Technology*". Proceedings of an Engineering Foundation Conference, ASCE.

McClellan, J. W. 1999. Reston Pond 913 Retrofit Preliminary Design. William H. Gordon Associates, Inc. Chantilly, Virginia.

Reed, P. B. Jr. 1988. National List of Plant Species That Occur in Wetlands: National Summary. U.S. Fish and Wildlife Service, Washington DC. Biol. Rpt. 88(24). 244 pp.

Schueler, T. R. 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing BMPs. Metropolitan Washington Council of Governments, Washington D.C.

PFM. 1997. Public Facilities Manual. Fairfax County, Virginia.

WSSI. 1999. Wetland Investigation. Reston 913 – Parcel 3, Fairfax County, Virginia. WSSI #6120. Report submitted to Comstock Partners, L. C. Wetland Studies and Solutions, Inc. Chantilly, Virginia.

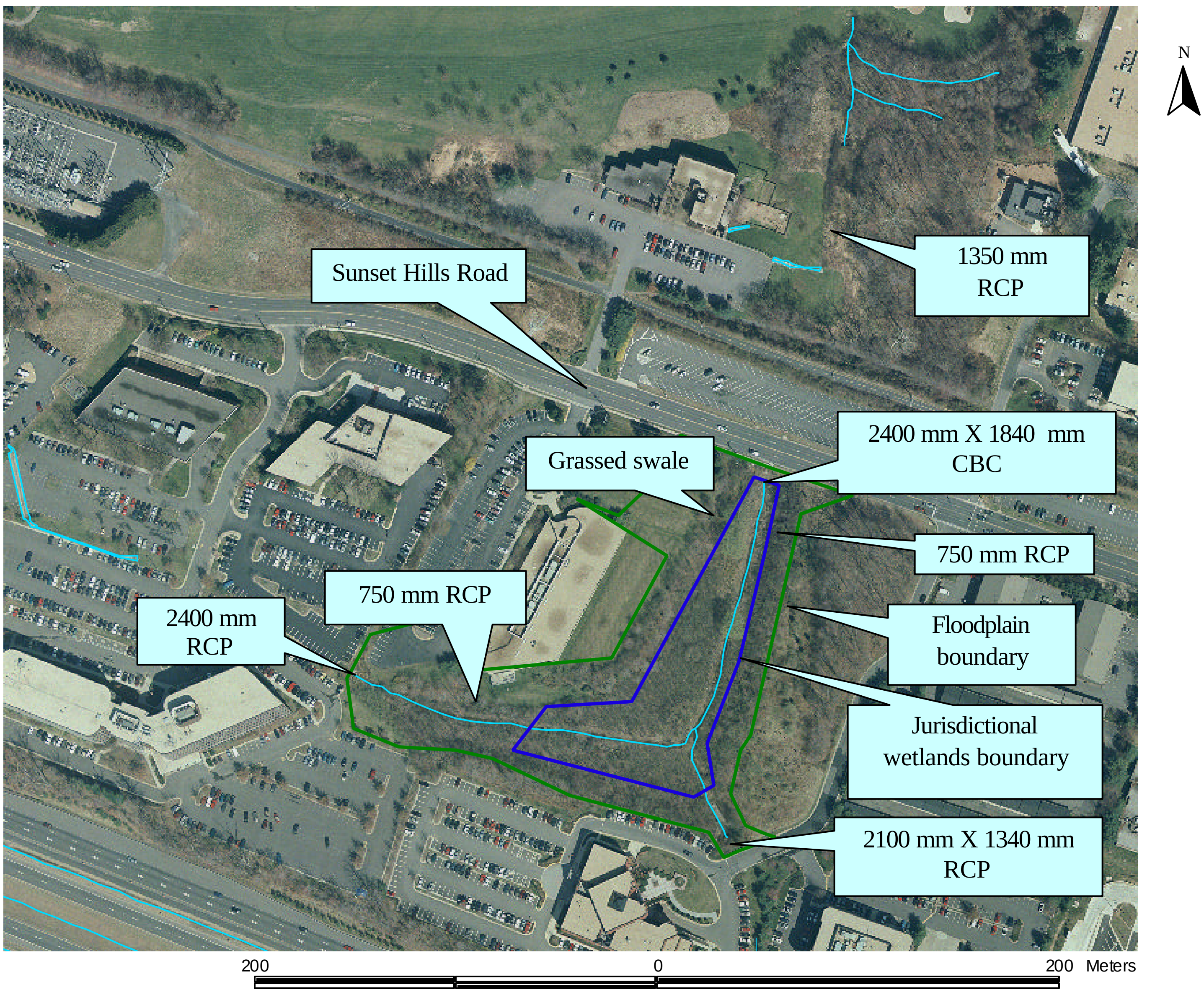



**Figure 1**. Reston 913 – Inflows, Outlet, and jurisdictional wetlands.

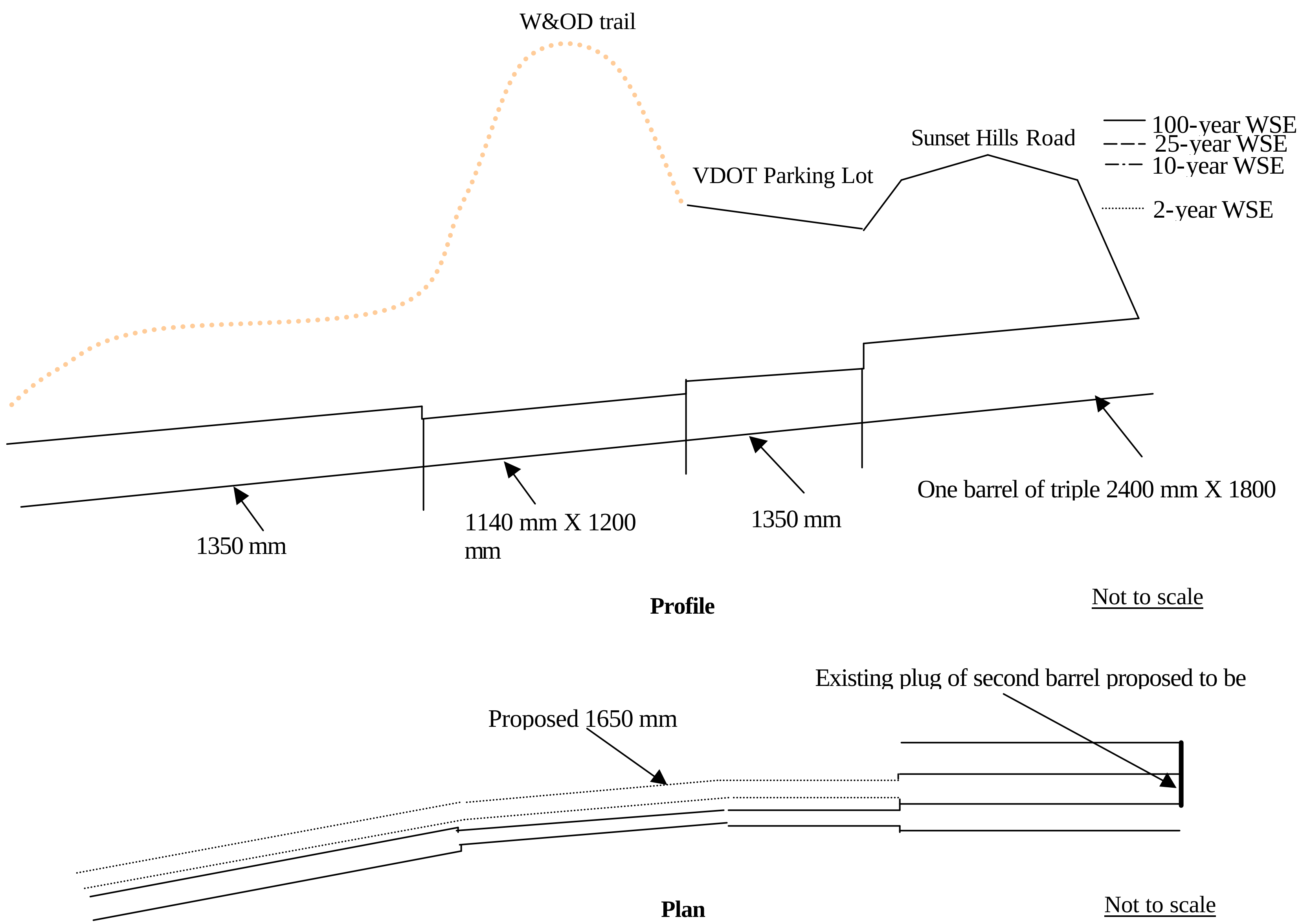



**Figure 2**. Profile and plan sketch of existing Reston 913 embankment and outfall.

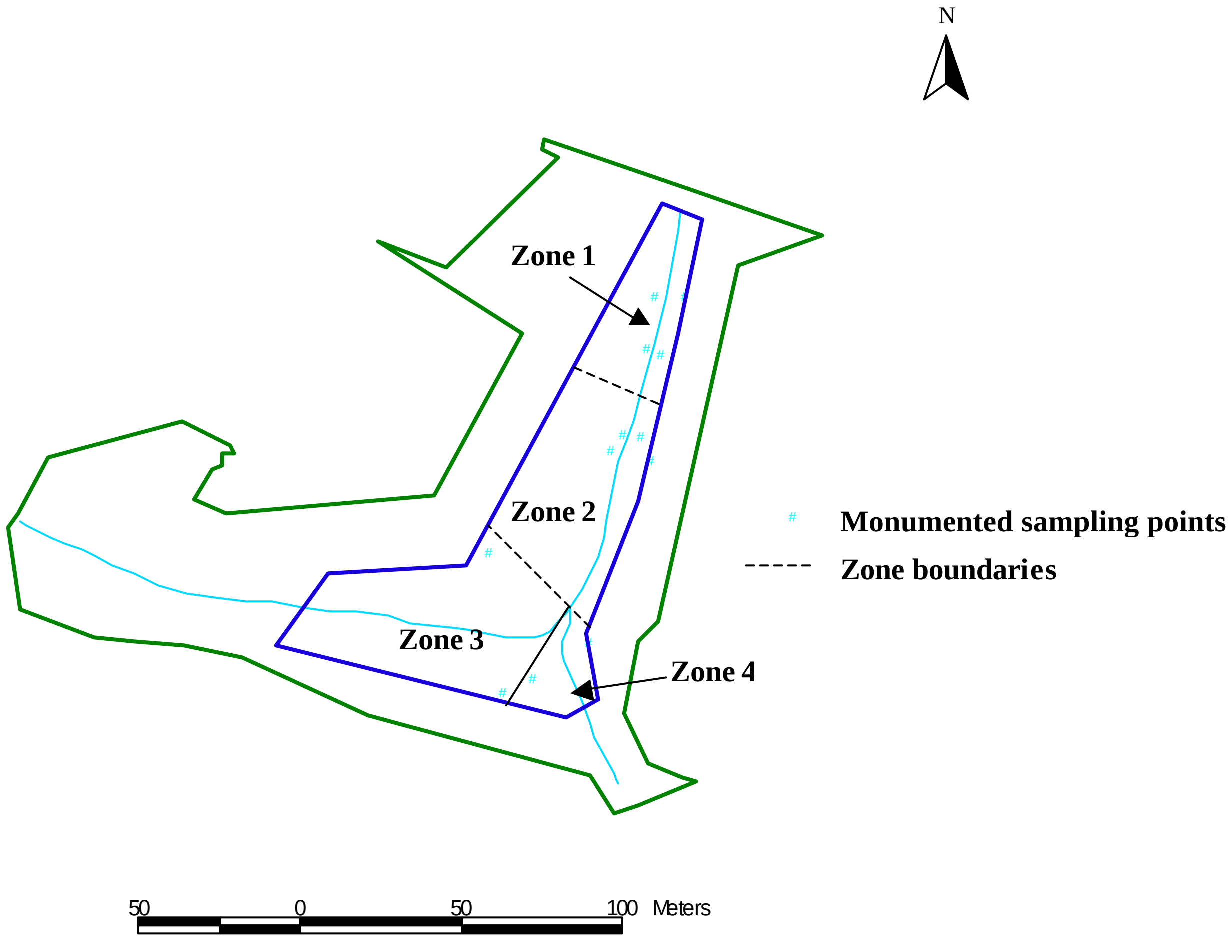



**Figure 3. W**etland vegetation zones and monumented sampling points in Reston 913.

**Table 1.** Summary of different herbaceous and woody species identified in Reston 913.

| **Latin Name** | **Common Name** | **Plant type** | **Indicator**[1] |
|---|---|---|---|
| *Acer negundo* | Box Elder | Herb | FAC+ |
| *Alnus serrulata* | Brook-side Alder | Herb | OBL |
| *Bidens frondosa* | Begger-Ticks, Devil's | Herb | FACW |
| *Boehmeria cylindrica* | False-Nettle | Herb | FACW |
| *Carex stricta* | Tussock Sedge | Herb | OBL |
| *Cornus amomum* | Silky Dogwood | Herb | FACW |
| *Duchesnea indica* | Mock-Strawberry, Indian | Herb | FACU- |
| *Impatiens capensis* | Spotted Touch-Me-Not | Herb | FACW |
| *Phalaris arundinacea* | Reed Canary Grass | Herb | FACW+ |
| *Polygonum hydropiperoides* | Mild Water Pepper | Herb | OBL |
| *Polygonum persicaria* | Ladyes Thumb | Herb | FACW |
| *Acer negundo* | Box Elder | Sapling | FAC+ |
| *Acer rubrum* | Red Maple | Sapling | FAC |
| *Alnus serrulata* | Brook-side Alder | Shrub | OBL |
| *Asimina triloba* | Pawpaw, Common | Shrub | FACU+ |
| *Cornus amomum* | Silky Dogwood | Shrub | FACW |
| *Rhus copallinum* | Sumac, Winged | Shrub | NI |
| *Viburnum dentatum* | Arrow-wood | Shrub | FAC |
| *Acer rubrum* | Red Maple | Tree | FAC |
| *Salix nigra* | Black Willow | Tree | FACW+ |
| *Lonicera japonica* | Japanese Honeysuckle | Vine | FAC- |
| *Toxicodendron radicans* | Ivy, Poison | Vine | FAC |
| *Vitis* sp. | Grape Vine | Vine | NI |

[1] OBL: Obligate Wetland. Almost always occurs in wetlands (estimated probability > 99%) under natural conditions.
FACW: Facultative Wetland. Usually occurs in wetlands (estimated probability 67-99%) but occasionally found in non-wetlands.
FAC: Facultative. Equally likely to occur in wetlands (estimated probability 34-66%) or non-wetlands.
FACU: Facultative upland. Usually occur in non-wetlands (estimated probability 67-99%) but occasionally found in wetlands.
UPL: Obligate upland. Occur almost always (estimated probablility > 99%) in non-wetlands under natural conditions
NI: No indicator assigned.
A positive or negative sign is assigned to the facultative categories. A + sign indicates frequency towards the wetter end of the category (more frequently found in wetlands), while a – sign indicates frequency towards the drier end of the category (less frequently found in wetlands).

**Table 2.** Relative occurrence of herbaceous and woody species within wetland zones.

| Wetland Zone | Site ID | % herbaceous cover[1] | Stem Count[2] | Herbaceous species | | | Woody species | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | % H[3] | % M[3] | % L[3] | % H[3] | % M[3] | % L[3] |
| 1 | 1BL | 30 | 2 | 98 | 2 | 0 | 0 | 100 | 0 |
| | 1BR | * | 7 | - | - | - | 100 | 0 | 0 |
| | 1AL | 100 | 0 | 0 | 100 | 0 | - | - | - |
| | 1AR | 5 | 81 | 50 | 50 | 0 | 96 | 4 | 0 |
| 2 | 2BL | 5 | 80 | 72 | 28 | 0 | 100 | 0 | 0 |
| | 2BR | 1 | 49 | 100 | 0 | 0 | 80 | 20 | 0 |
| | 2AL | 8 | 235 | 100 | 0 | 0 | 99 | 1 | 0 |
| | 2AR | 2 | 44 | 0 | 100 | 0 | 70 | 30 | 0 |
| 3 | 3L | 1 | 14 | 50 | 50 | 0 | 40 | 60 | 0 |
| | 3R | 1 | 40 | 50 | 50 | 0 | 70 | 30 | 0 |
| 4 | 4L | 5 | 108 | 80 | 20 | | 68 | 27 | 5 |
| | 4R | 25 | 18 | 5 | 95 | | 55 | 45 | 0 |

[1] % of total 1 $m^2$ sampling area with herbaceous cover (any portion of the sampling area with standing or running water is not included). Asterisk for site 1BR indicates that at the time of sampling consisted entirely of a channel formed by a new stormwater outfall discharge).

[2] Number of woody stems counted in the 3 m diameter sampling area.

[3] % H is the percentage of species with a high (> 99%) probability of occuring in a wetland (i.e. OBL species). % M is the percentage of species with a moderate ( > 33 %) probabilty of occuring in a wetland (i.e. FACW or FAC species). % L is the percentage of species with a low (≤ 33 %) probability of occring in a wetland (i.e. FACU or UPL species).

# SAV Transplant Survival in the Tidal James River Relative to Water Quality

Kenneth A. Moore, Virginia Institute of Marine Science, Gloucester Point, VA 23062
E-mail: moore@vims.edu
Jeanie Grandstaff, Hopewell Regional Wastewater Facility, P.O. Box 969, Hopewell, VA 23860
E-mail: jgrandstaff @hrwtf.org

## ABSTRACT

Native species of submerged aquatic vegetation (SAV) including wild celery *(Vallisneria americana*), and sago pondweed *(Potamogeton pectinatus*) were transplanted in May 1999, to four, shallow water (0.3 m MLW) sites in the Hopewell estuary region of the James River. The SAV transplants were sampled for survivorship and growth at bi-weekly to monthly intervals. Concurrently, water quality sampling was conducted at bi-weekly intervals for nutrients, chlorophyll a, suspended solids, water transparency and other chemical and physical constituents. This marks the first time SAV has been successfully transplanted into this region since declines were observed in the 1940's. At three of the four sites the SAV grew throughout the 1999 growing season and at two sites the SAV re-sprouted in the spring of 2000. There was significant initial herbivory of the SAV transplants by fish, turtles or other animals, which was effectively stopped by encirclement of the plots by 1-inch, wire mesh fencing. Water quality conditions at most sites were characterized by high levels of suspended sediments (>50 mg/L) during the spring and high phytoplankton levels (>50 µg/L) during the summer, which typically exceeded 25% of the suspended particle loads during that time. Transparency, measured as secchi depth, was lowest during the summer when levels were typically at 0.3 m. Application of Chesapeake Bay Program water quality criteria models to the monitoring data confirmed that conditions in the region are poor for SAV growth to depths of one meter or greater due principally to high turbidity, however, growth at depths of less than 0.5 m are supported. Model estimates of periphyton loading on the SAV, however, overestimated by ten-fold the actual loadings that were measured using artificial substrates. Lack of current SAV re-colonization in the very shallow water areas of this region may be related to physical and biological factors not directly related to the water quality constraints of deeper sites.

**Keywords:** SAV, submerged aquatic vegetation, James River

## INTRODUCTION

The Commonwealth of Virginia Draft Tributary Strategy, "Goals for Nutrient and Sediment Reduction in the James River", identifies reduced light penetration preventing the growth of submerged aquatic vegetation (SAV) as one of the key issues regarding water quality and living resource impacts. The strategy states, "Restoration of grass beds to the upper tidal river will greatly expand existing recreational fishing opportunities for largemouth bass and other tidal fresh sport fish. Once grass beds gain a foothold, they will also begin to improve water quality themselves by stabilizing shorelines, minimizing re-suspension of sediments into the water due to wind and waves, and filtering nutrients out of the water."

In addition, EPA has listed the James River on the 303(d) List as impaired for aquatic life use attainment. Since, SAV is a vital resource that produces oxygen, provides nursery habitat, food, and protection for a variety of aquatic organisms, reduces the erosion effect of wave energy, absorbs nutrients and other pollutants, and traps sediment, it serves as a crucial component of a healthy James River. Therefore, restoration efforts are closely tied to water quality and water quality improvements.

Analyses of historical aerial photographs and ground survey reports for submerged aquatic vegetation (SAV) in the James River have revealed evidence that some areas of the James River near the City of

Hopewell may have supported SAV growth until the mid-1940's (Moore et al. 1999). Currently, SAV is found only in some tributary creeks in the vicinity of the Chickahominy River. The current lack of growth of SAV in many shallow areas of the tidal, freshwater James River may be related to a number of factors including:

- Poor water quality due to high turbidity and high nutrient levels,
- Poor sediment characteristics (High organic content),
- Physical limitation due to biological or physical disturbance (Batiuk et al. 1992)
- Limited SAV propagule supply.

Although many freshwater SAV species can be transported by a variety of mechanisms such as seed dispersal by waterfowl or rafting of shoots by tidal currents, limited propagule supply or survival may be contributing to the lack of re-growth in this region. One way to assess these various hypotheses is to use experimental SAV transplants to test the current suitability of the areas for SAV growth and then evaluate the various factors that may impact their survival. Using SAV plants directly can provide an integrated measure of habitat suitability that cannot be determined solely by discreet monitoring of physical and chemical habitat conditions.

Previous studies, beginning in 1996, conducted for the Hopewell Regional Wastewater Treatment Facility (HRWTF) as part of the Hopewell Estuary Region Monitoring Assessment (HERMA) project consisted of an initial screening assessment of existing water quality and biological monitoring data as well as modeling results for the Hopewell estuary region. This was followed in 1998 by an ambient water quality monitoring study at a series of stations in this region. This study was designed to gain a greater understanding of the status and controls on water quality in both the region directly impacted by the combined Gravelly Run discharge, and in the James River region near Hopewell outside of the mixing zone (Malcolm Pirnie, Inc. 1999). Results indicated that light conditions in the Hopewell estuary region are generally poor for SAV growth and do not meet habitat criteria for restoration of SAV growth to a depth of one meter. Water column light attenuation by suspended sediments and phytoplankton was also found to be higher in near-shore stations than in mid-channel stations, possibly due to tidal and wind re-suspension of sediments. However, target concentrations for attainment of SAV habitat criteria to depths of 0.5 m or less were met, suggesting that SAV growth at these depths may be possible. A SAV restoration study would be necessary to determine if SAV could actually grow in this area.
Based upon the results of these water quality monitoring and historical SAV distribution studies, the Hopewell Regional Wastewater Treatment Facility (HRWTF) sponsored a study, in cooperation with the Virginia Institute of Marine Science (VIMS) and the Chesapeake Bay Foundation (CBF), to investigate the relationships between current water quality conditions and SAV growth and survival in this region. The study included the transplanting of SAV (wild celery and Sago pond weed) into freshwater tidal section of the middle James, as well as semi-monthly water quality monitoring for BOD, solids, nutrients, chlorophyll and physical parameters.

## OBJECTIVES

The objectives of this first year SAV restoration study, funded by the Hopewell Regional Wastewater Treatment Facility (HRWTF) and assisted by the Chesapeake Bay Foundation (CBF), were to:

1) Develop and evaluate techniques for effective transplantation of native SAV species to this region of the estuary;
2) Determine, if under current conditions, SAV transplants could survive in selected sites in the Hopewell Region of the James River estuary;

3) Evaluate the response of transplants relative to specific water quality conditions at the sites (monitored by HRWTF), site characteristics, or physical disturbance, as well as various model predictions of habitat suitability.

## METHODS

### Study Sites

Four sites were selected for test transplanting in the Hopewell region of the James River estuary. Site selection was based upon review of a number of factors including: water depth (<0.5m), site orientation and location (low erosion shoreline), sediment type (<5% organic content), photographic evidence of historical SAV occurrence, and background review of general water quality conditions in the area. Based upon this review and a field survey of the area four sites were selected along the littoral zone of the river for the transplanting efforts (Fig.1).

| | | |
|---|---|---|
| Turkey Island | Lat. 37.3826 N | Long. 77.2527 W |
| Shirley Cove | Lat. 37.3326 N | Long. 77.2631 W |
| Tar Bay Island | Lat. 37.3075 N | Long. 77.1902 W |
| Powell's Creek | Lat. 37.2979 N | Long. 77.1622 W |

### Preliminary Transplanting

On May 1, 1999, an initial pilot transplanting was undertaken at the Shirley Cove site. Whole plants of *Vallisneria americana* (wild celery) were supplied by the Chesapeake Bay Foundation (CBF). With the help of CBF personnel and citizen volunteers, approximately 600 plants, ranging from 5 to 10 cm in height were cleaned of sediments, then planted by a Virginia Institute of Marine Science (VIMS) scientific diver in 6 replicate (2m x 2m) arrays of 100 planting units spaced at 0.12 m intervals. Planting units consisted of single, bare-rooted shoots that were placed directly in the sediment to a depth of approximately 5 cm using no anchoring device (cf. Orth et al. 1999). Water depths varied between approximately 0.1 and 1.0 m below MLW. Each replicate plot was delineated with white PVC poles

On May 17, 1999, the transplanted arrays were checked for survival. Plot survival ranged from 0-25 % with survivors showing evidence of shoot cropping by unknown herbivores such as fish, turtles, or waterfowl. Surviving shoots were only 3-5 cm in length with jagged, cut off leaf tips. In addition, most of the PVC poles which extended above water had been moved and replaced further channelward by unknown individuals. There was also evidence that the bottom within the plots had been disturbed, possibly by burrowing or browsing activities.

### Multi-site Transplanting: Plant Establishment and Site Monitoring

<u>SAV Transplant Establishment</u>
On June 1-2, 1999, replicate 2m x 2m plots of *V. americana* and *Potamogeton pectinatus* (sago pondweed) planting units were planted on 0.25 m intervals at each of the four transplant sites by VIMS with assistance from CBF and HRWTF personnel. Water depths at the planting sites were estimated to be between 25-50 cm below MLW. The wild celery plants were supplied by CBF, while the sago pondweed plants were harvested by VIMS personnel from native populations in the Poropotank River, VA. Each set of transplant plots was protected from disturbance by use of exclosures consisting of staked, wire fencing of 1-inch mesh, which extended from the sediment surface to above high water. Each site was sampled by divers for SAV planting unit survival, SAV relative abundance and plant vigor monitored at semi-monthly to monthly intervals throughout the 1999 growing season and again in the spring of 2000. In July

the transplants at the Shirley Cove site had to be removed and replanted at the Turkey Island site due to dredge spoil deposition at the site by the US Army Corps of Engineers.

Periphyton Monitoring

Rates of periphyton accumulation (ie. combined algae growth and sediment loading on the plants) were monitored by use of artificial substrates (Neckles 1990). Although the use of artificial substrates precludes any potential biological or chemical influences of the SAV on periphyton composition or mass, the benefits of standardization and replication have made this technique valuable for relative site comparisons of fouling (Robinson 1983). Artificial plants consisted of two 50 cm long strips of 5 mm wide polypropylene ribbon attached to a 0.25 m square made of iron bars criss-crossed with lines at 10 cm intervals to form a base. Replicate squares of the artificial substrates were placed within each exclosure at each site. Two sets of artificial leaves were sampled from each square at semi-monthly to monthly intervals by clipping the strips at their base, and placing the entire strip in a zip-lock bag. The bags with the artificial leaves were transported to the lab where they were frozen. At a later date the samples were thawed, the fouling community gently scraped off the substrate into freshwater using the edge of a glass slide, collected by filtration, dried at 50 °C and weighed.

Sediment Characterization

Sediments at each transplant site was characterized by use of replicate cores taken at each of two locations (shallow side and deepest side) within each exclosure. The six-inch deep cores were mixed to provide a homogeneous sample, dried weight at 50 °C to a constant, weighed for dry weight, ashed for 5 hours at 550 °C and weighed again. Organic content was determined by weight difference.

Water Quality Monitoring

Water quality sampling was conducted at bi-weekly intervals by HRWTF. Water samples were typically collected at depths of 0.5 to 1.0 m in the shallow littoral area immediately adjacent to the transplant locations. Parameters measured included air and water temperatures, secchi depth, pH, dissolved oxygen (DO), conductance, total Kjeldahl nitrogen (TKN), nitrate + nitrite (NOx), ammonium, orthophosphate (DIP), total phosphorus (TP), total suspended solids (TSS), total organic carbon (TOC), and chlorophyll a (Chl a).

## RESULTS

### Transplant Survival

Wild Celery Survival

Survival of the wild celery transplants is summarized in Figure 2A. Survival of the planting units at each of the planting sites was first determined on June 18, 1999. In contrast to the loss of plants observed in the unprotected preliminary plantings at the Shirley Cove site in May, initial survival at three of the four protected sites ranged from 50% to 100%. The Powell’s Creek site had the lowest initial survival with survival rates averaging 60%. The other three sites demonstrated 100 % survival for over one month. Qualitative observations indicated that within several weeks there was new vegetative growth, suggesting that the plants were beginning to become established at this time. By six weeks (July 16) the leaves had grown to a length of approximately one meter and new shoot clusters were observed. The successful results of the protected transplants, in comparison to the apparent herbivorous cropping and lack of survival of the earlier unprotected transplants suggests that, at least initially, wild celery transplant survival in this region may be limited by grazing or disturbance activities of fish, turtles, or other animals. These confounding sources of impact have also been observed during transplanting efforts in other freshwater tidal regions of the Chesapeake Bay system, including the Potomac River and upper bay in Maryland (Carter and Rybicki 1985).

Although, in general, the wild celery plants were observed to be growing throughout the summer there was an apparent loss of planting units at the Turkey Island and Tar Bay sites. However, because of poor water clarity and nature of the soft sediment, which was easily stirred up by walking, it was possible that the survival rates at these sites were underestimated. Transplants at the Shirley Cove site were removed and replanted at the Turkey Island site in mid-July, just prior to the deposition of dredge spoil in the cove. The planting units at the Shirley Cove site that were removed and replanted at Turkey Island on July 16 were found to quite healthy, with many having produced three or more new leaf clusters. By August all three of the remaining sites demonstrated 60% to 80% survival rates. Declines in the survivorship at all of the sites between August and the end of September were likely related to the normal end-of-season die-back of shoot material and the resultant storage of the plant resources as below-ground tubers and over-wintering buds.

Sago Pondweed Survival

In contrast to the long-term growth and survival of the wild celery transplants, the sago pondweed planting units, while demonstrating general expansion and elongation of shoot material, gradually disappeared throughout the summer (Fig. 2B). Few new shoots were produced during this time. This may have been related to the type of planting material, which consisted of transplants of shoots that were harvested from a natural stock in the Poropatank River in Virginia. A lack of apical meristems in the rootstock of the source material may have contributed to poor new shoot production. Loss of planting units appeared more related to physical breakage or dislodgment of the shoots due to currents or wave action than dieback of the plants themselves. The canopy-type growth of this species may have contributed to this dislodgment as the individual plants consisted of a dense, canopy of leaves at the distal end of a long, thin stem that measured a meter or more in length. A number of broken shoots were observed throughout the summer as they were caught on the inside of the wire fencing

**Habitat Monitoring**

Sediments

Sediments at the transplant sites (Fig. 3A) were within the general range of organic content that will support SAV growth. Typically, the range of suitable sediments is between 0.5 and 5 % organic content, although SAV have been observed to grow successfully in higher organic substrates (Barko and Smart 1983). Tar Bay was situated between two islands and tidal currents and wave action likely maintained the low organic conditions there. At the Turkey Island and Shirley Cove sites there were marked increases in sediment organic content with water depth. A large sand bar that was an apparent relic of previous dredge disposal operations characterized the shallowest area of the Shirley Cove transplant site. Organic content rapidly increased with water depth here. The Turkey Island site was adjacent to an eroding bank that was the likely source of sand in the shallowest depths where the organic content was lowest. The Powell's Creek site, which was situated along a reach of sandy shoreline, demonstrated less variability in substrate type within the planting area. Just offshore of the transplant plots the sediments were qualitatively much more organic rich. All of the sites, however, were chosen so as to minimize the organic content of the sediments compared to surrounding areas which were primarily composed of soft muds.

Artifical Substrate Fouling

Periphyton mass accumulations on the artificial substrates (Fig. 3B) remained consistent after placement at the sites in mid-July. Accumulations were minimal until the end of October when strips at both the Powell's Creek and Tar Bay sites showed high levels of periphyton mass. However, the strips at both of these sites appeared to have lost some buoyancy by the end of October when they were found to not be floating vertically in the water. This increased mass may have been from bottom sediments that had become attached to the strips. In contrast, strips at the Turkey Island site that remained buoyant showed little change throughout the growing season.

Water Quality

Approximately one year of water quality data are summarized in this report. In general there are few consistently large differences between sites. Water temperatures (Fig. 4A) demonstrated strong seasonal patterns, with lows in mid-February of 7-8 °C and highs in late July of 30-34 °C. Conductivity increased throughout the summer (Fig. 4B) achieving a maximum at all sites in mid-September just prior to the passage of a Tropical Storm Floyd. Highest values were observed in the most downstream station (Powell's Creek) where peak conductance levels of 900 µmhos equated to a salinity of approximately 0.3 PSU, or less than 1% that of seawater. These salinities are well within the range of tolerance for both transplanted SAV species. DO concentrations demonstrated bi-modal annual patterns (Fig. 4C) and lowest values were recorded in the spring to early summer period as well as in the fall. Daytime values reported for the shallow water transplant sites did not typically fall below 5 mg/l. pH levels (Fig. 4D) were relatively consistent throughout the year, although lowest levels coincided with periods of minimum DO in June and September.

Suspended particle loads (TSS) were typically highest in the spring and decreased to lowest levels in October (Fig. 5A). In contrast to other measured water quality parameters no apparent effect of Tropical Storm Floyd on TSS levels were observed at any of the sites. TSS levels were consistently lowest within the embayment of the Shirley Cove site, especially during the spring, with little annual variability observed at this location compared to the other transplant locations. TSS consistently exceeded the habitat requirement of 15 mg/l established by the Chesapeake Bay Program (Batiuk et al. 1992) for SAV restoration to one-meter depth at all sites (Table 1). Phytoplankton, measured as Chl a, demonstrated consistently high levels during the summer as well as a second smaller peak during the winter (Fig. 5B). These peaks greatly exceeded the SAV habitat requirement of 15 µg/l for freshwater regions. However, seasonal medians for all the sites were below the established habitat requirement (Table 1). This was due, in part, to the low phytoplankton abundance during the spring and early summer when non-phytoplankton derived suspended particles in the water column were high. There were no marked differences among the sites. Phytoplankton comprised a relatively large proportion of the total suspended particle concentrations (Fig. 5C) during August and again during several peaks in December and February. Water transparency measured as secchi depth (Fig. 5D) showed a seasonal decline to minimum levels of 0.3 m or less in August and September. Light transparencies reported as median seasonal light attenuation (Kd) or secchi depth (Table 1) did not meet the habitat criteria for SAV growth to one meter at any of the sites.

**Table 1. 1999 James River Transplant Site Water Quality and Habitat Criteria for Restoration of SAV to a Depth of One Meter**

| | SAV Growing Season Medians (April-October) | | | | |
|---|---|---|---|---|---|
| | SAV Habitat Criteria | 1999 Transplant Sites | | | |
| **Parameter** | | **Powell's Creek** | **Tar Bay** | **Shirley Cove** | **Turkey Island** |
| Light Attenuation (Kd ; $m^{-1}$) | <2 | 4.0 | 3.4 | 2.5 | 3.6 |
| Secchi Depth (m) | >0.7 | 0.30 | 0.33 | 0.40 | 0.3 |
| TSS (mg/L) | <15 | 38 | 31 | 21 | 33.5 |
| Chl a (µg/L) | <15 | 12.6 | 12.0 | 13.7 | 11.1 |
| DIP (mg/L) | <0.02 | 0.01 | 0.02 | 0.01 | 0.01 |

TOC, TKN and TP were characterized by somewhat different patterns of abundance. Nitrogen levels (Fig. 6A) were variable among the sites and generally decreased to below detection limits (0.5 mg/l) during the fall. Total organic carbon concentrations (Fig. 6B) increased throughout the summer and reached a peak after the passage of Tropical Storm Floyd in late September. Total phosphorus concentrations (Fig. 6C) were relatively consistent throughout the year at all sites, although a slight downward trend throughout the year paralleled that of total TSS and TKN.

Dissolved inorganic nutrients (nitrogen and phosphorus), which are readily available for uptake by phytoplankton and epiphytic algae, demonstrated relatively high levels for nitrogen and low levels for phosphorus. DIP concentrations typically met the SAV habitat requirement threshold of 0.02 mg/l for the tidal fresh salinity regime at all sites for most of the year (Fig. 6D; Table 1). Dissolved inorganic nitrogen (DIN) habitat requirements have not been established for freshwater tidal areas as with phosphorus. Phosphorus is typically considered the limiting nutrient for algae growth in these areas, whereas nitrogen can be quite high in many areas of freshwater SAV growth. In low salinity regions, however, SAV growth to one-meter depths has been found to be associated with DIN concentrations of 0.15 mg/l or lower. In this study ammonium, which is one component of DIN (ammonium + nitrate + nitrite), was nearly always at or below the detection limit of 0.2 mg/l (data not shown). NOx concentrations (Fig. 6E) were typically highest in the fall and winter and lowest during July and August

<u>Attainment of Conditions Suitable for SAV Growth</u>
Using a "Diagnostic Tool" developed by Dr. Charles Gallegos of the Smithsonian Environmental Research Center for the Chesapeake Bay Program SAV Technical Synthesis II Workgroup, TSS and Chl a data collected during the SAV growing season were evaluated to predict if SAV growth was possible at specific target depths. This modeling tool also estimates the median TSS and Chl a concentrations that would be necessary for attainment of habitat criteria at those depths. The target concentrations correspond to the scenarios of reductions in both TSS and Chl a, reducing only Chl a, or reducing only TSS. Evaluations of the 1999 growing season water quality monitoring data for each of the transplant sites are presented in Table 2. Median growing season conditions at all of the transplant sites are estimated to meet the habitat criteria (ie. 13% light available through the water column) at water depths of 0.5 m or less. Combined reductions in TSS and Chl a, or TSS only to the specified levels would be required for attainment of one-meter or two-meter habitat criteria. These projected reductions would have to be quite significant. For example at Powell's Creek growing season median TSS and Chl a levels of 38 mg/L and 12.6 µg/L respectively (Table 1) would have to be reduced to 17.2 mg/L and 6.2 µg/L (TSS and Chlorophyll Reduction) to achieve the light conditions estimated for SAV growth to one meter. For TSS reduction alone (TSS Reduction Only) a target of 16.2 mg/L would have to be met. Reductions in Chl a alone, even to zero concentration, are predicted to be insufficient for SAV growth at one and two-meter depths due to the residual turbidity from the suspended sediments.

<u>Light Availability for SAV Growth</u>
Light availability for SAV growth was also calculated using a second empirical model developed by the Chesapeake Bay Program SAV Technical Synthesis II Work Group (Batiuk et al. In press.). This model predicts the percent of incident light available to SAV at specified depths in the water column (PLW), as well as the residual light available for SAV photosynthesis after passage through the predicted periphyton layer on leaf surfaces (PLL). The PLW determination is a function of water column light attenuation (secchi depth or Kd ), water depth and mean tidal range. The PLL determination is a function of PLW as well as water column TSS, DIN and DIP. Seasonal median thresholds or requirements for growth of freshwater SAV are estimated as 13% for PLW and 9% for PLL. For the data presented here it was assumed that DIN was equivalent to dissolved NOx , and DIP was equivalent to orthophosphate. Transplant depth was set at 0.3 m and tidal range at 0.66 m.

Results of the calculations of SAV growing season median PLW and PLL for each of the transplant sites are presented in Table 3. According to this model sufficient light (both PLW and PLL) for SAV growth during 1999 would only be predicted for the Shirley Cove site. This is in contrast to the successful transplant results observed at all locations, suggesting under-prediction of actual light conditions at the sites by the model, lower light requirements of the SAV than estimated, and/or shallower effective water column depth at the leaf surface due to SAV canopy development.

**Table 2. Predicted Concentrations Necessary for Using Several Reduction Strategies**

| Station | Target Depth (m) | Growing Season Median Concentration[1] | | | | | |
|---|---|---|---|---|---|---|---|
| | | TSS and Chlorophyll a Reductions | | TSS Reduction Only | | Chlorophyll a Reduction Only | |
| | | Chla (µg/L) | TSS (mg/L) | Chla (µg/L) | TSS (mg/L) | Chla (µg/L) | TSS (mg/L) |
| Powell's Creek | 0.5 | Met | Met | Met | Met | Met | Met |
| | 1.0 | 6.2 | 17.2 | 12.6 | 16.2 | N/A | N/A |
| | 2.0 | 2.5 | 7.0 | 12.6 | 5.3 | N/A | N/A |
| Tar Bay | 0.5 | Met | Met | Met | Met | Met | Met |
| | 1.0 | 6.7 | 17.2 | 12.0 | 16.3 | N/A | N/A |
| | 2.0 | 2.7 | 7.0 | 12.0 | 5.4 | N/A | N/A |
| Shirley Cove | 0.5 | Met | Met | Met | Met | Met | Met |
| | 1.0 | 11.2 | 16.4 | 13.7 | 16.0 | N/A | N/A |
| | 2.0 | 4.6 | 6.7 | 13.7 | 5.1 | N/A | N/A |
| Turkey Island | 0.5 | Met | Met | Met | Met | Met | Met |
| | 1.0 | 6.0 | 17.3 | 11.1 | 16.4 | N/A | N/A |
| | 2.0 | 2.5 | 7 | 11.1 | 5.6 | N/A | N/A |

**1** 'Met' indicates that no reductions are necessary for SAV growth at that depth; 'N/A' indicates that a 100% reduction in that parameter would still not permit sufficient water clarity for SAV at that depth

**Table 3. Median Growing Season Light Availability for SAV Growth After Predicted Attenuation Through the Water Column (PLW) and Predicted Leaf Periphyton Mass (PLL)**

| **Site** | **PLW** (% of Surface Irradiance) | **PLL** (% of Surface Irradiance) |
|---|---|---|
| Powell's Creek | 8.1 | 5.7 |
| Tar Bay | 11.8 | 6.9 |
| Shirley Cove | 20.1 | 15.6 |
| Turkey Island | 10.4 | 7.4 |

## CONCLUSIONS

### Evaluation of Transplanting Techniques

The transplantation of nursery-grown, bare-rooted, unanchored shoots of wild celery into shallow water areas of the Hopewell region of the tidal James River was successful and marks the first time that SAV has been successfully transplanted into this tidal freshwater region of the James River. Transplantation in late May or early June allows sufficient time for the plants to become established, flower and produce reproductive structures prior to fall die-back. These successful results indicate that SAV transplanting in the Hopewell estuary region of the James River can be successful without damaging the remaining natural bay stocks. The production of the planting stock used here was accomplished non-destructively using field collected seeds from natural beds. The seeds were then germinated and seedings grown under variety artificial conditions, including those designed by the Chesapeake Bay Foundation for use by private citizens and schools.

The transplanting of wild stock of sago pondweed met with less success than the wild celery. This may have been due to the lack of rhizome apical meristems in the source material as well as elongated canopy development that precluded survival in the wave and current conditions of this tidal James River region. This species does occur throughout many low salinity and freshwater regions of the Chesapeake Bay (Orth et al. 1999, Moore et al. 2000). Therefore, its use in further transplanting efforts should be pursued if adequate nursery grown stock can be obtained.

Protection of the transplanted SAV material from herbivory for at least the first growing season appears necessary for adequate survival. Qualitative observations from transplanting efforts with wild celery in the Potomac River as well as other areas in the upper Chesapeake Bay in Maryland suggest that eventually, as the stands of SAV become established and more numerous in an area, the herbivory will decrease. However, flocks of waterfowl or other animals can also cause a significant damage to even established beds.

The use of chicken wire screening to protect the beds met with mixed success. Although it proved successful in protecting the transplants, corrosion reduced it effectiveness after approximately two months. This may have affected the survival rates of the transplants at the Tar Bay and Turkey Island sites where the wire fencing became less effective over time. Further studies should investigate the use of other exclosure materials such as plastic mesh screening, which withstand exposures of one growing season or longer, as protection of transplants from herbivory is imperative.

**Response of SAV to Habitat Conditions**

Current habitat conditions at the transplant sites appear adequate for the successful growth of both wild celery and sago pondweed transplants at the depths planted. Although sago pondweed is more commonly found throughout the bay in areas of low salinity rather than freshwater it also survived, and the shoots elongated and grew until the stems broke. The wild celery was much more successful and demonstrated vigorous growth during the first growing season and there was significant re-growth during the spring of 2000 at both the Powell's Creek and Turkey Island sites. Physical and biological factors appeared important in limiting SAV survival at the shallow water transplant sites. Rapid elongation of shoots to over one-meter in length is one possible mechanism by which these plants may reduce the negative effect of light limitation from the turbid waters. Suspended sediments are the major component of light attenuation in these shallow water sites, although phytoplankton is an important component of light attenuation during the summer. Algae, sediments, and other fouling components on the shoots of the SAV here appeared minor and there was little accumulation throughout the growing season

Results of the evaluation of the water quality conditions in the Hopewell estuary during this study in 1999 are very similar to those of the HERMA study in 1998 (Malcom Pirnie, Inc. 1999) that concluded that there should be sufficient light for SAV growth at depths < 0.5m. That work also suggested that there was significantly greater turbidity in the shallows compared to channel, due in part to re-suspension of sediments by tidal currents as well as wave action that may affect this survival. We did not evaluate this here, although given these successful results the SAV appear to be receiving sufficient light for growth at depths of less than 0.5 m. Development of established bed canopies could help to reduce this turbidity by baffling wave action and reducing re-suspension

These transplant sites were chosen to provide the best sediment substrate possible (ie. low organic content) for SAV growth. However, sediments in much of the surrounding shallow water areas are comprised of very soft mud. The survival of SAV in this substrate type is unknown. In any event, transplanting SAV by the methods used here would be very difficult in soft sediments that are easily stirred up and offer no support. There are other SAV species *(Ceratophyllun demersum, Myriophyllum demersu*s) that can grow in muddy substrates, but *C. demersu*m, in particular, because of its lack of root material does not withstand strong currents or wave action. During the year 2000, transplants locations will include areas with more organic-rich substrates.

**Evaluation of SAV Habitat Quality Models**

The concurrent measurements of water quality and SAV growth and survival at these transplant sites provided the opportunity to evaluate several different SAV habitat quality models for their effectiveness in predicting SAV growth in the Hopewell estuary region. Results of the first model (Table 1), which was developed as part of the CBP SAV Technical Synthesis (Batiuk et al. 1992), suggest that total suspended solids and light attenuation should be insufficient for SAV growth to one-meter depth but that Chl a levels should be sufficient. While we could not evaluate that here directly, we did show that persistent growth at shallower depths even to 20 cm or less is possible and that the shallowness of growth may only be limited by exposure during extreme low tides. The use of growing season medians (<15 µg/L) greatly under-represented the high Chl a levels (>50 µg/L) observed during the summer. Since SAV have been demonstrated to respond negatively to extreme conditions over time scales of less than a single growing season, the use of median values may be an underestimate of the true environmental stress at water depths greater than those planted here.

The Gallegos' model which predicts growth at depths other than one meter using TSS and Chl a data (Table 2) was successful in predicting SAV transplant survival at all sites at a target depth of 0.5 m or less. Nearly a 50% reduction in seasonal median TSS levels, however, is predicted to be required for SAV growth to one meter. Given the ambient TSS levels, even complete removal of Chl a is predicted to be insufficient for SAV growth to one meter or greater. As with the previous SAV Habitat Requirements' model, the use of seasonal medians underestimates the potential impacts of the high Chl a observed here during the summer. Reductions of Chl a during this period would result in much greater increases in available light than predicted for the growing season as a whole.

Finally, the calculations of PLW and PLL by the third model did not predict the successful growth of SAV at any of the transplant sites, except Shirley Cove, at the transplant water depth of 0.3 m MLW. As previously mentioned this may have been due to the rapid SAV canopy development and shoot elongation that diminished the effective water column over the plant leaves. Once having grown to one meter in length the leaves were very close to the water surface during low tides. It may be that adequate light can be obtained during these periods to sustain growth. The plants may also require less than the predicted 9% of incident light at the leaf surface for growth. Experimental studies of wild celery suggest that this may be so. PLL predicted by the model for the three sites ranged from 5.7 to 7.4 %, which may be adequate levels for these SAV species. Finally, the model may underestimate the actual light available to SAV at the leaf surface by overestimating light attenuation, especially through the periphyton layer. The fouling estimates made in this study indicate that in this freshwater region substrate fouling is quite low. For example periphyton typically accumulated to levels of 0.02 mgdw $cm^{-2}$ or less throughout the growing season. In contrast model predictions of epiphyte loads using growing season medians are 1.29, 1.80, 0.90, and 1.18 mgdw $cm^{-2}$ for the Powell's Creek, Tar Bay, Shirley Cove and Turkey Island transplant sites, respectively.

These comparisons suggest that habitat requirement models of SAV growth are only some of the tools that should be used for general guidance, not absolute predictors, in the evaluation of suitable habitat conditions for SAV growth, especially in very shallow water conditions. In addition to water quality measures, biological and physical as well as sediment substrate factors should also be considered. In most cases, overall habitat conditions in a region such as this may only be effectively evaluated through actual transplantation studies that are repeated under multiple years of varying climatic circumstances.

The results of this study are a promising start for continued investigations of SAV restoration in the Hopewell estuary region. Future investigations on the effects of substrate type, herbivory, as well as studies of habitat effects on other SAV species will enhance the probability of success of larger scale transplant efforts. Such information will be useful in management of the region for the enhancement of SAV re-colonization.

## { TC"5.0 LITERATURE CITED" \l1 }REFERENCES

Batiuk, R.A., Orth, R.J., Moore, K.A., Dennison, W.C., Stevenson, J.C., Staver, L.W., Carter, V., Rybicki, N.B., Hickman, R.E., Kollar, S., Bieber, S., Heasly, P. 1992. Submerged Aquatic Vegetatio Habitat Requirements and Restoration Targets: a Technical Synthesis. USEPA. Chesapeake Bay Program, Annalpolis, Md. 186 pp.

Barko, J.W. and Smart, R.M. 1983. Effects of organic matter additions to sediment on the growth of aquatic plants. J.of Ecol. 71: 161-175.

Carter, V. and Rybicki, N.B. 1985. The effects of grazers and light penetration on the survival of transplants of *Vallisneria americana* Michx in the tidal Potomac River, Maryland. Aquat. Bot., 23: 197 213.

Malcolm Pirnie, Inc. 1999. Hopewell Estuary Region Monitoring and Assessment Project. Report of 1998 Results. Malcolm Pirnie, Inc. Newport News, Va.

Moore, K.A., Wilcox, D.J., Orth, R.J. 2000. Analysis of the abundance of submersed aquatic vegetation communities in the Chesapeake Bay. Estuaries, 23: 115-127

Neckles, H.A. 1990. Relative effects of nutrient enrichment and grazing on Epiphyton-macrophyte (*Zostera marina*) dynamics. Ph.D. Dissertation. College of William and Mary, Gloucester Point, Va. 105 pp.

Orth. R.J., Nowak, J.F., Wilcox, D.J., Whiting, J.F., L.S. Nagey. 1999. Distribution of submerged aquatic vegetation in the Chesapeake Bay and Coastal Bays – 1998. Virginia Institute of Marine Science. Gloucester Point, Va. 370 pp.

Robinson, G.G.C. 1983. The key to understanding periphyton. In: R.G. Wetzel (ed), Periphyton of Freshwater Ecosystems. Dr. W. Junk Publ., The Hague, pp. 63-71.

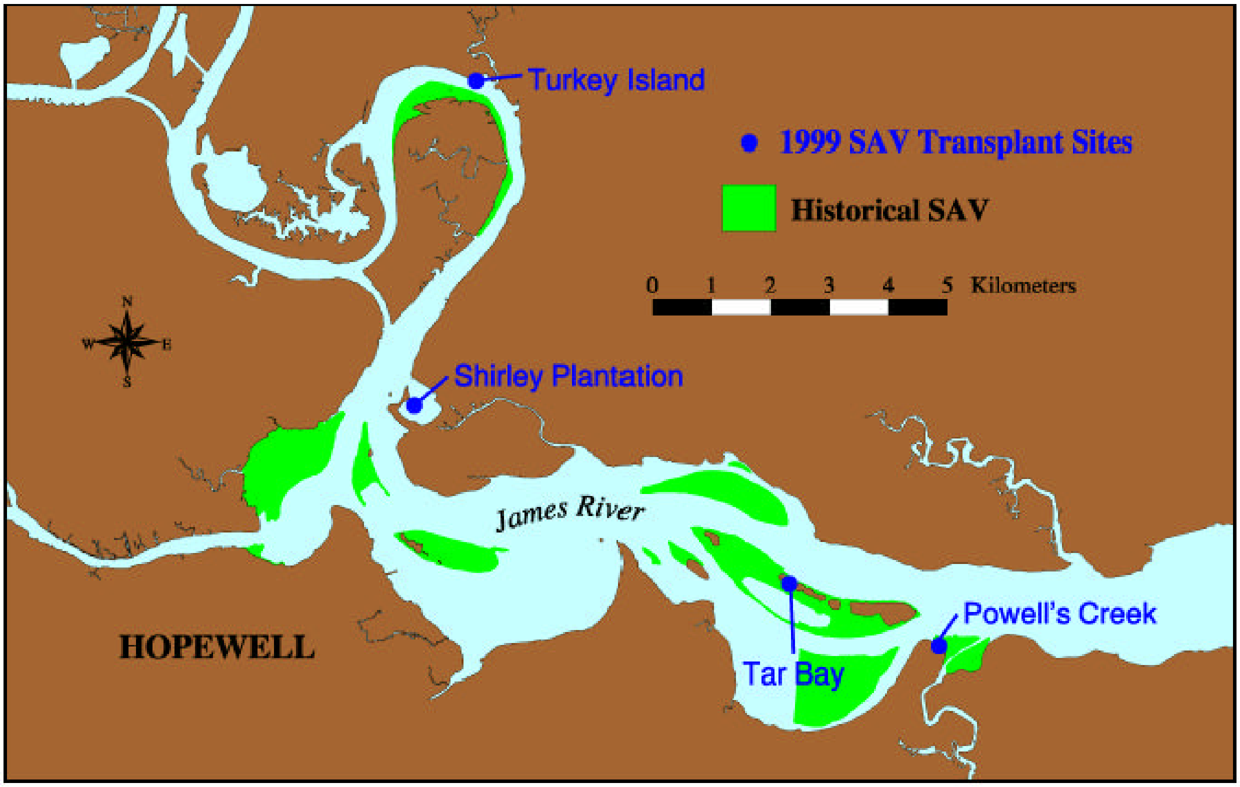



**Figure 1: Location of SAV Transplant Sites**

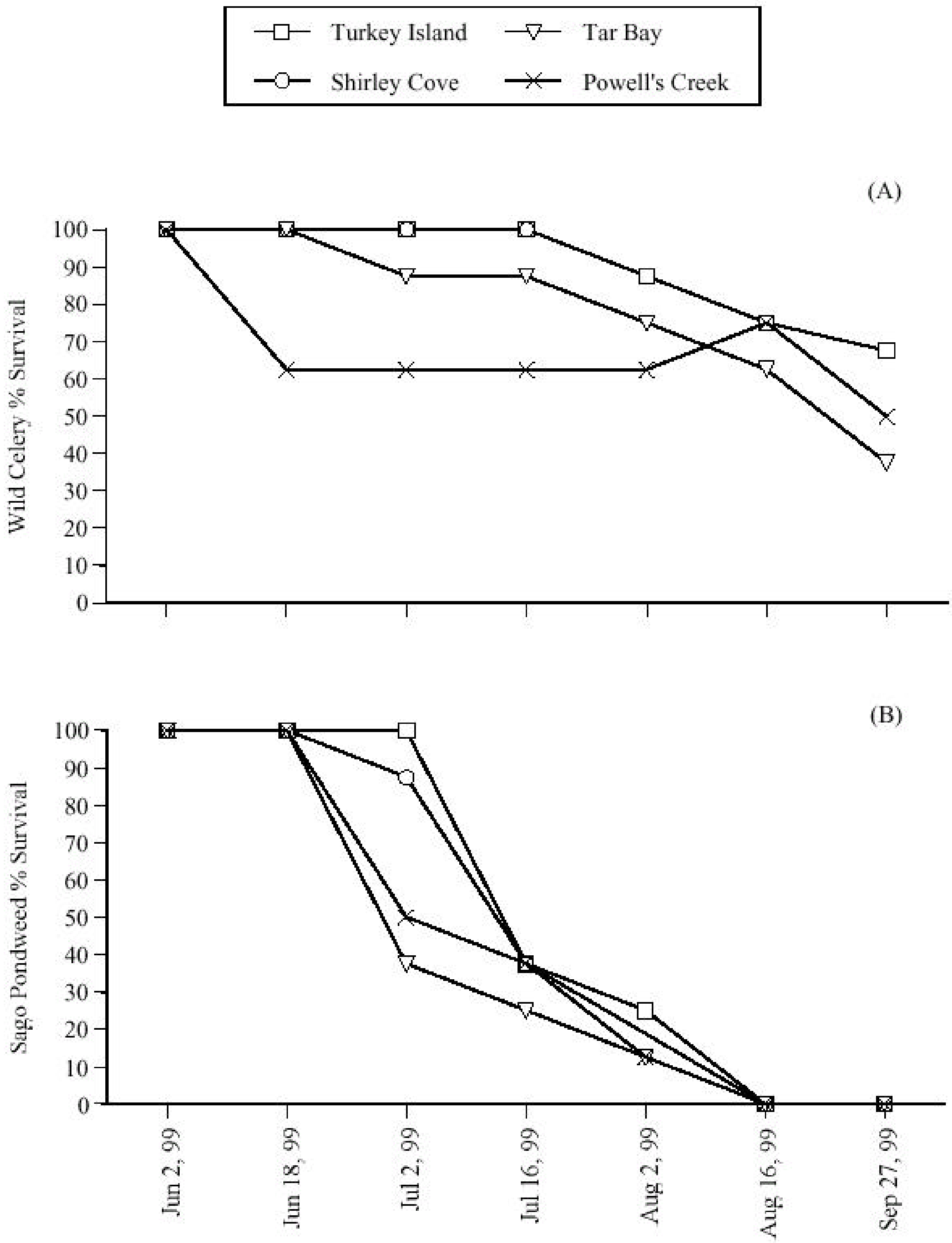



**Figure 2: Wild Celery (*Vallisneria americana*) and Sago Pondweed (*Potamogeton pectinatus*) Survival**

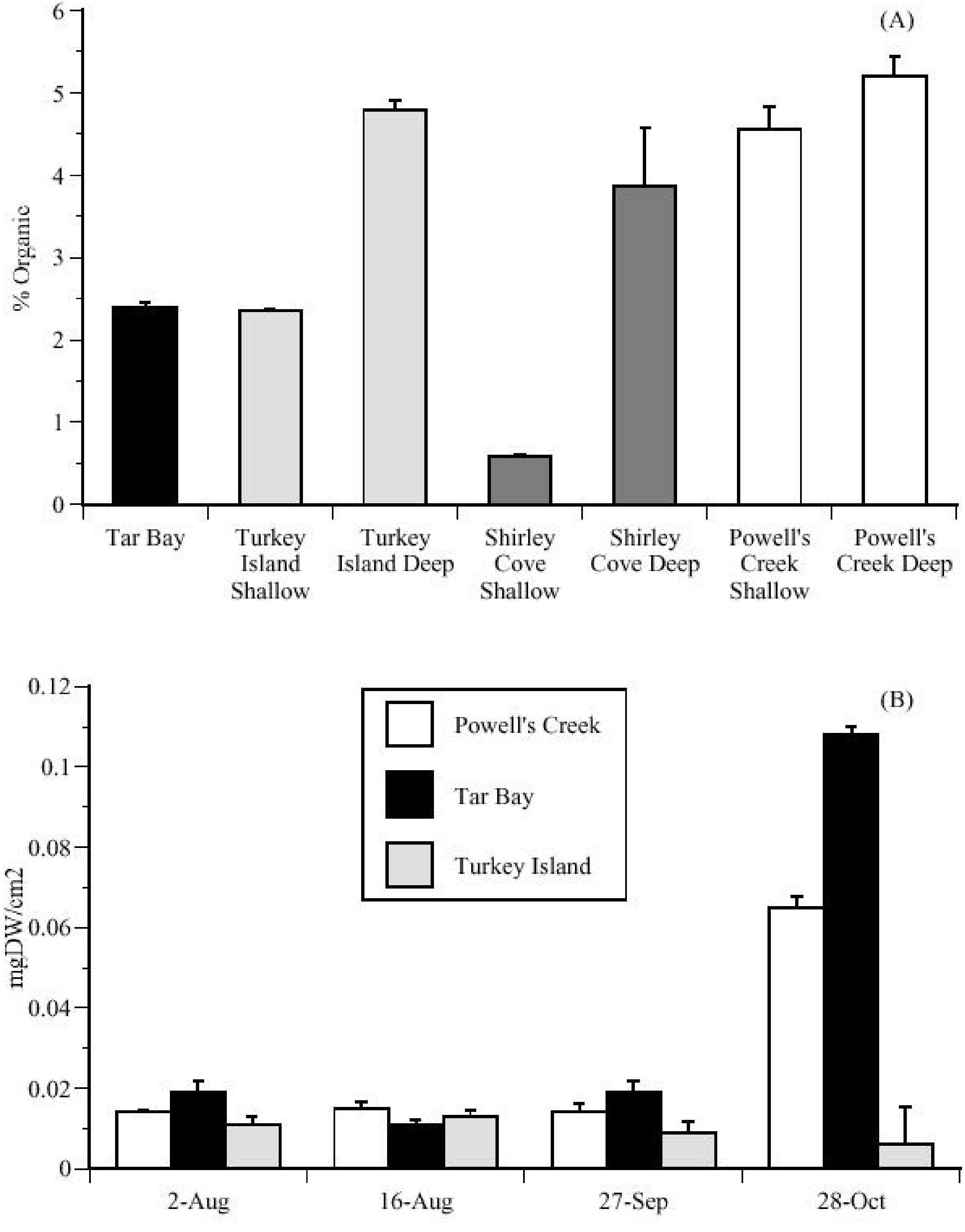



**Figure 3: Sediment Organic Content (A) and Periphyton Dry Weight on Artificial Substrates (B) at Transplants Sites**

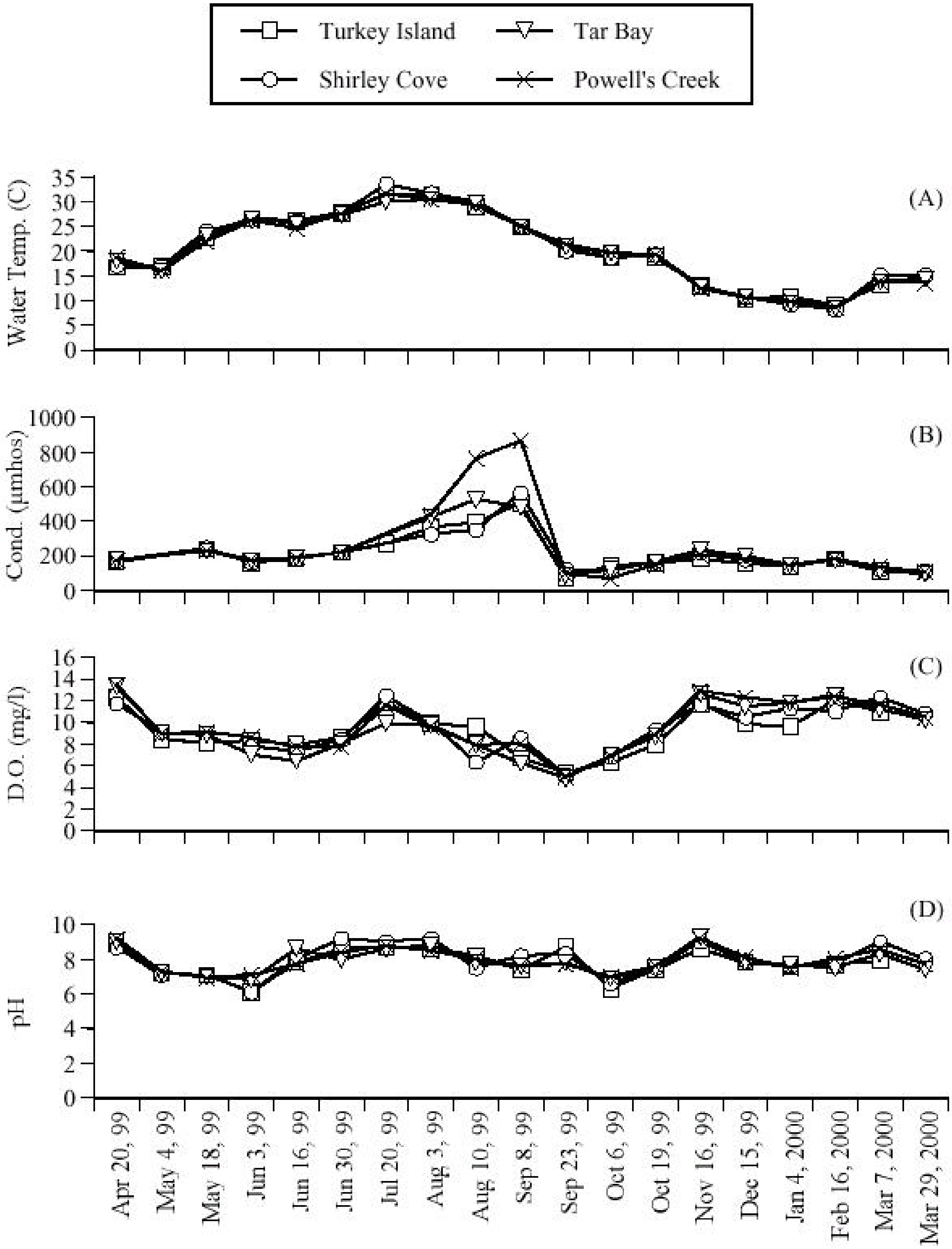



**Figure 4: Water Column Measurements of Temperature, Conductivity, Dissolved Oxygen, and pH**

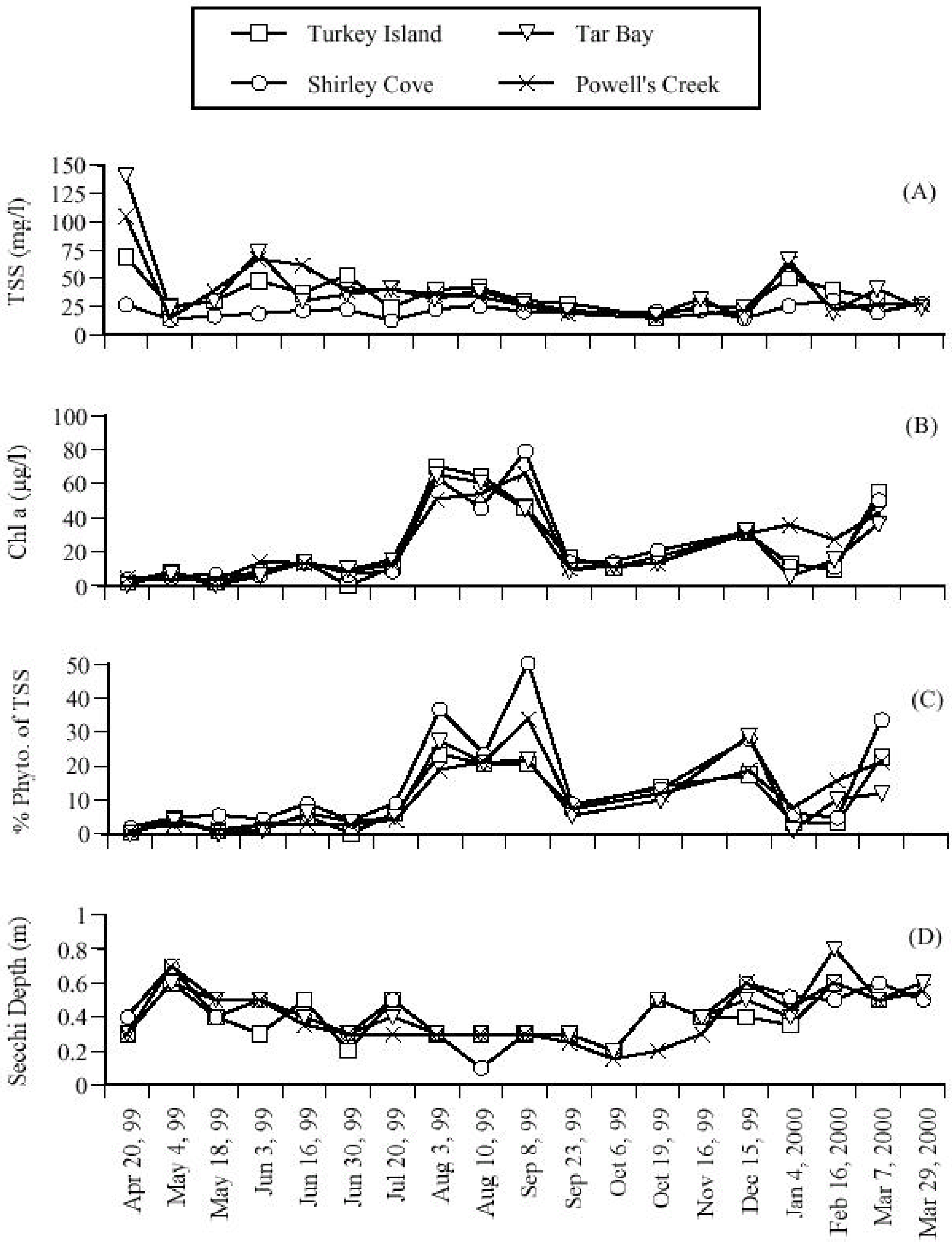



**Figure 5: Water Column Measurements of Total Suspended Solids, Phytoplankton as Chlorophyll a, % Phytoplankton Component of TSS, and Secchi Depth**

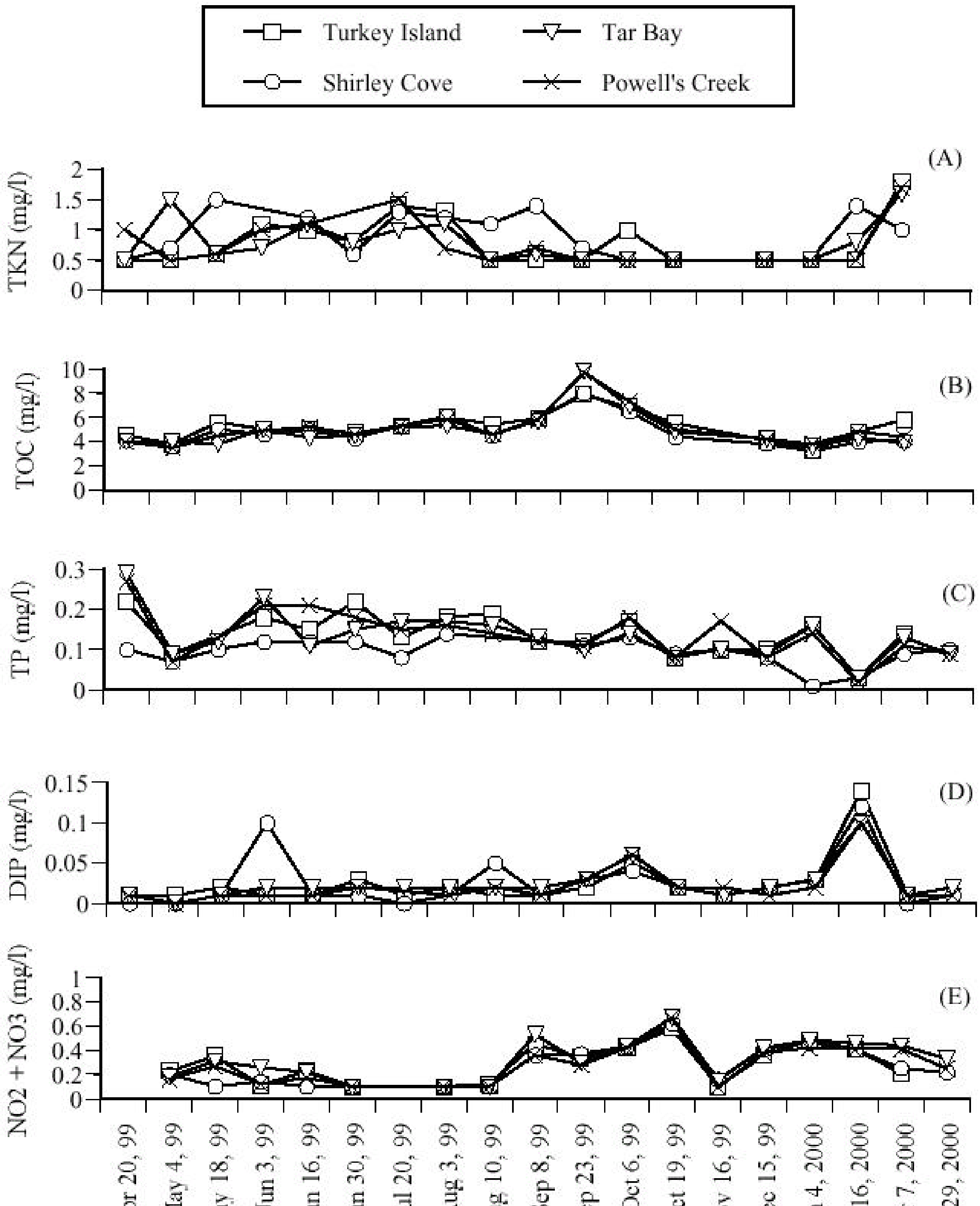



**Figure 6: Water Column Measurements of Total Kjeldahl Nitrogen, Total Organic Carbon, Total Phosphorus, Dissolved Inorganic Phosphate, and Nitrite + Nitrate**

# Lake Water Clarity Assessment of Minnesota's 10,000 Lakes: A Comprehensive View from Space

Leif G. Olmanson , Patrick L. Brezonik, Steve M. Kloiber , Marvin E. Bauer, Erin E. Day

Water Resources Center
University of Minnesota
St. Paul, Minnesota 55108

E-mail: Olman002@tc.umn.edu

**Keywords:** Lake water clarity, lake water quality, assessment, Landsat, remote sensing

## ABSTRACT

This paper describes development of regional-scale assessment techniques for lake water clarity assessment using Landsat TM satellite imagery. In addition to an image-processing and assessment protocol, a lake water-clarity database was created for more than 500 Twin Cities (Minneapolis and St. Paul, Minnesota) Metropolitan Area (TCMA) lakes using 14 images from 10 years spanning the period 1973 to 1998. This database was used to assess spatial patterns along with temporal and seasonal trends in lake water clarity within the TCMA. The image-processing and assessment protocol also was used to study the cumulative impacts of land development on lakes. In this work we assessed the impacts of land development on lake water clarity for ~300 lakes in the Alexandria Lakes Area of west-central Minnesota, as well as lakes in the TCMA. Landsat data were used to estimate lake water clarity because ground-based data exist for only a small fraction of lakes in the two regions. A geographical information system (GIS) was used to link the lake clarity data with land-use features. The procedures for satellite image analysis recently were enhanced so that they can be applied efficiently to the much larger number of lakes (many thousands) across the Upper Midwest for statewide lake water clarity assessments in Minnesota, Wisconsin and Michigan. Difficulties, encountered in expanding the procedure to this broader geographic scale, such as working with satellite data from remote areas where field data (used to calibrate the Landsat data) are scarce or not available, will be discussed, and methods to circumvent these problems are described.

**Keywords:** Lake water clarity; lake water quality; assessment; Landsat; remote sensing

## INTRODUCTION

Minnesota is fortunate to have a large number of lakes (approximately 12,000 lakes >10 acres). These lakes are important recreational and aesthetic resources that add to the economic stability and quality of life. Protecting and monitoring lake water quality is a major concern for many local and state agencies. However, because of expense and time requirements for ground-based monitoring, it is impractical to monitor more than a small fraction of this large resource by conventional field methods. The use of remote, satellite-based sensing is a potentially cost-effective way to gather the information needed for regional water quality assessments in lake-rich areas like Minnesota.

For effective environmental planning and management, it is vital to have long-term water quality information on a broad regional scale. It is not possible to go back in time and collect additional water quality information using conventional field methods to fill gaps from previous efforts. Landsat data have been collected regularly since the early 1970s allowing the possibility of extracting historical water quality information from Landsat images. The extraction of historic and current water quality data from satellite images, coupled with existing data collection efforts may facilitate the development of comprehensive regional databases that can be used to evaluate regional differences and water quality trends over time. If used along with land-use data, this information can help determine the impacts different land-use practices have on lake conditions. Results of such analyses can aid local and state agencies to make informed decisions about development policy and improve the management of lake resources.

Although several satellite remote sensing systems have been used for water quality assessment, the relatively low cost, temporal coverage, spatial resolution, and data availability of the Landsat system make it particularly useful for assessment of inland lakes. Several studies have demonstrated a strong relationship between Landsat Multispectral Scanner (MSS) or Thematic Mapper (TM) data and ground observations of water clarity and chlorophyll *a* (Brown et al. 1977, Lillesand et al. 1983; Lathrop and Lillesand 1986; Lathrop 1992; Cox et al. 1998). However, use of satellite remote sensing for water clarity assessments as a routine application has moved slowly. This may be due to a lack of familiarity among inland aquatic scientists with remote sensing technology and a perception that the data are expensive and difficult to process. In 1999 with the launch of Landsat 7, data distribution was returned to the public sector, and the cost of data acquisition dropped significantly. Along with today's powerful desktop computers and sophisticated software, processing and analysis of satellite imagery has become relatively inexpensive and easy to perform.

## METHODS

This section describes methods for applying Landsat imagery to regional-scale and statewide assessment of lake water quality. The methods were developed from work initiated by Olmanson (1997) and continued in subsequent studies conducted by the University of Minnesota Remote Sensing Laboratory and Water Resources Center. The image processing procedures were developed specifically for Minnesota, but with appropriate modifications these procedures should work equally well for other regions. The procedures may continue to evolve as the work moves from pilot scale to full implementation.

### Lake Reference Data

The principal objective is to perform regional assessments of lake water quality, in particular to estimate variables related to key management indicators, such as the trophic state indices of Carlson (1977). The three water quality variables that have been most commonly used to indicate trophic state are total phosphorus (TP), chlorophyll *a* (chla), and Secchi disk transparency (SDT). Measurements of these water quality variables, along with various transformations such as the trophic state indices (TSIs), have been widely used by lake management agencies and organizations.

Secchi disk transparency, a measure of water clarity, is strongly related to satellite spectral-radiometric observations of lakes. For lakes whose clarity is dominated by phytoplankton abundance, chlorophyll will be highly correlated with the satellite observations. Because phosphorus is the limiting nutrient for most lakes, it is generally correlated with chlorophyll in multi-lake analyses, but it is not directly measurable by optical instruments. Moreover, there are enough exceptions and deviations in the relationships between TP, chla, and SDT that we do not recommend estimating TP values for lakes from Landsat data.

The availability of lake reference data varies both over time and spatially within any project area. Ideally, there should be at least 20 ground observation points for each image, and they should be distributed over a wide range of water quality/clarity. Contemporary studies usually can be designed to meet this criterion, but collecting the necessary ground observation data in remote areas may be difficult. In retrospective analyses, one must work within the confines of historical sampling programs. SDT is the most consistently collected trophic state indicator, and it is strongly correlated with Landsat data (Olmanson 1997, Kloiber et. al. 2000). Because SDT is inexpensive to measure and can be determined reliably by volunteer monitoring programs, it is likely that these measurements will be collected on a regular basis throughout the Midwest and other regions.

Most of our research to date has involved calibrating Landsat TM data with ground-based SDT measurements and inferring SDT for all lakes in an image from the regression equation developed in the calibration step. The results then can be mapped directly as distributions of SDT in the lakes, or the estimated SDT can be converted to Carlson's (1977) trophic state index based on transparency TSI(SDT):

$$TSI(SDT) = 60 - 14.41 \ln(SDT)$$

It is important to recognize that other factors aside from algal turbidity (as indicated by chlorophyll levels) may affect SDT in lakes. Most important of these (non-trophic-state) factors are humic color and non-algal turbidity (including soil-derived clays and suspended sediment). For this reason, it is preferable to report Landsat results based on SDT calibrations directly as satellite-estimated SDT or to use the explicit term, TSI(SDT), which clearly identifies the value as an index based on transparency, rather than the generic term, TSI.

## Satellite Image Data

Images selected for water quality assessment were high-quality, cloud-free images selected from a late summer index period (which is discussed in the next section). Images were registered to the Universal Transverse Mercator (UTM) coordinate system using the NAD 83 datum and subsequently resampled using nearest neighbor. Careful selection of approximately 40 well-defined, and well-distributed ground control points (GCPs) resulted in positional accuracy (RMSE) equal or better than ± 0.25 pixels, or 7.5 meters. We found the Minnesota Department of Transportation highway map (available in ArcInfo GIS format) provides an efficient and effective means of easily finding a large number of well-distributed GCPs. Atmospheric correction or normalization of the Landsat data is not necessary for the regression method described. However, it may be necessary if *in-situ* data are not available for a particular scene.

## Classification Procedures

This section summarizes image classification procedures; more detail is provided by Olmanson et al. (2000), and the rationale for the procedures was described by Kloiber et al. (2000). ERDAS Imagine and ArcView were used for the image processing steps. One of the most important steps is acquiring a representative image sample from each lake. Ideally, the sample should represent the center portion of the lake in at least 5 m of water (or twice the SDT measurement), where reflectance from vegetation, the shoreline, or the lake bottom will not affect the spectral signature.

The first step was to make a "water-only" image by running an unsupervised classification in ERDAS Imagine. Because water is very different spectrally from terrestrial features, water is put into one or more distinct classes that can be easily identified. Terrestrial features then were masked creating a water-only image. A second unsupervised classification was performed on the water-only image. Average brightness values from the unsupervised classification of this image were graphed to show spectral signatures of each class. These signatures were used to differentiate classes containing clear water, turbid water, and shallow water (where sediment and/or macrophytes affects spectral response). Based on this information classes were re-colored so that vegetation, bottom and terrestrial effects could be avoided when selecting lake sample locations or areas of interest (AOI). AOIs are the locations where brightness values from the Landsat image were obtained to develop relationships with measured SDT. AOIs were delineated by a polygon layer to help automate the process. ERDAS Imagine's signature editor was used to extract the spectral data from the image for each AOI.

All available SDT data collected within a prescribed number of days of the satellite image were used for calibration purposes. A data set that includes 20 or more ground observations per image, spanning a wide range of ambient conditions was considered acceptable. A multiple regression was performed using log-tranformed SDT data as the dependent variable and Landsat Thematic Mapper Band 1(TM1) and the TM3:TM1 ratio as independent variables. Based on the relationship from the regression model, lake water clarity maps were created by two methods. The first method used the ERDAS modeler to create a pixel-level lake map (Figure 1). With this map all water pixels are classified and intra-lake variability can be evaluated. The other method used a spreadsheet program to calculate water clarity for each lake. The data were then linked to a lake polygon layer in ArcView or another GIS program to create a lake-level water clarity map (Figure 2).

## EXAMPLE APPLICATIONS

### Twin Cities Metropolitan Area Lake Water Clarity

Our initial work focused on the seven-county Twin Cities (Minneapolis and St. Paul) Metropolitan Area (TCMA), a 7,800 km$^2$ area that contains more than 600 small and medium-sized lakes. This study focused on developing and applying an image-processing and assessment method. Method development issues that were addressed included defining timing and frequency requirements for an assessment program, selecting satellite images, data extraction from images, selecting ground observation data, and developing an appropriate standardized regression model. To date, ten Landsat Thematic Mapper (TM) images and four Multi-Spectral Scanner (MSS) images, spanning the period 1973-1998 have been analyzed. The resulting regional database of water clarity for more than 500 lakes over the 25-year period is being used to assess spatial and temporal trends in water clarity.A large database of chla and SDT measurements on TCMA lakes was analyzed to determine the consistency of seasonal patterns among lakes in the region (Anderle 1999; Brezonik et al. 2000; Anderle et al., submitted) and evaluate the reliability of inferences made from a few measurements (1-3) in a growing season. The database included 370 "lake-years" of measurements on 145 lakes obtained between 1986 and 1997. Distinct patterns were found for both variables, and both fit a sine function (Anderle et al, submitted). SDT starts high in spring and early summer and decreases to a minimum in mid to late summer, followed by an increase in early fall. The opposite pattern was observed for chla. Strong relationships were observed between the growing season mean SDT and minimum SDT and between growing season mean chla and maximum chla. An index period of late July to early September was found to be the optimal time for measuring trophic conditions in TCMA lakes (Anderle et al., submitted). During his period temporal and in-lake variability is at a minimum for SDT. Statistical analysis of these data indicated that 1-3 observations within the index period are sufficient to estimate the growing season mean SDT to within 30%.The successful use of satellite data to assess water clarity depends on the quality of the image data. Although instrument problems, such as banding and striping can hinder the use of satellite data, these problems were minimal or non-existent in the images we have examined. A more common problem is the effect of cloud cover and cloud shadows obscuring features of interest. Because of this, only cloud free images (< 10% cloud cover) were used. This criterion significantly reduced the pool of images suitable for analysis, but for most years there was at least one late-summer image that met this criterion (Kloiber et al., 2000). From the pool of available images we selected ten Landsat TM images and MSS images of the Twin Cities Metropolitan Area from 1973-1998. Images were selected from years with relatively normal climatic conditions except for one dry year (1988) and one wet year (1993). Four images from the same year (1991) were selected to analyze seasonal variability, and TM and MSS images from the same day (September 4, 1991) were acquired to evaluate the compatibility of the two sensors for water clarity assessments.

Spectral data for TCMA lakes were extracted based on AOIs that were delineated for each lake in a way that avoided interference from mixed land-water pixels, macrophytes, and bottom reflectance. The number of pixels used in the AOI may affect the strength of the relationship between the satellite image and the ground observation data, and we used a range of pixels from the profundal zone of each lake, similar to the method recommended by Lillesand et al. (1983). This approach was tested by experimenting with the size of the AOIs for a consistent subset of lakes. The results suggest at least nine pixels should be used for AOIs, but the marginal benefits of increasing the AOI size is minimal beyond 25 pixels (Kloiber et al., submitted).

Another factor potentially influencing the strength and reliability of ground-to-satellite relationships for water clarity is length of time between ground and satellite observations. To address this issue a series of regression analyses were conducted for a Landsat TM image from July 29, 1995 (Kloiber et al., submitted). As expected, increasing the time window from $\pm$ 1 to $\pm$ 7 days decreased the statistical fit (i.e., decreased the $r^2$ and increased the standard error of the estimate). For the TCMA, where lake clarity data are fairly abundant, we found that a window of $\pm$1 day between satellite and ground observations was optimal for our ground observation data set. However, for regions with fewer available ground observations, a window of up to $\pm$ 10 days may be necessary and still should yield sufficient accuracy.Multiple regression analysis was used to evaluate the relationship between Landsat spectral data and ground observations of SD. The results show that the ratio of TM3:TM1 plus TM1 or the ratio of MSS2:MSS1 plus MSS1 provided strong predictive relationships for multiple images over a 25-year period (Kloiber et al. 2000; Kloiber et al., submitted). The regression coefficients were uniquely derived for each image, but in general, the coefficients were similar in magnitude for the various images. Other investigators have found similar relationships (Lathrop 1992; Lavery et al. 1993; Pattiaratchi et al. 1994; Cox et al. 1998; and Day 2000) between SD and Landsat imagery across a broad geographic range (Australia, Wyoming, Minnesota Wisconsin, and the Southeastern U.S.) and types of water bodies (estuaries, reservoirs, and inland lakes). This suggests that there is a physical basis for the relationships, and indeed the spectral characteristics of algae and other aquatic particles correspond to the spectral bands used in these equations (Koiber et al., submitted).

The lake clarity data (expressed as TSI(SDT)) derived from this study were analyzed to determine whether significant water clarity trends exist for TCMA lakes. Of the lakes in the study, 482 lakes had at least five satellite based TSI(SD) measurements over the 25-year period. Sixty-four of the evaluated lakes had temporal trends. Water clarity increased for 33 lakes and decreased for 31 lakes. Over half (58%) of the lakes with apparent trends changed by 10 or more TSI units (equivalent to a doubling or halving of SDT) from 1973 to 1998.

**NCHF Ecoregion – Cumulative Impacts of Land Development on Lake Water Clarity**

The image-processing protocol also has been used in a study of cumulative impacts of land development on lakes (Brezonik et al. 2000). In this work we assessed the impacts of land development on lake water clarity for ~300 lakes in the Alexandria Lakes Area (ALA) of west-central Minnesota, as well as lakes in the TCMA. Landsat data were used to estimate lake water clarity because ground-based data exist for only a small fraction of lakes in the two regions. A geographical information system (GIS) was used to link the lake clarity data with land-use features The two areas are mostly within the North Central Hardwoods Forest (NCHF) ecoregion. The ALA represents lakes primarily in a rural setting, and much of the development around its lakes is in the form of resorts, cabins and lake homes. The area includes Otter Tail, Grant, Douglas, Stevens, Pope and the southwestern portion of Todd Counties. The seven-county Twin Cities (Minneapolis and St. Paul, Minnesota) Metropolitan Area (TCMA) represents lakes in a mostly urban/suburban setting and includes Anoka, Carver, Dakota, Hennepin, Ramsey, Scott and Washington Counties.

Landscape-level variables that may be related to nutrient transport from watersheds to lakes can be organized into two major categories: *land-use/cover* (e.g., forest, open, agricultural, urban) and *land use intensity* (e.g., human density, animal density). These categories were used in the analysis. Because of the large amount of data needed to assess cumulative impacts over large areas, we limited our search to data available in digitized format. A Geographical Information System (ArcView 3.1 on a Windows NT platform) was an essential tool in developing the database. The land-use/cover data for the ALA was a map developed by the International Coalition principally from ca. 1990 aerial photography. The map had 15 land use/land cover classes including: urban and industrial; farmsteads and rural residences; rural residential development complex; other rural development; cultivated land; transitional agricultural land; grassland; grassland-shrub-tree; deciduous forest; coniferous forest; water; wetlands; gravel pits and open mines; exposed soil, sandbars, and sand dunes; and unclassified. For the TCMA we created a hybrid land cover map with imperviousness estimated from a 1991 Landsat TM image (Doyle, Bauer and Olmanson unpublished). This map had nine land use/land cover classes including: non-row crop; row crop; small grain crop; forest; grassland; fallow; wetland; water; and urban land divided into 16 classes of imperviousness. Land-use intensity data consisted of (i) 1990 human population density by census block group from the U.S. Census of Population and Housing, and (ii) 1992 animal unit density by zip code from the U.S. Census of Agriculture. Population density varies greatly between the TCMA and ALA. Although not all of the TCMA is urbanized, even the rural areas of this region tend to have higher population densities than most of the ALA.

We used two types of lake data to relate landscape stressors with in-lake response. One type included morphometric variables that affect lake sensitivity to impacts from a given amount (e.g., surface area, mean and maximum depth, shore length). Lake water specific conductance was used as a surrogate for information on watershed hydrology and geology. The other type included "response variables" associated with lake trophic state: Secchi disk transparency, chlorophyll a, and total phosphorus. Other indicators of impacts from development, such as shoreline habitat, noise/solitude, and crowding, are not measured routinely on lakes, and thus we limited our analysis to the simple measures of trophic state.

The initial set of lakes selected in each area was obtained from the Division of Water (DOW) data warehouse of the Minnesota Department of Natural Resources. A total of 1445 ALA lakes and 573 TCMA lakes were in the DOW's lake list. However, only 270 ALA lakes and 383 TCMA lakes had been surveyed by the DNR and had morphometric data in the DOW database. These lakes were used as our study lakes.

There was a general paucity of lake data, and the availability of data varied according to location and variable. Due to this lack of lake data we inferred TSI(SD) from 1991 Landsat images for all of our study lakes. Figure 3 shows the Landsat inferred water clarity for the ALA lakes. This period was selected to correspond with the available landscape stressor data. For the TCMA we used a September 4, 1991 Landsat 5 Thematic Mapper image that had been previously processed in our TCMA project. For the ALA we used an August 17, 1991 Landsat 5 Thematic Mapper image that was processed using the procedures discussed in the methods section.

Initially, we assumed we would analyze landscape stressor-lake response using watershed-level information for stressor variables. Watersheds have been recognized as the appropriate landscape unit for water and nutrient budget analyses on lakes for many decades. The lack of delineated watersheds and hydrologic information for most lakes severely limited our options, but because we were interested in relating landscape development to lake conditions rather than relating specific substance loading (input) rates to water quality, the lack of delineated watersheds and hydrologic data became less critical. Instead of watersheds, we related landscape development factors to lake conditions using the concept of a landscape buffer (the land area surrounding each lake within a defined distance from the shoreline). Two buffer distances were used: (i) a 150-m buffer to represent shoreline development and (ii) a larger, variable-width buffer (determined by lake size) to approximate watershed development. The width of the latter buffer varied with lake area in the following way: 10 acre, 0.3 km; 100 acre, 1.0 km buffer; 1,000 acre, 1.7 km. Figure 4 illustrates the relative sizes of the 150-m and variable-width buffers for two ALA lakes near Glenwood, Minnesota. Land use and land cover characteristics of the variable-width buffers are illustrated in the figure, but the map scale is too coarse to show this information for the 150-m buffer.

For analyses of all lakes from both areas, the land use/cover classes were merged into two broader classes: *developed land* (all developed classes: urban and industrial; farmsteads and rural residences; rural residential development complex; other rural development for ALA and urban for TCMA) and *cropland* (all croplands: cultivated land; and transitional agricultural land for ALA and non-row crop; row crop; and small grain crop for TCMA). The average percent developed land was considerably higher for the lake buffers in the TCMA than for the ALA, but the opposite was true for cropland. However, a wide range of each class occurred in both sets of buffers in each region.

The first step of the analysis involved compiling the data into a single database that contained all the lakes and the corresponding data for the land uses and population measures for the two buffer zones we defined for each lake. The next step searched for general trends using the master database with all study lakes (both TCMA and ALA) by running simple linear and multiple regressions using TSI(SD) as the dependent variable and all population and land-use variables as independent variables. Next, the same regressions were run on the ALA and TCMA lake separately. This was done to determine what differences there might be between similar lakes that are surrounded by contrasting land cover/uses. Subsequently, the data sets were divided into lake classes based on a tripartite (area, depth, conductivity) classification system (Table 1) (Brezonik et al. 2000). After completing these analyses, it became obvious that agriculture had a great effect on lake clarity and was potentially masking out the extent to which some of the variables were affecting TSI(SD). To minimize the influence of agriculture on the analysis, we separated subsets of lakes with little influence from agricultural activities from the rest of the lakes in the data sets. We also examined lakes of similar area and conductivity but varied the depth class to explore the role of depth as a mitigating factor in trophic conditions.

Simple regression analyses performed on the pooled data sets of the TCMA and ALA lakes showed only very weak correlations (Table 2), and no single variable related to landscape development was a strong predictor of lake water clarity/trophic state ($r^2$ generally < 0.1). Water clarity of lakes in the NCHF ecoregion thus cannot be attributed to any single development-related factor (at least among those for which we had data). The contrasting land uses in the two regions provide at least a partial explanation for the poor fit of the pooled data. Agricultural land uses and development (both are changes in natural ground cover) combined had the most significant relation to water clarity for all lakes; these are most likely the two most influential factors for lake water clarity (and by inference, trophic state) in the region.

Some relationships were slightly stronger for the separated areas (TCMA and ALA), but in general the simple regressions still explained only a small portion of the variance in water clarity. Stronger regressions were found (Tables 3,4) when the lakes were sorted by depth/area/conductivity class (Table 1). In this sense, more (i.e., greater sample size) is not always better in exploratory statistical analyses. Classifying lakes into smaller but more homogeneous groups (based on depth, size, etc.) led to better correlations between landscape (stressors) and lake (response) variables.

The large scatter in the simple regressions and complexities of the multiple regressions make it difficult to determine whether there are any threshold levels for impact of landscape development on lake clarity. For similar reasons, it was not possible to use the results to infer "critical levels" of landscape development, above which impacts are considered unacceptable. A few general statements can be made about the most significant stressors of lake clarity in the NCHF ecoregion. Overall, forested land had the strongest correlation with water clarity; decreasing forested land was associated with decreasing water clarity in nearly all the lake subsets. The lack of strong statistical relationships (i.e., high $r^2$ values) is not surprising when one realizes that lakes are complicated ecosystems and that in addition to land-use conditions, numerous other landscape, hydrologic and morphometric factors control lake water clarity.

### Minnesota statewide lake water clarity assessment

As part of a NASA-funded inter-state initiative for the Upper Great Lakes region, we have begun to classify the state's ~9000 lakes that are 20 acres or larger. We acquired 16 Landsat images from a late summer index period for the early 1990s covering the entire state. The images are of high quality, and only a few images have any cloud cover. Because image overlap resulted in many lakes being covered by more than one image, virtually all inland lakes $\geq$ 20 acres should be included.

We modified the satellite image analysis procedure for this assessment to increase its efficiency for the much larger number of lakes (many thousands) across the state. The main modification is in the way signatures are extracted. Instead of using a polygon or AOI that may need to be maneuvered manually to select an unaffected portion of a lake, we are selecting a "filtered" sample of pixels from the entire lake polygon. This is accomplished by using the unsupervised classification of the water-only image and masking out any affected pixels. Signatures were extracted from the modified open-water image using the lake polygon layer instead of the AOI. Instead of using the mean brightness value of the sample, we used the program "Get_Hist" (developed by our Wisconsin RESAC partners) to calculate a mean spectral response from the darkest 50% of the remaining unaffected pixels. The darkest pixels are associated with the clearest water. Comparisons between the AOI and "Get_Hist" methods indicate that the results are similar. A regression analysis between TSI(SDT) inferred from a Landsat image using each method resulted in an $R^2 = 0.97$ and standard error of estimate (SEE) = 1.43 TSI(SDT) units. A preliminary lake clarity map of lakes in central and southern Minnesota based on this approach is shown in Figure 5.

Use of this method requires that high quality lake polygon coverage is available to identify and extract lake spectral response data. Significant editing of an available hydrography layer was necessary to create an acceptable lake polygon coverage. The requirements for this coverage include a unique lake identification numbers with sub-basin identification for each lake. Lakes with many basins or significant bays were broken up into sub-basins. The unsupervised classification of the water-only image was used to identify basins with different water clarity. ArcView then was used to split the lake polygon into unique polygons that were identified with unique sub-basin identification numbers.

For remote areas where field data to calibrate Landsat data are scarce or not available, it is necessary to use alternative calibration methods. These methods may include using data from an adjacent scene or from a different time period for the same scene to develop a regression relationship. Several preliminary studies were conducted to test the reliability of these approaches using Landsat images from central and southern Minnesota. Scenes from the same path taken on the same day can use the model developed from an upper or lower scene as long as atmospheric conditions are similar. We classified the Path 28 Row 30 image from August 26, 1991, (which had limited reference data—only 9 SDT measurements, with all SDT measurements below 1 m, within seven days of the image date) based on the model developed from the Path 28 Row 29 image for the same date (which had 25 SDT measurements within seven days of the image date). A comparison of the classified lakes from the overlap area of both images indicates very good agreement with $R^2 = 0.996$ and the SEE = 0.48 TSI(SDT) units.

We also conducted an experiment using an adjacent scene from a different path and time to calibrate a Landsat image. The first image (Path 28 Row 28(east) from August 7, 1990) had 140 SDT measurements within 7 days of the overpass date. The regression model for ln(SDT) versus Landsat spectral-radiometric data for this scene has an $R^2 = 0.81$ and SEE = 0.31. The adjacent scene (Path 29 Row 28(west) from August 17, 1991) has 57 SD measurements within 7 days of the Landsat overpass date. The regression model for ln(SDT) versus Landsat spectral-radiometric data for this scene has an $R^2 = 0.81$ and SEE = 0.29. For the experiment we used Landsat inferred values of ln(SDT) from the "east" image as reference data for the overlapping lakes of the "west" image. This resulted in 714 reference lakes and resulted in a regression model (inferred ln(SDT) versus Landsat spectral-radiometric data) with an $R^2$ of 0.58 and SEE = 0.50. To improve the relationship we limited the reference lakes to those for which $\geq$100 pixels had been used to calculate the spectral mean. This reduced the number to 330 reference lakes and yielded a regression model with $R^2 = 0.67$ and SEE = 0.48. This latter relationship was used to calculate lake water clarity for the "west" image. A statistical comparison of water clarity results from the SDT reference data model and the adjacent scene reference data model for the "west" image indicate a very strong relationship with an $R^2$ of 0.95 and SEE = 2.25 TSI(SDT) units.

Finally, preliminary work on using an image from the same scene but different time period to calibrate other images indicates that it is feasible as long as some method of radiometric calibration is applied (Kloiber et al., submitted). However, the resulting predictions from this method have a greater uncertainty than those obtained using *in-situ* data to develop a scene-specific regression relationship. Therefore, it is preferable to develop scene-specific relationships with *in-situ* data whenever possible. In the future, as better atmospheric correction technologies become available, this approach may prove to be as accurate.

## CONCLUSIONS

Our work has demonstrated the usefulness of remote sensing systems for the assessment of variables (lake water clarity) that are not usually available on a regional scale from conventional data collection programs. We have demonstrated how a regional lake water clarity database can be created using historical Landsat images to extract data not available for most lakes by other methods and how landscape stressors can be related to lake responses on a regional scale using GIS and remote sensing techniques.

Our current work expands the approach to a much broader scale statewide assessment of lake water clarity in Minnesota with the ultimate goal of developing a better understanding of lake systems on a regional scale promoting the protection of this valuable resource.

## ACKNOWLEDGEMENTS

We gratefully acknowledge others who contributed to the work described in this report: Jean Doyle, and Teresa Anderle (University of Minnesota) and Jonathan W. Chipman (University of Wisconsin).

Funding for the work described in this paper was provided by the Metropolitan Council - Twin Cities Water Quality Initiative Grant Program, Minnesota Department of Natural Resources, NASA – Upper Great Lakes Regional Earth Sciences Application Center, and University of Minnesota.

## REFERENCES

Anderle, T.H., 1999. Analysis of stormwater runoff and lake water quality for the Twin Cities metropolitan area. M.S. thesis, University of Minnesota, Minneapolis.

Anderle, T.H., Brezonik, P.L., and Kloiber, S.M. Seasonal patterns of chlorophyll *a* and Secchi disk transparency in lakes of east-central Minnesota: implications for design of ground- and satellite-based monitoring programs. submitted to Lake Reserv. Manage.

Brezonik, P.L., Osgood, R.A., Olmanson, L., Day, E., Hatch, L., Doyle, J., Perry, J.A., Bauer, M., MacBeth, E., and Anderle, T., 2000, Cumulative impacts of development on lakes in the North Central Hardwood Forest ecoregion of Minnesota: an exploratory study. Project completion report submitted to the Minnesota Department of Natural Resources, University of Minnesota, St. Paul, MN.

Brown, D., Warwick, R. and Skaggs, R., 1977, Lake conditions in east central Minnesota. Rept. No.5022. Minnesota Land Management Information System, Center for Urban and Regional Affairs, University of Minnesota, Minneapolis, MN.

Carlson, R.E., 1977, A trophic state index for lakes. Limnol. Oceanogr., 22, 361-369.

Cox, R.M., Forsythe, R.D., Vaughan, G.E., and Olmsted, L.L., 1998, Assessing water quality in the Catawba River reservoirs using Landsat Thematic Mapper Satellite Data. J. Lake and Res. Manag., 14, 405-416.

Day, E., 2000, Landsat 5 and 7 as tools for monitoring water quality. M.S. thesis, Univ. of Minnesota, St. Paul.

Kloiber, S.M., Anderle, T.H., Brezonik, P.L., Olmanson, L.G., Bauer, M.E., and Brown, D.A., 2000, Trophic state assessment of lakes in the Twin Cities (Minnesota, USA) region by satellite imagery, Arch. Hydrobiol. Spec. Issues. Advanc. Limnol., 55, 137-151.

Kloiber, S.M., Brezonik, P.L., Olmanson, L.G., and Bauer, M.E, Development of a remote sensing method for synoptic, regional water quality assessment. Submitted to Remote Sens. Environ.

Lathrop, R.G., 1992, Landsat thematic mapper monitoring of turbid inland water quality, Photogram. Engrg. Remote Sens., 58, 465-470.

Lathrop, R.G., and Lillesand, T.M., 1986, Utility of Thematic Mapper data to assess water quality in southern Green Bay and west-central Lake Michigan. Photogram. Engrg. Remote Sens., 52, 671-680.

Lavery, P., Pattiaratchi, C., Wyllie, A. and Hick, P., 1993. Water quality monitoring in estuarine waters using the Landsat Thematic Mapper. Remote Sens. Environ., 46, 268-280.

Lillesand, T.M., Johnson, W.L., Deuell, R.L., Lindstrom, O.M. and Meisner, D.E., 1983, Use of Landsat data to predict the trophic state of Minnesota lakes. Photogram. Engrg. Remote Sens., 49, 219-229

Olmanson, L.G., 1997, Satellite remote sensing of the trophic state conditions of the lakes in the Twin Cities Metropolitan Area. Masters Paper, Department of Water Resources Science, University of Minnesota, St. Paul, MN.

Olmanson, L.G., Kloiber, S.M., Bauer, M.E., Day, E.E., and Brezonik, P.L. 2000, Upper Great Lakes RESAC Lake water quality, Image processing protocol, University of Minnesota, St. Paul, MN.

Pattiaratchi, C., Lavery, P., Wyllie, A. and Hick, P., 1994, Estimates of water quality in coastal waters using multi-date Landsat Thematic Mapper data. Int. J. Remote Sens., 15, 1571-1584.

**Table 1. Lake Classification**

| Class Number | Area | Depth | Conductivity |
|---|---|---|---|
| 1 | Small (<100 acres) | Shallow (<7 ft) | (<29 uS/cm) |
| 2 | Intermediate (101-500 acres) | Intermediate (7-16 ft) | (29-141 uS/cm) |
| 3 | Large (>500 Acres) | Deep (>16 ft) | (142-500 uS/cm) |
| 4 | | | (501-7078 uS/cm) |

**Table 2. All DNR Study Lakes: Simple linear regressions for all population and land use variables**

| Variable | R-squared | p | Slope |
|---|---|---|---|
| Population Density | 0.079 | <0.0001 | + |
| % Developed (150) | 0.076 | <0.0001 | + |
| % Developed (var) | 0.109 | <0.0001 | + |
| Animal Density | 0.000 | 0.831 | horizontal |
| % Agriculture (150) | 0.001 | 0.817 | horizontal |
| % Agriculture (var) | 0.018 | 0.003 | - |
| % Grassland (150) | 0.049 | <0.0001 | - |
| % Grassland (var) | 0.056 | <0.0001 | - |
| % Wetland (150) | 0.014 | 0.009 | + |
| % Wetland (var) | 0.014 | 0.007 | + |
| % Forest (150) | 0.113 | <0.0001 | - |
| % Forest (var) | 0.090 | <0.0001 | - |

**Table 3. R-squared values for TCMA multiple regressions for each class group**

| Lake Class | Both Buffers | 150 m Buffer | Variable Buffer |
|---|---|---|---|
| All TCMA Lakes | 0.12 | 0.09 | 0.10 |
| 123 | 0.25 | 0.23 | 0.25 |
| 223 | 0.35 | 0.15 | 0.09 |
| 213 | 0.32 | 0.18 | 0.23 |
| 113 | 0.37 | 0.26 | 0.21 |
| 233 | 0.39 | 0.70 | 0.50 |
| 323 | n=too small | 0.71 | 0.57 |
| 333 | n=too small | 0.98 | 0.70 |

**Table 4. R-squared values for ALA multiple regressions for each class group**

| Lake Class | Both Buffers | 150 m Buffer | Variable Buffer |
|---|---|---|---|
| All ALA Lakes | 0.25 | 0.14 | 0.19 |
| 223 | 0.39 | 0.28 | 0.47 |
| 123 | 0.28 | 0.20 | 0.39 |
| 333 | 0.36 | 0.36 | 0.05 |
| 233 | 0.67 | 0.35 | 0.61 |
| 224 | 0.43 | 0.16 | 0.22 |
| 323 | 0.42 | 0.28 | 0.34 |

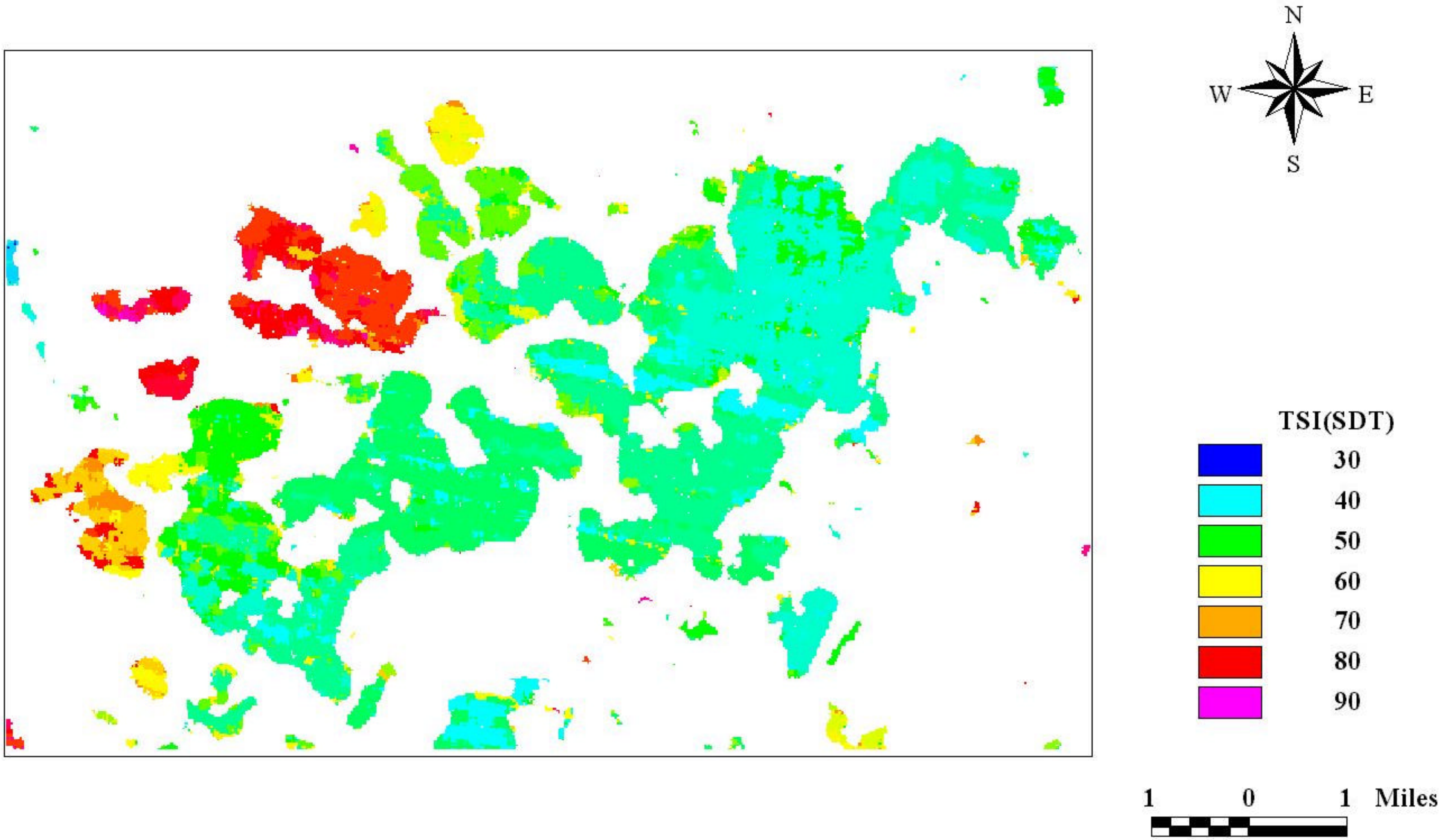



**Figure 1** September 4, 1991 Pixel-Level water clarity/tropic condition map for Lake Minnetonka and surrounding lakes.

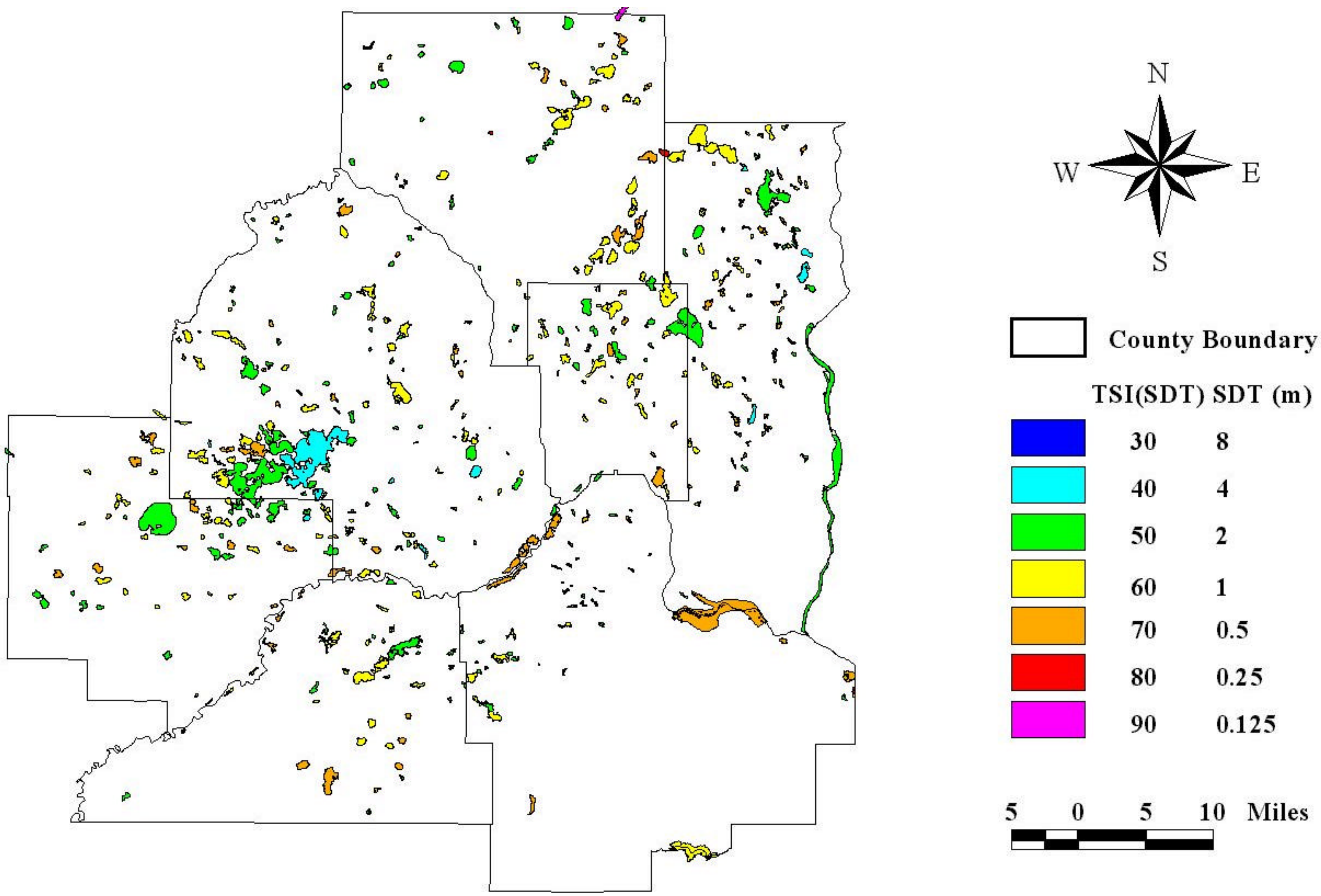



**Figure 2 September 4, 1991 lake-level water clarity /tropic condition map for lakes in the TCMA.**

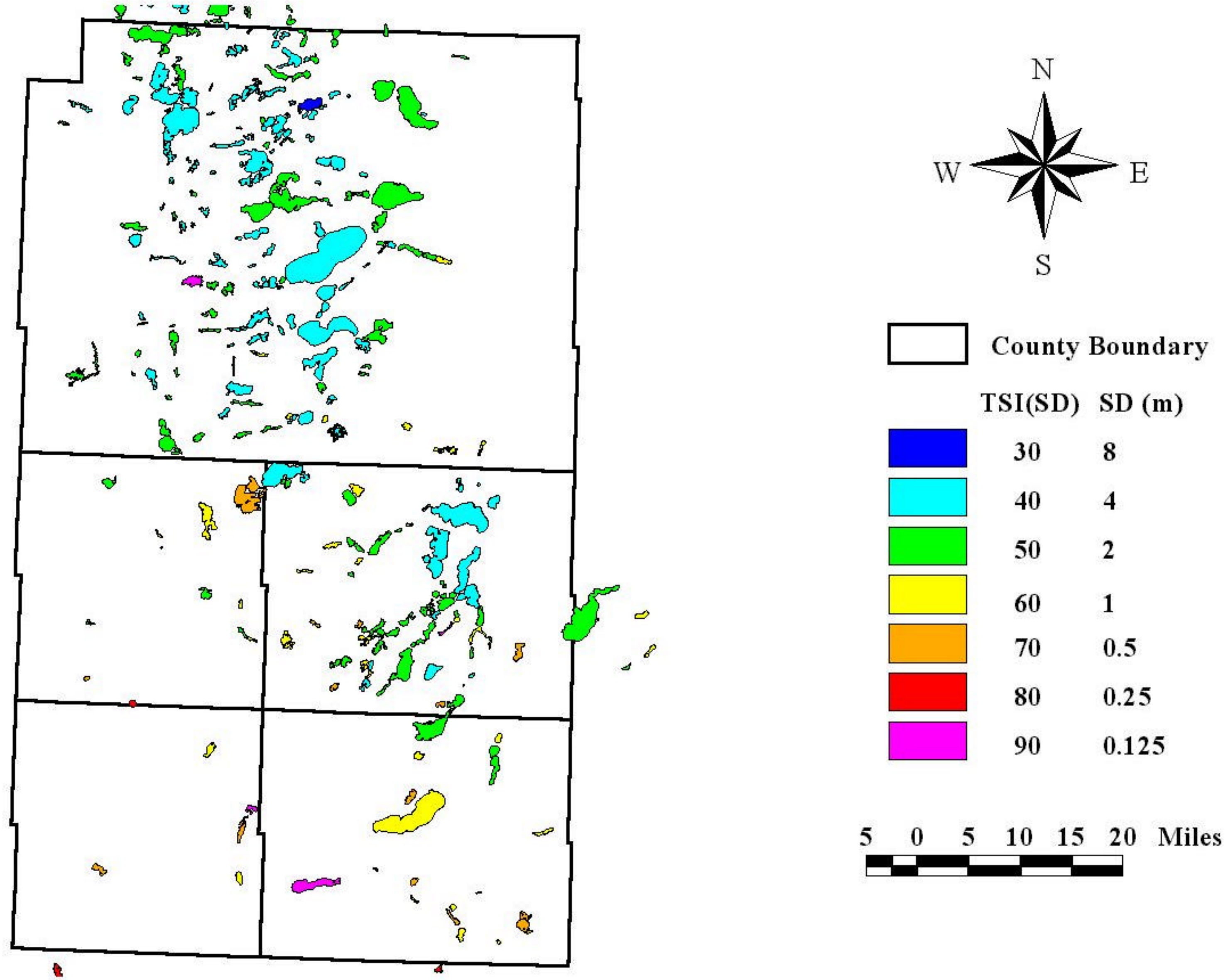



**Figure 3 August 17, 1991 Water clarity/trophic condition map for lakes in ALA.**

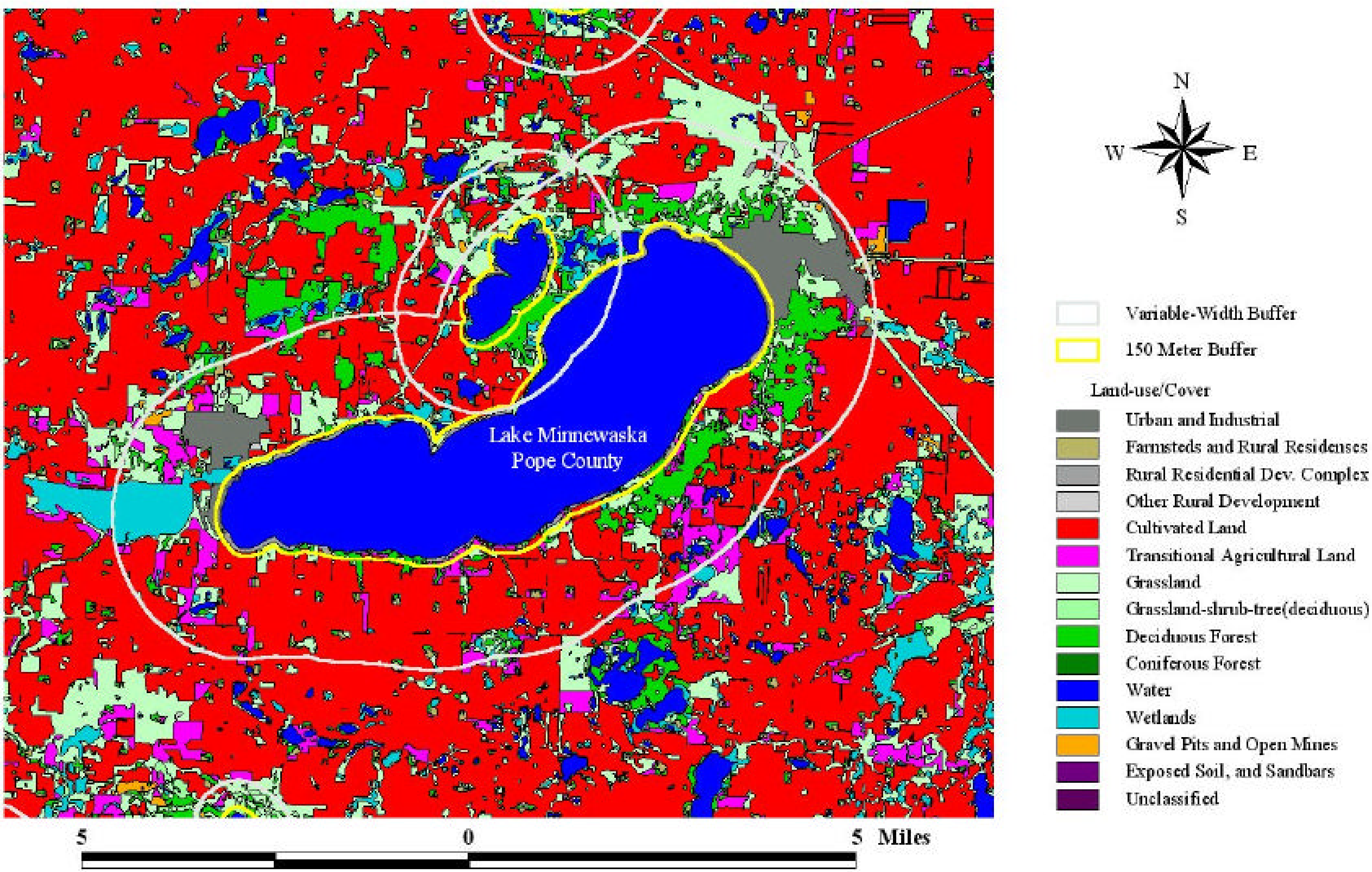



**Figure 4 Illustration of 150 m (near shore) and variable-width landscape buffers for two ALA lakes (Minnewaska [7110 acres] and Pelican[519 acres]) with land-uses dileneated within the buffer.**

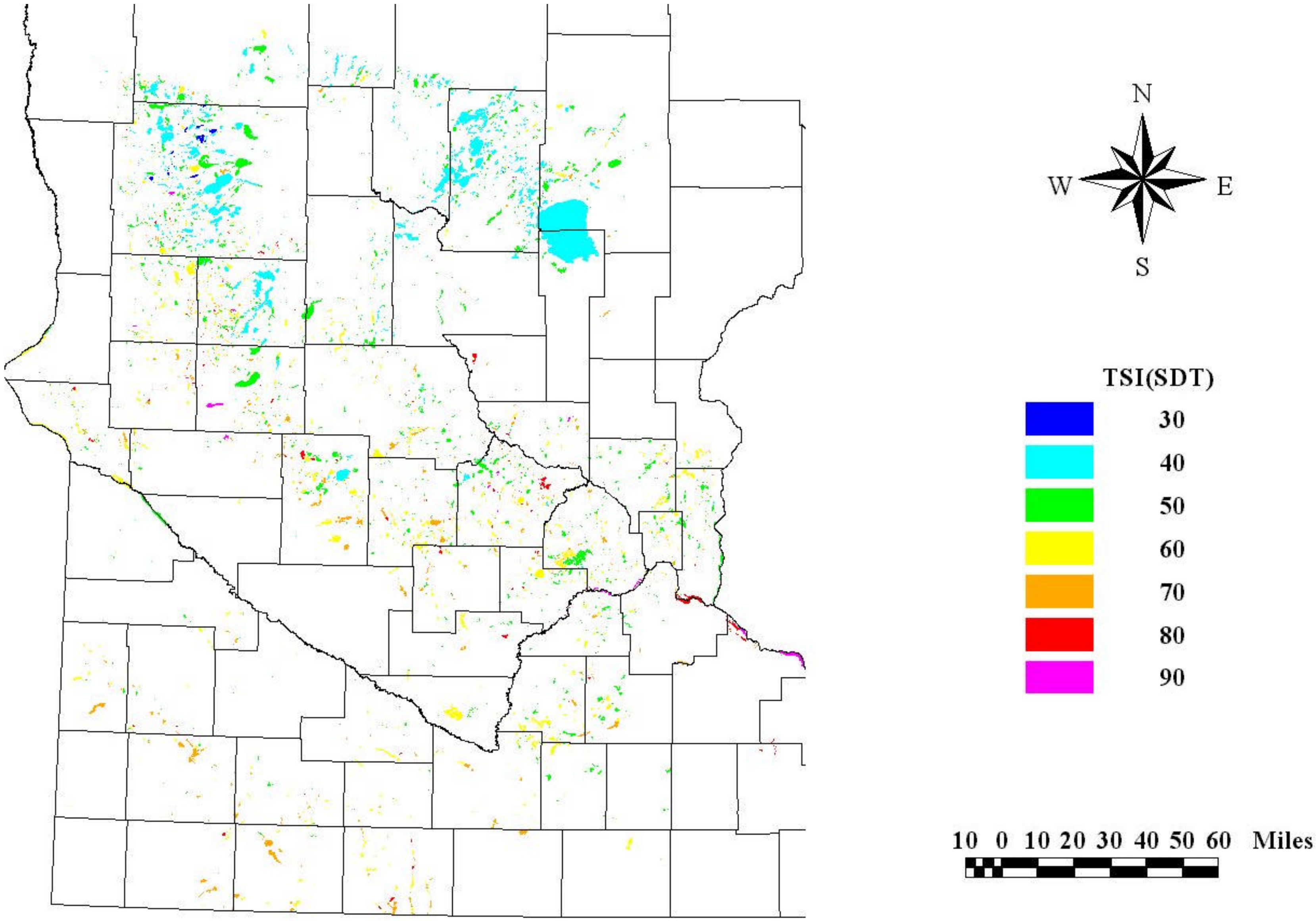



**Figure 5  Early 1990s water clarity/trophic condition map for 4057 lakes in central and southern Minnesota.**

# Reservoir Mapping and Volume Measurement using a GPS/Depth Sounding System

Harold E. Post, Thomas J. Grizzard, and Adil N. Godrej
Virginia Tech
The Charles E. Via, Jr., Department of Civil and Environmental Engineering
Occoquan Watershed Monitoring Lab
9408 Prince William St.
Manassas, VA 20110
E-mail: hpost@vt.edu

## ABSTRACT

With an increased national awareness of the need to insure the integrity of surface water supplies, watershed managers and water purveyors have become more focused on developing and maintaining information to support long-term management decisions. For surface water supplies, this often includes having a more complete understanding of issues that relate to storage and safe yield, such as the pool elevation - storage relationship. It is essential to have accurate, cost-effective methods for making such bathymetric measurements so that surveys may be undertaken at frequencies that will provide useful trend information. The method described herein provides a means for accurately measuring reservoir volumes, mapping the reservoir bottom, and estimating volume loss over the life of the reservoir.

Bathymetric surveys were performed using an integrated system consisting of acoustic depth sounding equipment coupled with a real-time differential global positioning system (DGPS). The entire instrumentation suite was sufficiently portable that it could be mounted and operated from a small boat. The sounding equipment provided a data set of point elevations on the reservoir bottom. When coupled with spatial positioning provided by GPS, a dataset of x,y,z coordinates was created to describe the reservoir bottom. The instrumentation employed provided horizontal and vertical positioning accuracy of less than 1 meter and 0.03 meters, respectively. Coordinates were typically recorded at a time interval of one second while the boat cruised a pattern of lines selected to provide adequate spatial coverage of the reservoir surface.

Collected field data were compiled using software that applied bottom elevations to a horizontal grid of evenly spaced points. These data were used to create a topographic map of the reservoir bottom, and to calculate storage volume. Volumes and surface areas were calculated for various elevations to create an Area/Capacity curve for the reservoir.

Estimates of storage change may be made from comparison of reservoir volumes from surveys appropriately spaced in time. Performance of the described method may be expected to be superior to conventional surveys based on measurements at fixed cross sections, as the latter may be subject to bias caused by localized sedimentation and scour at the measurement site chosen.

**Keywords:** Bathymetric survey, GPS, mapping, reservoirs.

## INTRODUCTION

With an increased national awareness of the need to insure the integrity of surface water supplies, watershed managers and water purveyors have become more focused on developing and maintaining information to support long-term management decisions. For surface impoundments, this often includes having a more complete understanding of issues that relate to storage and safe yield. In order to have an adequate understanding of the safe yield of a watershed-reservoir system, accurate knowledge of the pool elevation - storage relationship of the impounded body is essential. Further, in reservoir systems exposed to significant loads of runoff-borne sediments, it is necessary to periodically update the storage estimate in the interest of detecting any temporal trends as they develop.

Two northern Virginia water supply reservoirs, the Occoquan Reservoir in Fairfax County, and the Beaverdam Creek Reservoir in Loudoun County, were mapped with the method described herein. The surveys were conducted by the Occoquan Watershed Monitoring Laboratory (OWML) of the Charles E. Via, Jr., Department of Civil and Environmental Engineering at Virginia Tech. The survey of Beaverdam Creek Reservoir represented an initial determination of storage, while that on the Occoquan Reservoir followed work done in 1995 with a Global Positioning System (GPS)/acoustic depth sounder system. The recent survey of the Occoquan Reservoir was used to update the storage capacity estimate, and to provide an estimate of the rate of storage change.

The surveys were conducted using an integrated differential GPS and depth sounding system leased from Innerspace Technology, Inc. (1998). The company has developed hydrographic surveying systems of this type in order to support measurements of navigation channel depths and dredge spoil volume computations for harbor maintenance. The horizontal plane positioning component of the system consisted of a differential GPS unit. Depth from the water surface to the bottom was measured using an acoustic depth sounder, and later adjusted to elevation above mean sea level by subtracting from the pool elevation. A laptop microcomputer was interfaced with the system to provide real-time data logging and interface with the GPS system.

### Bathymetric Survey Methodology

**Bathymetric Survey Equipment.** The integrated survey instrumentation from Innerspace Technology, Inc. was the Differential GPS Survey System II, consisting of a Trimble AgGPS 132 differential GPS unit with an OmniSTAR satellite differential correction receiver (Trimble, 1999), and an ITI Model 448 acoustic depth sounder (Innerspace Technology, Inc., 1998). Data logging guidance was provided by ITI Datalog With Guidance software (Innerspace Technology, Inc., 1998) installed on a notebook microcomputer. The computer system and software provided navigation information, which was used by the boat helmsman to locate and move along established survey lines. Horizontal position recorded from the GPS unit and depth from water surface recorded by the acoustic depth sounder were stored as ASCII files for each line surveyed.

**Differential GPS.** The Navigation Satellite Timing and Ranging (NAVSTAR) Global Positioning System was developed by the U.S. Department of Defense to provide a real-time, rapid, and accurate navigation and positioning system capable of functioning under all weather conditions. The GPS consists of a constellation of 21 satellites (plus 3 spares) which orbit the earth once every 12 hours, continuously transmitting navigation signals. A GPS ground receiver acquires position information from each of the satellites for which above-the-horizon line-of-sight is available. If at least three satellites are in view, a GPS receiver may calculate its horizontal position. With four satellites in view, a three dimensional position may be determined. While positioning in the vertical dimension (when 4 satellites are in view) is useful, it is not accurate at the scale required for depth sounding in water bodies.

GPS systems use triangulation to determine location, wherein a line is drawn from each observed point (in this case the satellites) to the observer. The location of the observer is at the point where the three (or more) lines converge. The small time corrections required by movement of the earth on its axis and the satellites in their orbits are provided by atomic clock signals from each orbiting transmitter.

Because of the high accuracy of GPS positioning and its world-wide availability, for many years the United States Department of Defense (DOD) incorporated a *selective availability* capability, which may be described, in its most basic terms, as an error in the signal made available to the general public. The accuracy of a position obtained from a single receiver when *selective availability* is in effect is in the 40 to 100 meter range. Obviously, horizontal positioning errors in this range would be unacceptably large for survey purposes. To remove that error and provide data accurate enough for surveying needs, a system of *differential correction* was developed. Local corrections may be made by referencing the satellite navigation signals to a fixed secondary receiver established at a known location. This correction may be applied to the mobile receiver, thereby providing a *differential* GPS, (or DGPS) positioning accuracy of less than one meter in the horizontal plane. The stationary receiver is used to calculate the difference in known location data and that indicated by the GPS system. Corrections may then be computed, and sent to the mobile GPS receiver by radio or satellite.

Recent changes in the world political climate have resulted in a re-evaluation by DOD of the *selective availability* policy. In May, 2000, the U.S. government removed *selective availability* from the NAVSTAR system. However, differential correction is still used to correct for other causes of error, such as errors caused by errors in the satellite constellation geometry, timing, and atmospheric conditions. With *selective availability* removed, position accuracy without differential correction is approximately 10 meters. With differential correction, accuracy is in the sub-meter range.

The equipment used for the survey provided real-time differential GPS positioning. The mobile GPS receiver was designed to acquire both the GPS satellite data feed and the correction data simultaneously, and corrected positions were immediately stored in a computer file, thereby eliminating the need for post-processing data. The GPS receiver utilized was equipped with an internal OmniSTAR (1999) satellite differential correction receiver, which accessed the differential correction signal from a geostationary satellite broadcasting across the entire continental United States. Using the differential correction, the GPS was then used to compute a *true* set of location coordinates. After pairing with the depth measurement from the ultrasonic sounding device, the position of each point on the reservoir bottom was stored in an ASCII data file. Vertical elevations were corrected to ft., msl after subtracting from the pool elevation.

Figure 1 is a schematic diagram of the essentials of the OmniSTAR DGPS system (OmniSTAR, 1999) for positioning in the horizontal plane. The system consists of:

(1) The constellation of NAVSTAR GPS satellites.
(2) A series of eleven known-position GPS monitoring sites located around the North American continent or the purpose of providing correction signals.
(3) A land line system for transmitting correction signals to a central station.
(4) A control center located in Houston, Texas which uplinks correction data to the geostationary satellite.
(5) The GE Spacenet 3 geostationary satellite which transmits correction signals to ground stations.
(6) The continent-wide footprint where correction signals may be received.
(7) A typical differential GPS receiver being employed in some positioning or surveying activity.

**Depth Sounder.** The depth sounder employed in the bathymetric surveys was an Innerspace Technology Model 448 depth sounder (Innerspace Technology, 1998). The transducer operated at 208 kHz with an 8 degree beam width. Operation at 208 kHz allowed the transducer to resolve the elevation of the sediment layer on the bottom, rather than penetrating some distance into semi-unconsolidated material that may be on the bottom. The 8 degree beam width gives a reasonable signal footprint at typical depths found in reservoirs along the East Coast. For example, at a 10 meter depth, the signal "footprint" would be 1.2 meters in diameter. The depth sounder was capable of making a maximum of 20 soundings per second, and the data logging system was configured to store depth and position once each second.

**Equipment Integration.** A laptop computer with ITI Datalog With Guidance software (Innerspace

Technology, Inc., 1998) collected and recorded horizontal positioning data from the GPS equipment, and data from the depth sounder through serial connections to each instrument. In addition, the software stored a predetermined survey pattern and provided guidance to the boat helmsman to steer along predetermined survey lines. The computer provided a display of navigation information consisting of direction and distance information, thereby guiding the helmsman to the start of each survey line, and then along each line. Horizontal position recorded from the GPS unit and depth from water surface recorded by the acoustic depth sounder were stored as ASCII files for each line surveyed. The paired instrumentation suite (consisting of differential GPS and ultrasonic sounding equipment) provided positioning accuracy in the horizontal plane within 1.0 meter, and within 0.03 meters (0.1 feet) in the vertical dimension (Innerspace Technology, Inc., 1998).

**Survey Plan**

A series of contiguous straight lines known as sections were established along the length of the reservoirs to be surveyed, with the orientation maintained as close as possible to the centerline. Section lengths were adjusted as required to maintain good alignment with the centerline. For the Occoquan Reservoir, 29 sections were established, varying in length from 80 meters to 3 050 meters (600 to 10,000 feet). A series of parallel lines running perpendicular to each section were established at a longitudinal spacing of 60 meters (200 feet). A total of 380 such lines were established, and were used as the basis for the field survey cross-sections. The survey plan was developed using Fairfax County 1 inch:500 ft contour maps and Prince William County 1 inch:200 ft contour maps (Air Survey and Design, Inc., 1984; Prince William County Office of Mapping and Information Resources, 1987).

For Beaverdam Creek Reservoir, which was a much smaller water body, a series of parallel lines spaced 46 meters (150 feet) apart were used for the survey. The lines were oriented east-west, running perpendicular to the centerline of the reservoir, and covered the reservoir from the dam to the upstream end of the reservoir.

The length and bearing of each survey line was stored in the survey system microcomputer, and the helm guidance software programmed to direct the survey boat along the lines while continuously recording paired coordinates of horizontal position and depth to bottom. In small coves and other tributaries along the length of the reservoirs, the survey boat cruised a random pattern in order to collect sufficient data to represent the bottom contour. Separate spatial coordinate and depth data files were stored for each survey line or cove surveyed. Overall, approximately 60 000 and 26 000 separate points were mapped on the Occoquan Reservoir and Beaverdam Creek Reservoir bottoms, respectively, each having a discretely measured horizontal and vertical position.

**Field Activities**

In the performance of required field activities for the bathymetric measurements, the previously described survey equipment was mounted on OWML's 14-foot sampling boat or the larger 17-foot sampling boat. The photographs included in Figures 2 through 5 illustrate key components of the system instrumentation. Figures 6 and 7 show the entire boat-mounted system ready for use during a typical survey operation.

Figure 2 shows the Trimble GPS receiver and the ITI Helmsman Guidance Display. As previously described, horizontal positioning calculations were performed by the GPS receiver and then communicated to a laptop computer (not shown) *via* a serial port connection. A software application installed on the laptop continuously compared real-time GPS positioning signals to previously stored survey course data, and calculated course or steering directions as required to maintain the boat on the appropriate survey line. This information was then communicated to the Helmsman Guidance Display, which provided required steering information to the survey boat operator.

Figure 3 shows the combination GPS and OmniSTAR antenna, which is mounted on the survey boat during operations. Both the GPS satellite positioning signals and the OmniSTAR differential correction signals are received by the combination antenna. The antenna was interfaced with the Trimble GPS receiver previously illustrated in Figure 2.

Figures 4 and 5 show, respectively, the ITI Model 448 depth sounder display unit and the associated ultrasonic depth sounding transducer. The depth sounding equipment was connected to the supervisory laptop microcomputer *via* a second serial port. The previously described Innerspace Technology software application integrated the depth-of-water sounding information with the horizontal positioning data from the differential GPS system. The resultant dataset consisted of x, y, and z coordinates describing the position of each measured point on the reservoir bottom.

Figure 6 is a photograph of the small survey boat ready to begin operations in the vicinity of the boat ramp on Beaverdam Creek. Figure 7 is a photograph of surveying underway on the Occoquan Reservoir in January, 2000 in a larger boat.

**Data Analysis**

**Establishment of Reservoir Grid.** Volume calculations and contour mapping were performed using *Surfer 7 for Windows*, which is a software application developed by Golden Software, Inc. (1999a). The data from the bathymetric survey was converted to ASCII files in (x,y,z) format with depth (z) expressed in ft., msl. Because the raw data file created from the field survey consisted of data that were irregularly spaced in the horizontal plane, the data could not be conveniently used in the raw form for mapping the reservoir bottoms. In order to remedy this deficiency, the software was used to establish a grid of evenly-spaced elements across the survey boundaries, and a series of interpolation algorithms was employed to assign an elevation value to the centroid of each grid element. Each row and column of the grid points established was referred to as a grid line, and these were established at 7.6-meter (25-foot) intervals throughout the the Beaverdam Creek Reservoir, and at 22.8-meter (75-foot) intervals on the Occoquan Reservoir. The 22.8-meter grid for the Occoquan Reservoir was chosen to provide an acceptable balance between accuracy and the time constraints imposed by calculations. Running on a 300 MHz Pentium class microcomputer, the 22.8-meter gridding took approximately 14 hours to complete. The software estimate to grid the raw data file at 7.6-meter intervals was 255 hours.

Although it was not practical to grid the raw data for the Occoquan Reservoir at a spacing of 7.6 meters, the use of a spline smoothing technique made it possible to add two additional grid lines between each calculated line, thereby obtaining a final spacing of 7.6 meters. The spline smoothing algorithm performed a cubic spline interpolation between adjacent grid points to add additional points between the original points. As described in the software manual (Golden Software, Inc. 1999b), "*[the] interpolation simulates a drafting technique where a flexible strip (a spline) is used to draw a smooth curve between data points.*"

The final data files, with grid spacing of 7.6 meters, were used to calculate reservoir volumes and pool areas. The decision to employ the finer grid resolution was taken in order to provide less angularity in the development of contours, thereby increasing the overall accuracy of the volume computation.

In order to extend the area-capacity relationship to elevations above that of the pool(s) at the time of the surveys, a series of contour lines were digitized from topographic maps of the area surrounding each reservoir. Elevations above full pool were included in the data file to provide a continuum of data to elevations well above the water surface. The digitized data consist of a series of (x, y) points with the z elevation of the contour line. These data were added to the field survey data in the volume computations and contour mapping. A boundary file was used to establish the limits of the area in the grid file used in the computations using the topmost contour line described above. The boundary file consisted of a series of (x, y) points defining a closed curve of fixed elevation around the desired survey boundary. The boundary file was applied to the grid file described above so that area-capacity calculations were performed only within the area of the boundary file.

**Volume Calculations.** The previously described software was used to calculate volumes using the *Extended Trapezoidal Rule* (Golden Software, 1999b, Press *et al.*, 1988). Volumes were calculated using the bounded grid file previously described and a plane surface which was set at each elevation for which intermediate volumes were desired. The software created a series of prisms extending from the grid points on the reservoir bottom to the selected plane surface. The volume of each prism was calculated, and iteratively summed to provide the total volume at the elevation of the plane surface. In order to provide adequate resolution for plotting the storage elevation relationship, reservoir volumes were calculated in one-foot increments.

## RESULTS

Area-capacity curves for both reservoirs were developed, and are shown in Figures 8 and 9. Contour maps of the reservoir bottoms were also created. Small-scale versions of the maps are shown in Figures 10 and 11.

The total volume of the Occoquan Reservoir was computed to be $3.15 \times 10^7$ $m^3$ (8.32 billion gallons) in the 2000 survey (OWML, 2000a), and the surface area was found to be 626 hectares (1,522 acres). The 1995 volume of the reservoir was computed as $3.22 \times 10^7$ $m^3$ (8.52 billion gallons) (OWML, 1995). The original estimate of storage volume at the time of reservoir impoundment in 1957 was $3.71 \times 10^7$ $m^3$ (9.8 billion gallons), and $4.24 \times 10^7$ $m^3$ (11.2 billion gallons) after the principal spillway was raised two feet in 1982. According to anecdotal information, the original volume calculations were made by employing planimetric determinations of pool area at a variety of elevations taken from topographic maps with a 20-foot contour interval (Cameron, 1975). Volumes were then calculated by applying the average end area method (Breed and Hosmer, 1958) to areas bounded by the adjacent contours. There are, of course, some clear differences in the precision to be expected from the current method and the desktop technique that represented the standard of practice in the mid-1950's. Because of improvements in method and the 38 year elapsed time between the original estimate and the current survey, the two values are not thought to be directly comparable.

In the five years following the 1995 survey, the estimated storage loss in the Occoquan Reservoir was calculated to be 2.4 percent. At this writing, however, it is not known how this value compares to any random or systematic errors present in the overall survey method, although it is certainly in the same order of magnitude. The third survey using the methods described in this paper is planned for 2005 and will doubtless shed additional light on these accuracy and precision issues.

The Beaverdam Creek Reservoir, with a surface area of 123 hectares (303 acres), was determined to be $5.32 \times 10^6$ $m^3$ (1.41 billion gallons) in the 2000 survey (OWML, 2000b). The reservoir was constructed in 1972, and the original volume estimate was $5.49 \times 10^6$ $m^3$ (1.45 billion gallons) (VSWCB, 1985). The method used to calculate the original volume is not known with certainty (Coughlin, 2000), but is likely to have been the average end area method previously described. In any case, the two estimates compare quite well, and may be seen to differ by less than 3 percent.

## SUMMARY

While the bathymetric survey method described does not directly measure the quantity of sediment deposited in a water body, it does facilitate the calculation of average sedimentation rates from the comparison of volume measurements separated in time. Also, because the method allows a much larger fraction of the bottom of a reservoir or lake to be directly surveyed, it is much less sensitive to the positioning errors that may be introduced in earlier techniques. For example, a survey method that depends upon precisely locating a small number of cross-section ranges on a water body, and then performing a small number of manual soundings may be expected to be quite sensitive to uncertainty in positioning of the survey boat. Even if positioning errors are discounted, it must be further assumed that changes in cross-sectional area at a survey range are representative of the changes that have occurred throughout a reach of reservoir between a given set of cross-sections.

The bathymetric survey technique described herein provides a rapid and cost-effective method for assessing the storage volume of lakes and reservoirs, and should be of great interest to those charged with watershed and/or water supply management responsibilities. The technique is sufficiently inexpensive that replicate surveys may be planned for intervals of a few years. When compared to the average service life of an impounded body, the ability to observe trends in storage change at increments of 5 years or less may be expected to engender a much higher probability of success when planning and executing management intervention(s) that relate to the maintenance of useful storage capacity.

## REFERENCES


Air Survey and Design, Inc., 1984. Fairfax County, Virginia 1 inch:500 foot contour maps, Herndon, VA.

Breed and Hosmer, 1958. Elementary Surveying, Volume I. John Wiley and Sons, Inc. New York, NY.

Cameron, C., 1975. Fairfax County Water Authority. Personal Communication.

Coughlin, T., 2000. Loudoun County Sanitation Authority. Personal Communication.

Golden Software, Inc., 1999a. Surfer Version 7.00 for Windows. Golden, Colorado.

Golden Software, Inc., 1999b. Surfer Users Guide, Golden, Colorado, page 369.

Innerspace Technology, Inc., 1998. Datalog with Guidance Hydrographic Surveying System II. Waldwick, NJ.

Occoquan Watershed Monitoring Laboratory, 1995. "Occoquan Bathymetric Survey Results", prepared for Black and Veatch, Inc.

Occoquan Watershed Monitoring Laboratory, 2000a. "Occoquan Reservoir Bathymetric Survey 2000", prepared for Fairfax County Water Authority.

Occoquan Watershed Monitoring Laboratory, 2000b. "Beaverdam Creek Reservoir Survey 2000", prepared for Loudoun County Sanitation Authority.

OmniSTAR, Inc., 1999. OmniSTAR Worldwide DGPS Service homepage, www.omnistar.com. Houston, Texas.

Press, W.H., Flannery, B.P., Teukolsky, S.A., and Vetterling, W.T. (1988) Numerical Recipes in C, Cambridge University Press, referenced in Golden Software, 1999, Surfer Users Guide, Golden, Colorado.

Prince William County Office of Mapping and Information Resources, 1987. Prince William County, Virginia 1 inch:200 foot contour maps., Prince William, Virginia.

Trimble Navigation Limited, 1999. "AgGPS 132 Operation Manual", Sunnyvale, CA.

Virginia State Water Control Board, 1985. "Safe Yield of Municipal Surface Water Supply Systems in Virginia, Planning Bulletin #335", Richmond, Virginia, page 85.

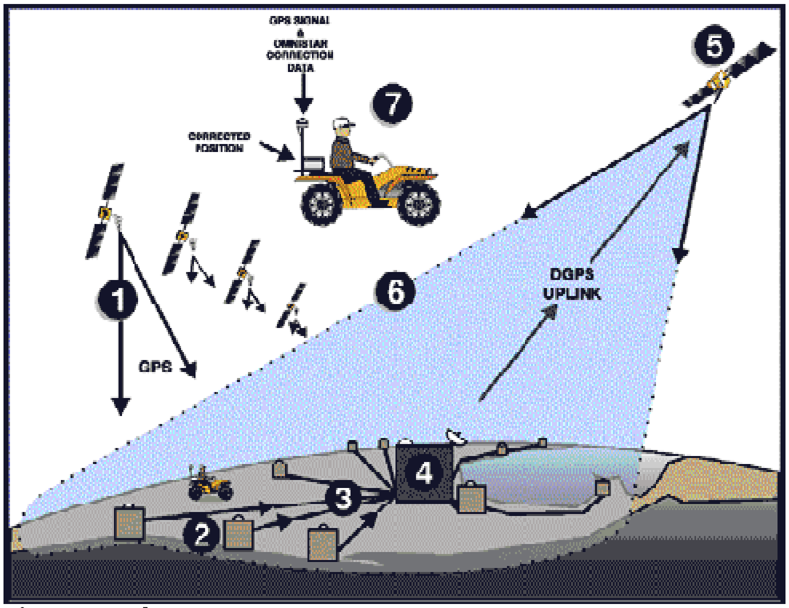



**Figure 1**. The OmniSTAR System
(From OmniSTAR, www.omnistar.com/tech.html)

**Figure 2.** Trimble GPS unit (left) and ITI Helmsman Guidance unit (right).

**Figure 3.** Trimble Combination GPS and OmniSTAR antenna.

**Figure 4.** ITI Model 448 Ultrasonic Depth Sounder.

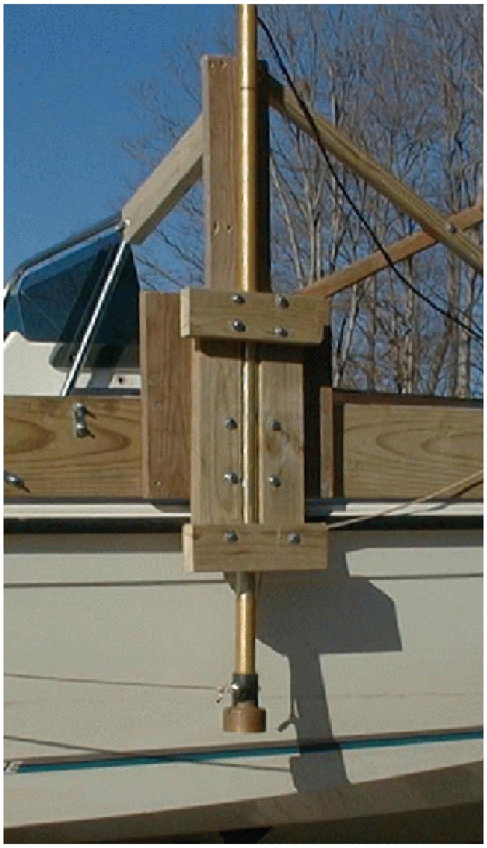


**Figure 5.** ITI Depth Sounder Transducer (raised for transport).


**Figure 6.** Survey boat ready to begin operations.

**Figure 7.** Survey operations underway.

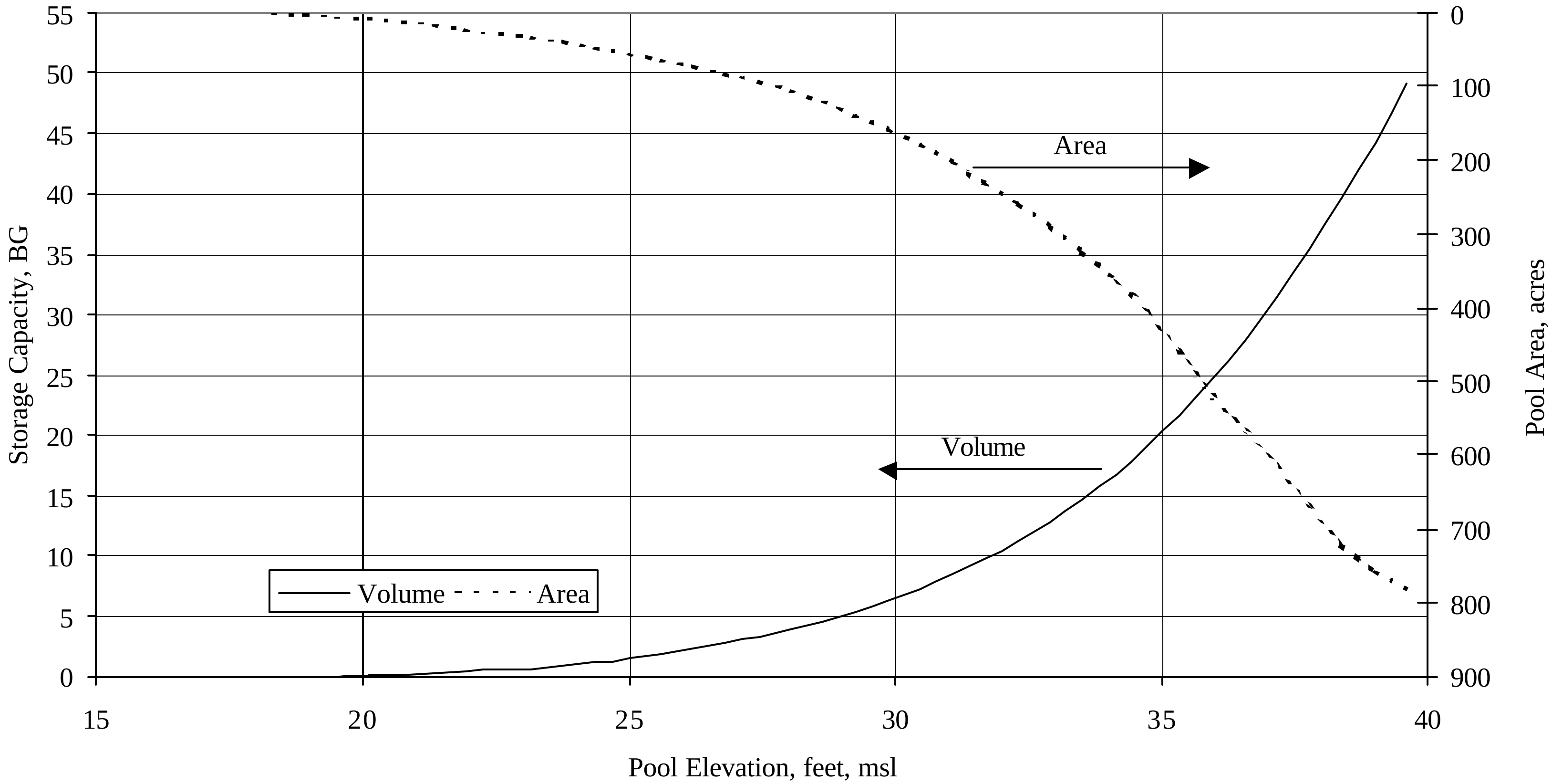


Figure 8. Area-Capacity Curve from Year 2000 Survey of Occoquan Reservoir (Data Source: OWML)

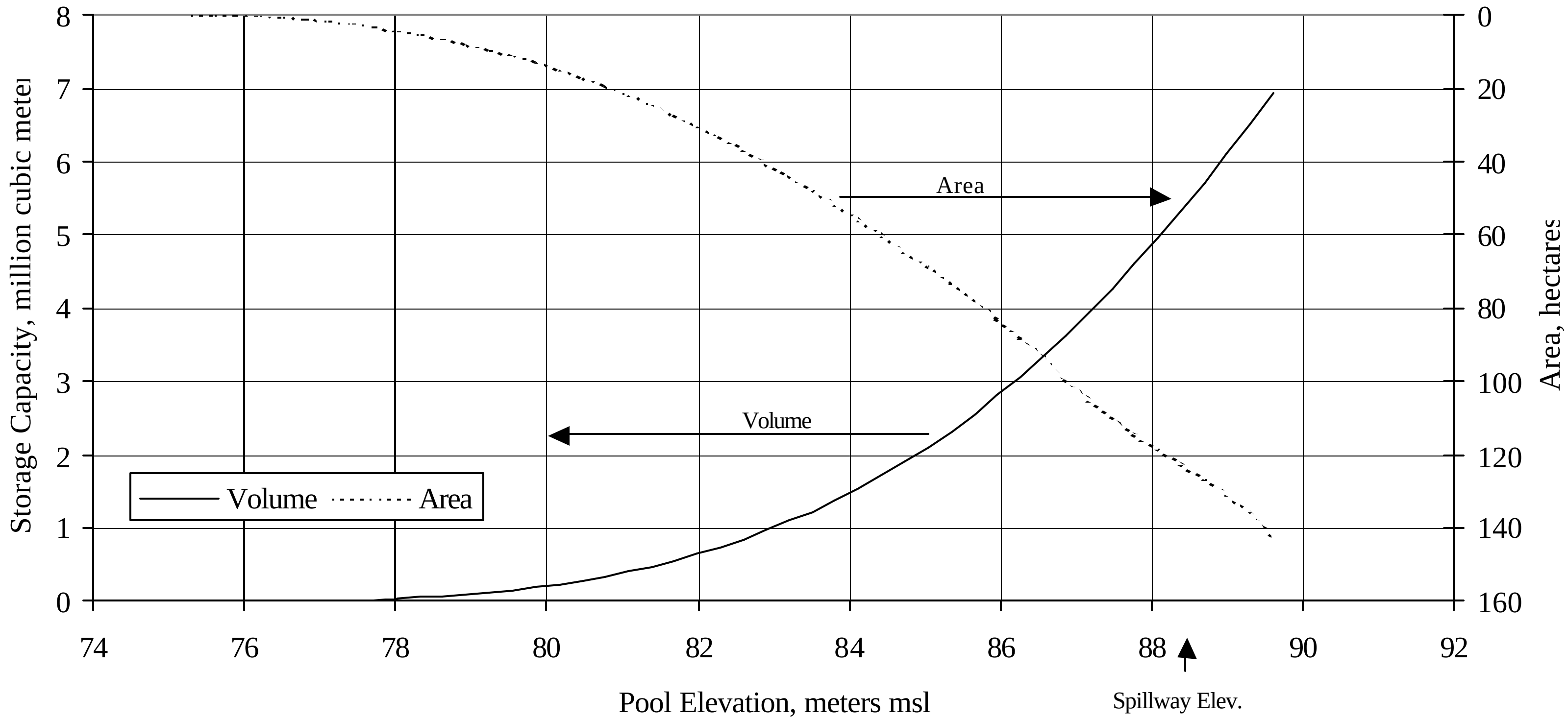



Figure 9. Area-Capacity Curve from Year 2000 Survey of Beaverdam Creek Reservoir. (Data Source: OWML)

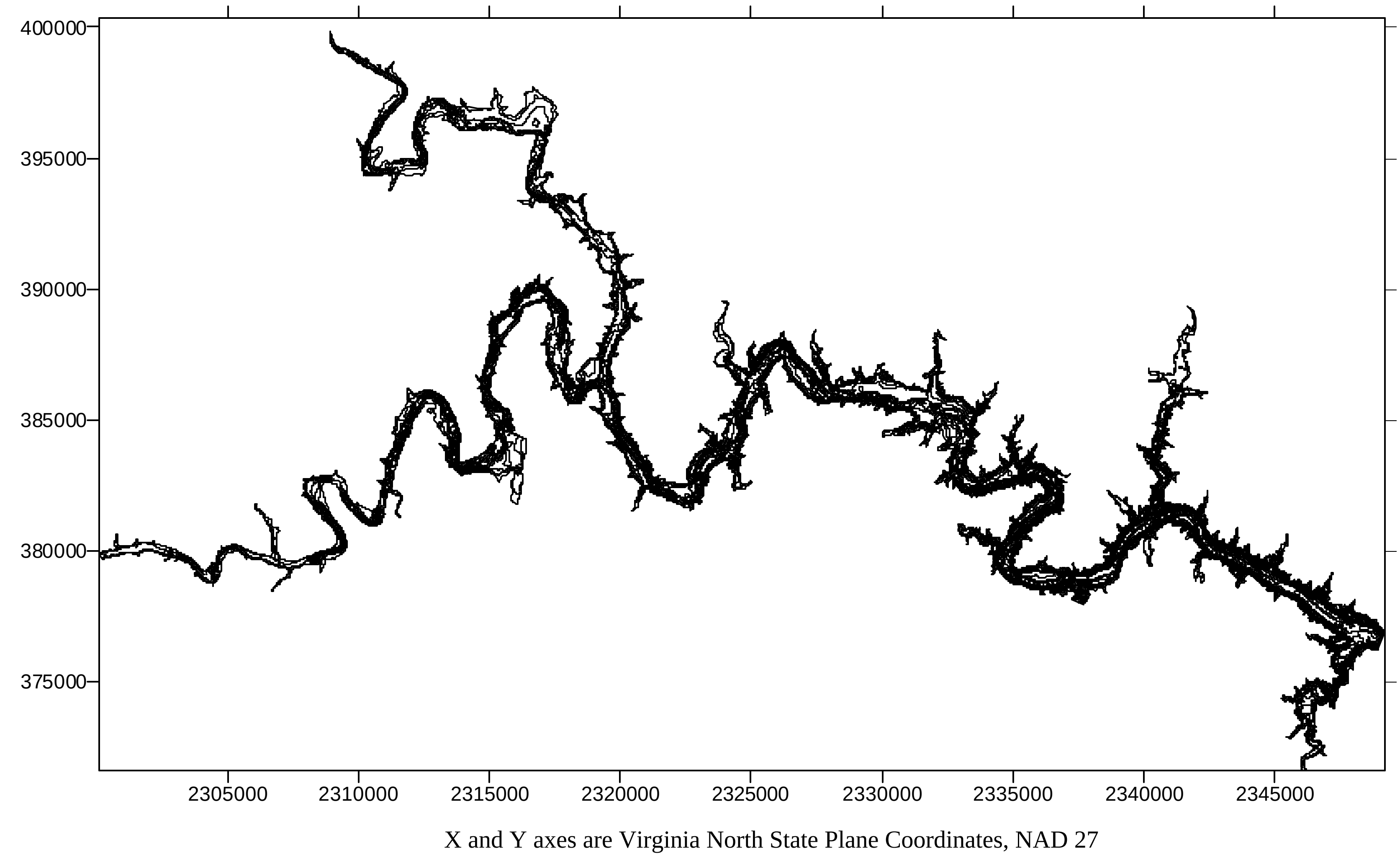
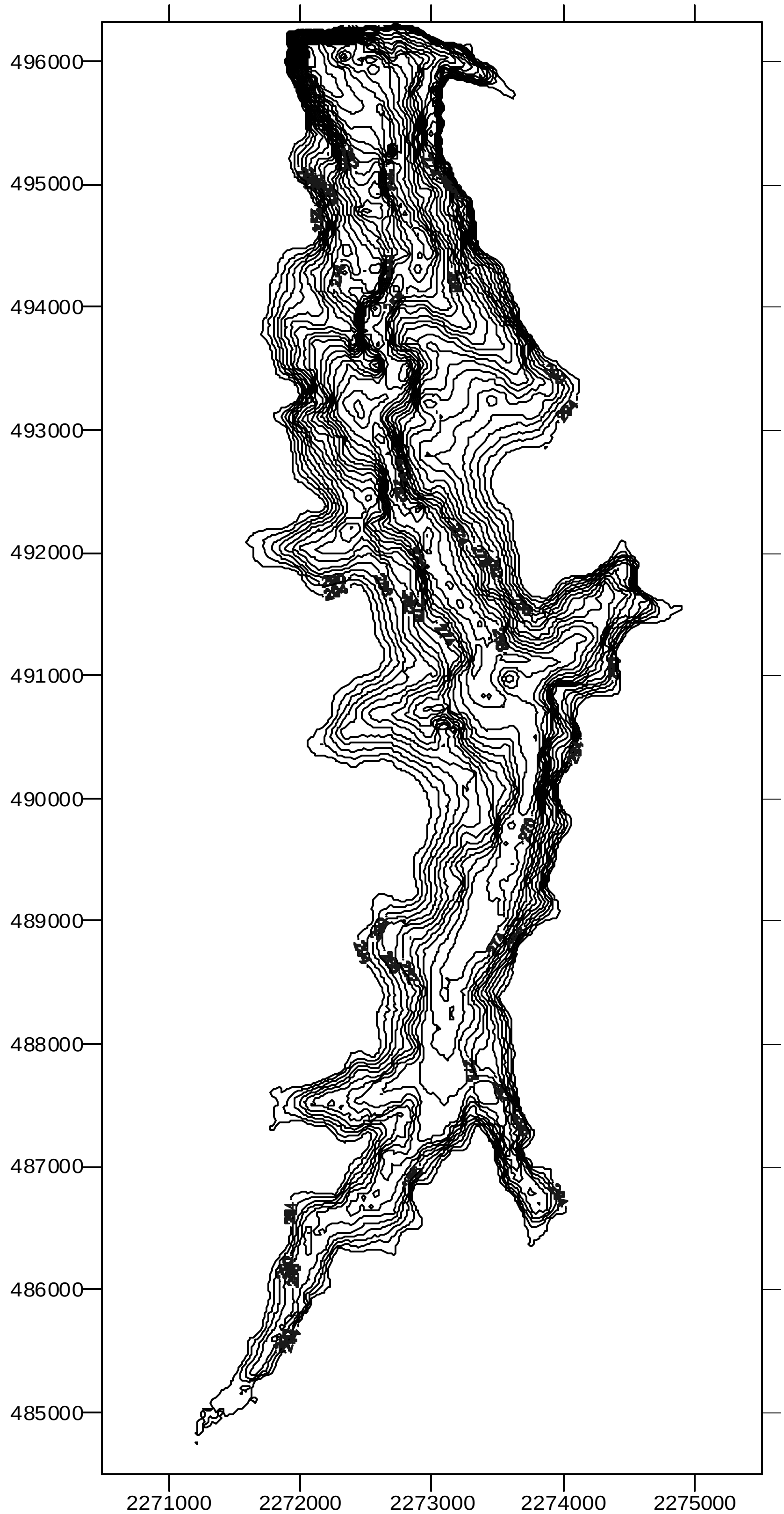



Figure 10. Occoquan Reservoir map, from 2000 survey.




X and Y coordinates are Virginia North State Plane, NAD27

Figure 11. Beaverdam Creek Reservoir map, from 2000 survey.

# Estimating Nonpoint Fecal Coliform Sources in Northern Virginia's Four Mile Run Watershed

George M. Simmons, Jr., Biology Department, Virginia Tech; Donald F. Waye, Northern Virginia Regional Commission (formerly the Northern Virginia Planning District Commission); Sue Herbein, Biology Department, Virginia Tech; Sharon Myers, Applied Statistics Laboratory, Radford University; Ellen Walker, Mathematics Department, Virginia Tech

E-mail: dwaye@novaregion.org

## ABSTRACT

Pulsed Field Gel Electrophoresis (PFGE) was conducted on *E. Coli* DNA from seasonally-varied stream and sediment samples in the ultra-urban Four Mile Run watershed in Northern Virginia. This study found:

1) without regard to specific host animals, *E. coli* bacteria seem to regrow, through cloning, within the storm drains and stream sediments, which in turn perpetuate elevated bacteria levels within the connected surface waters of Four Mile Run; 2) nonhuman species are the dominant sources of *E. coli* to Four Mile Run and its tributaries; 3) waterfowl contribute over one-third (37%) of those isolates that could be identified; 4) the presence of human *E. coli* is localized; 5) the predominant nonhuman sources are wildlife species that have intimate association with the waterways; 6) the major nonhuman mammal contributors are raccoon, dog, deer, and Norway rat; 7) the combined human and canine contribution is approximately 25% of those isolates that could be identified;

The continued presence of *E. coli* suggests an ecosystem out of balance irrespective of the source. It is neither desirable nor practical to eliminate wildlife animal species in the watershed. Rather, it is suggested that, wherever possible, nutrient loadings be controlled to restore a more balanced microbial community to the stream network.

**Keywords:** urban streams, bacteria, *E. coli*, Pulsed Field Gel Electrophoresis (PFGE), DNA, storm drains, regrowth, nonpoint source pollution

## INTRODUCTION

Since 1990, at least five separate organizations have cumulatively collected over 500 fecal coliform samples from the Four Mile Run watershed. Approximately 50% of these were found to have a Most Probable Number (MPN) greater than 1,000, which exceeds the state's water quality standard of fecal coliform density for the watershed (SWCB, 1997). Four Mile Run is listed as one of the streams on Virginia's 303(d) list of impaired stream segments because of the elevated levels of fecal coliform bacteria (Virginia DEQ, 1998). In addition to violating the fecal coliform standard, the Four Mile Run watershed is given a "high priority" ranking for potential nonpoint source pollution by the Virginia Department of Conservation and Recreation (Virginia DEQ and DCR, 1998), and is designated as a nutrient-enriched waterway by the State Water Control Board (1997).

In the 1992 re-authorization of the federal Clean Water Act, considerable emphasis was placed on developing watershed-based strategies that had potential to reduce nonpoint source pollution in impaired streams. The Northern Virginia Planning District Commission has initiated a phased approach for meeting the mandates of the Clean Water Act for Four Mile Run through a 604(b) Water Quality Grant to Virginia DEQ (NVPDC, 1998). This research serves as a starting point toward achieving this goal. The purpose of this research project was to determine potential animal sources for fecal coliform contamination of Four Mile Run and its tributaries in Northern Virginia.

Watershed Characteristics

The Four Mile Run watershed (12,600 acres, 19.7 square miles) is a densely populated urban watershed where the dominant land use is medium to high density residential housing. Approximately 165,000 people live in the watershed, resulting in a population density of 13 people per acre (over 8,000 people per square mile) (NVPDC, 1996a). There are two NPDES-permitted point source discharges in the watershed; a concrete batch plant near Shirlington and the Arlington Waste Water Treatment Plant (WWTP) near Route 1. The Arlington WWTP discharges into the tidal portion of Four Mile Run near its confluence with the Potomac River. There are no combined storm/sanitary sewer lines by design, and testing by NVPDC and Arlington County to determine the extent of cross-connections between the sanitary sewer system and the storm sewer system confirms the overall integrity of these separate sewer systems, with only minor problems occasionally discovered.

A very large pet population accompanies a very dense human population in the watershed. NVPDC staff has estimated the canine density of the watershed to be approximately one dog for every 10 people, resulting in a density of 1.3 dogs/acre (over 800 per square mile). NVPDC staff has further estimated that more than 2,400 kg (over 5,000 pounds) of fecal waste is deposited in the watershed on a daily basis, which is conservatively based on 150 g of solid waste per dog (one-third of a pound) [1.3 dogs/acre * 12,600 acres]. Besides humans and dogs, the watershed contains a variety of mammals and waterfowl that have adapted to an urbanized landscape.

## METHODS

Details of the sampling protocol and procedures related to Quality Assurance and Quality Control (QA/QC) are contained in a separate QA/QC Plan. Pulsed Field Gel Electrophoresis (PFGE) is a widely used technique to resolve microbial strain recognition in clinical and natural environments (Goering, 1993; Maslow, et al., 1993; Edberg, et al., 1994; Buchrieser, et al., 1995; Tynkkynen, et al., 1999). Details of isolate selection from DNA analyses are summarized in a separate document.

Sample Collection, Locations and Times

A total of 55 samples were collected in this study. These included samples from water column, water-sediment slurries, and sediment cores. The locations for the samples used in this study are presented in Figure 1. Station location and their respective identification numbers are presented in Table 1.

Four seasonally varied sampling periods were used to characterize potential nonpoint fecal coliform sources to the Four Mile Run watershed. These were: August 1998 (summer period); May 1999 (spring period); November 1999 (fall period); and February 2000 (winter period). In addition, fecal coliform density samples were taken in June 2000, but DNA results from this sampling period are not included in this study.

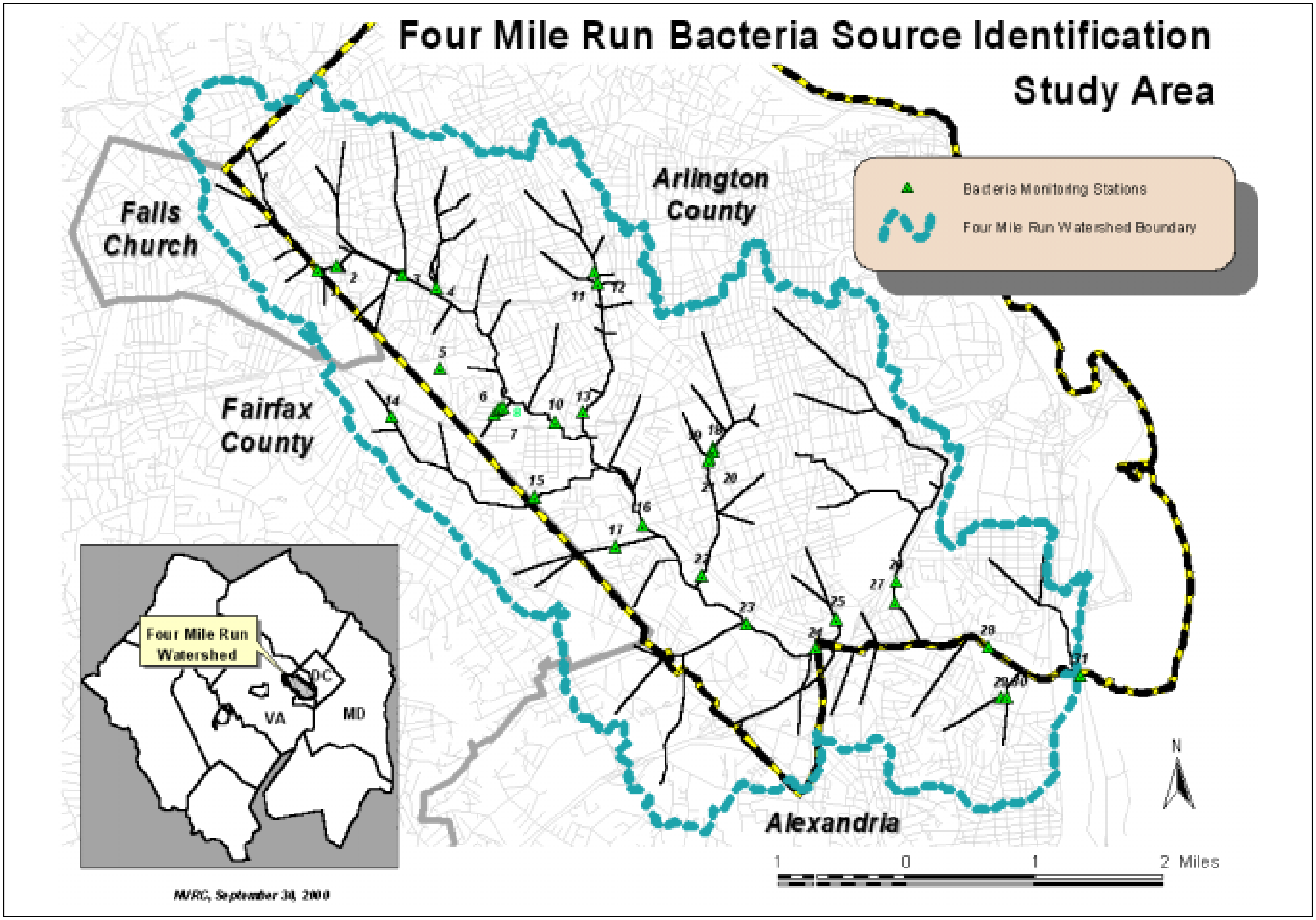



**Figure 1. Map of Four Mile Run Watershed with Sample Locations**

**Table 1.** Sample Locations and Identification Numbers

| I.D. | Location | Alternate I.D. |
|---|---|---|
| 1 | Upper Four Mile Run at Falls Church line (Van Buren Street) | NVPDC#7 |
| 2 | Upper Four Mile Run at Sycamore Street | |
| 3 | Ohio Street Branch at I-66 outfall | FM200 or FM210, Arlington |
| 4 | Westover Branch at I-66, outfall (twin box culvert to right of 2 m [78 in] circ.) | FM230, Arlington |
| 5 | Powhatan Run at N. Livingston Road, pristine site | u/s of FM300, Arlington |
| 6 | Manchester Street 1.1 m (42 in) outfall (Glencarlyn Branch) | FM 330, Arlington |
| 7 | 46 m (150 ft) downstream (d/s) of Manchester Street outfall | d/s of FM 330, Arlington |
| 8 | 91 m (300 ft) d/s of Manchester Street outfall | d/s of FM 330, Arlington |
| 9 | 137 m (450 ft) d/s of Manchester Street outfall | d/s of FM 330, Arlington |
| 10 | Middle Four Mile Run, bike trail crossing just u/s of Rt. 50 | NVPDC#6 |
| 11 | Ballston Beaver Pond, along open channel | Near LR112, Arlington |
| 12 | Box culvert under Ballston just d/s of Beaver Pond | |
| 13 | Lubber Run at Route 50 | NVPDC#5 |
| 14 | Upper Long Branch d/s of Patrick Henry Drive | |
| 15 | Upper Long Branch at Carlin Springs Road | NVPDC#4 |
| 16 | Four Mile Run at Columbia Pike | 1AFOU004.22, Va. DEQ |
| 17 | Baileys Branch at S. Frederick Street | FM350, Arlington |
| 18 | Doctors Run at S. 6th Street & S. Quincy Street, biggest outfall | DB100, Arlington |
| 19 | Doctors Run 61 m (200 ft) d/s of S. 6th Street & S. Quincy Street | d/s of DB100, Arlington |
| 20 | Doctors Run 122 m (400 ft) d/s of S. 6th Street & S. Quincy Street | d/s of DB100, Arlington |
| 21 | Doctors Run 183 m (600 ft) d/s of S. 6th Street & S. Quincy Street | d/s of DB100, Arlington |
| 22 | Doctors Run at Barcroft Park Footbridge | NVPDC#8 |
| 23 | Lucky Run outfall at Four Mile Run | NVPDC#3 |
| 24 | Four Mile Run at Shirlington Road | NVPDC#2 |
| 25 | Nauck Branch | FM450, Arlington |
| 26 | Lower Long Branch at I-395 near 28th Street S., outfall—quad box culvert | 274 m (900 ft) d/s of LL180, Arlington |
| 27 | Lower Long Branch in Arna Valley, 26th Street S. | NVPDC#1 |
| 28 | Arlington Sewage Treatment Plant outfall | FM545?, Arlington |
| 29 | Alexandria trib behind Cora Kelly Community Center, u/s of outfall | |
| 30 | Alexandria trib behind Cora Kelly Community Center, corrugated metal pipe outfall | |
| 31 | Four Mile Run at George Washington Parkway | 1AFOU000.19, Va. DEQ |

Statistical Comparison of Populations:

The $\chi^2$ Goodness-of-fit analysis for populations was used to test statistical differences between the *E. coli* clonal populations from the different animal groups based on their PFGE patterns. For these analyses, the entire banding profile (from 780-20 kilobase pairs) was divided into six equal units and the frequency of bands within each unit was used for comparative purposes at $\alpha = 0.10$. The percent of bands within each unit was also presented as a histogram in a separate document to visually display differences in banding patterns between *E. coli* populations of the different animal groups.

Computer-based Search of DNA Library:

The calculated numerical value of each band (molecular size as kb) was loaded into flat files with respect to each animal group. All animal groups were then combined to create a single library. A TCL computer program (Tool Command Language©, an embeddable scripting language, release 8.0p2; copyright by the Regents of the University of California, Sun Microsystems, Inc., and other parties) was used to compare *E. coli* strains from field samples with *E. coli* strains from known sources in our library. A band-to-band comparison was made and expressed as a percent similarity. The program allows the investigator to adjust the lower limit of percent comparison (i.e., 75%, 78%, 80%, etc.) between known and unknown strains, and the range of kilobase pairs used for each two bands being compared ( i.e. ± 5 kilobase pairs, ± 10 kilobase pairs, etc).

Libraries Used in This Study:

Several DNA libraries were used in this study. The libraries, their respective animal species, and number of PFGE patterns per species are listed in Table 2. The total number of strains used to determine potential animal sources was 843.

Assigning Potential Sources Based on DNA Profile Analysis:

In trying to assign a "best fit," the first factor considered was similarity as measured by the degree of correlation between the strain from an unknown source and a strain from a known animal in the Virginia Tech DNA library. For example, if the DNA bands from a strain of an unknown source matched 90% of the DNA bands with an *E. coli* strain from Canada Goose, and only 82% with a strain from a canine source, it would be concluded that the unknown strain was more likely to come from a Canada Goose because there was a higher correlation with the Canada Goose strain.

However, there were instances where a strain from an unknown source correlated with a human strain and a canine strain at the same similarity (88% for example). In this case, the library provided a match but it was not be possible to differentiate between canine and human. If, however, the unknown strain matched with several human strains and only one canine strain from the library, it was considered to be more likely to come from a human source based on the number of matches. Furthermore, there are fewer human strains in the Virginia Tech DNA library than canine, and if matches were random, then a greater number of canine matches would be expected. However, because *E. coli* from dogs and humans cannot be statistically separated by this methodology used in this study, it is not possible to conclude that the unknown strain is not from a canine source.

If an unknown strain was approximately equally similar to more than one animal group and the number of matches were also approximately equal among animal groups, a visual band-to-band comparison would be made to determine which animal group might be the more likely candidate. The presence or absence of matches in the heavier segments of DNA often provided clues as to the degree of greater similarity because there are many fewer bands in the 750-500 kilobase pair range than below this range.

Geography also played a role given that *E. coli* from known sources from several geographic areas were combined for this study, and given that there is very little known about geographic variability in *E. coli* PFGE patterns from the same animal species. Therefore, if the pattern from an unknown source matched an *E. coli* pattern from a goose in the Cornell library from the Long Island Sound area at 88%, but matched a raccoon strain from the Northern Virginia/Four Mile Run library at 84%, assignment to raccoon would probably be made, assuming a spurious correlation with the goose, and a more likely correlation with the raccoon.

Source ecology also played a factor in assigning most likely sources. In a situation where the strain from an unknown source matched approximately equally with a horse collected from scat in the Rappahannock basin, a raccoon from Northern Virginia, and a pelican from the Chesapeake Bay, it would be concluded that the unknown strain was most likely from the raccoon simply because horses and pelicans are far less common in the study watershed. Another example of the way ecology was considered is a situation of similar correlation with strains from a canine source in the Cornell library and a Norway rat from the Northern Virginia/Four Mile Run library. There are very few Norway rat samples in the Virginia Tech DNA library and the fact that the unknown strain of *E. coli* matched a Norway rat strain was a compelling reason to assign a likely match.

However, in some cases source assignments were unclear regardless of consideration of the factors described above. For example, if a strain from an unknown source matched with an *E. coli* strain from bovine (Dr. Eugene Yagow's library from Virginia's Rappahannock basin), and that was the only match, then that animal was assigned as the possible source. In this particular case, there are several possible theories for such a match. First, the match of the unknown strain to a bovine source could be spurious because there are no known bovines living in the Four Mile Run Watershed. A second theory is that the unknown strain could be a crossover strain of *E. coli* common to multiple animal groups, perhaps picked up by birds feeding on insect larvae in bovine dung, passed through the bird's digestive tract, and deposited in the watershed by the birds while in transit. A third possibility is that the match might be correct and the data could suggest that *E. coli* from bovine are somehow making their way into the watershed through a presently unknown transport mechanism (such as leachate from restaurant dumpsters). A fourth explanation is that because the *E. coli* populations of bovine and deer are not statistically different from each other (possibly due to the complex ruminant digestive system that each animal groups possesses) the bovine signatures may be serving as surrogates for deer *E. coli*.

**TABLE 2.** Numbers of Isolates from the Different Libraries Used in the Analysis of Potential Fecal Coliform Sources From Study Area Locations

(All library samples maintained by Virginia Tech, $n = 843$)

Eastern Shore/Chesapeake Bay Library (collected 1994 – 1997):

| | |
|---|---|
| Muskrat | 34 |
| Raccoon | 71 |
| Deer | 39 |
| Beaver | 20 |
| Otter | 22 |
| Human | 67 |
| Canine | 42 |
| Laughing Gull | 29 |
| Herring Gull | 33 |
| Pelican | 7 |
| Tern | 16 |
| Canada Goose | 45 |
| Wood Duck | 3 |
| Merganser | 5 |
| Porcine | 15 |
| *Total* | *448* |

Cornell Long Island Sound Library (collected 1994 – 1997):

| | |
|---|---|
| Human | 7 |
| Raccoon | 54 |
| Deer | 25 |
| Canine | 21 |
| Horse | 25 |
| Herring Gull | 24 |
| Black Back Gull | 16 |
| Canada Goose | 14 |
| Black Duck | 5 |
| Mallard Duck | 9 |
| Mute Swan | 14 |
| Mallard Duck | 11 |
| Teal | 5 |
| Black Duck | 26 |
| *Total* | *256* |

Four Mile Run (Northern Va) Library* (collected 1999 – 2000):

| | |
|---|---|
| Red Fox | 5 |
| Raccoon | 16 |
| Flying Squirrel | 3 |
| Gray Squirrel | 5 |
| Opossum | 7 |
| Canine | 27 |
| Norway Rat: | 6 |
| Feline | 5 |
| Human | 8 |
| Seagull | 4 |
| Canada Goose | 8 |
| *Total* | *94* |

Yagow (Rappahannock basin) Library (collected 1998 – 1999):

| | |
|---|---|
| Muskrat | 1 |
| Raccoon | 1 |
| Deer | 3 |
| Beaver | 1 |
| Canine | 8 |
| Horse | 8 |
| Bovine | 22 |
| Canada Goose | 1 |
| *Total* | *45* |

* Number of isolates does not correspond with the number of scat samples collected for this study because some samples contained multiple strains of *E. coli* and other samples lacked viable *E. coli*.

# RESULTS

## Fecal Coliform Densities

Sample locations and results of fecal coliform densities are presented in Table 3. Stormwater outfalls, fine sediments, and samples of microbial films from sediment/water mixture samples tended to have the higher densities. Most Probable Number (MPN) values of ≥1600 were scored as numerical values of 1700 for purposes of calculation.

## DNA Profiles (PFGE Patterns) From Four Mile Run and Its Tributaries

A total of 539 bacterial isolates were removed from 55 samples of either water, a water/sediment mix, or sediment from Four Mile Run and its tributaries during this study period. Of the 539 isolates that were removed for DNA profile analysis, 100 of these could not be analyzed for reasons of taxonomic or restriction failure. The remaining 439 isolates keyed to *Escherichia coli* (*E. coli*) using the Analytical Profile Index (API 20E) for the Enterobacteriaceae and other gram negative bacteria provided the basis for resolving potential animal sources that could contribute to the nonpoint fecal coliform problem in Four Mile Run and its tributaries. Of the 439 isolates, 133 showed no match at 80% similarity ± 10 kilobase pairs (kbp) with any of the 843 strains of *E. coli* from known sources in the Virginia Tech DNA library (Table 2). Twenty-eight (28) isolates from the study matched at equal similarity with multiple strains in the Virginia Tech DNA library, but were inconclusive with regard to a specific species. However, within this group of 28 isolates, all suggested a nonhuman source, and nearly all suggested a nonhuman mammal source. The remaining 278 isolates did show a match at 80% similarity ± 10 kbp with a particular animal species in the library. Data in Figure 3 and Table 3 summarize these matches. Some isolates experienced taxonomic and restriction failure and others were inconclusive with regard to potential animal source. Table 4 summarizes these results.

## Table 3. Fecal Coliform Densities at Study Area Locations

| | | | Fecal Coliform, MPN | | | | |
|---|---|---|---|---|---|---|---|
| | I.D. | Alternate Station I.D. | Water | Water/ Sed. | Sedi- ment | Digital Latitude | Digital Longitude |
| **28-Aug-98** | | | | | | | |
| *Note: Drought conditions* | | | | | | | |
| 1) Lower Long Branch in Arna Valley, 26th Street S. | **27** | NVPDC#1 | 2 | | | 38.8484 | -77.0748 |
| 2) Four Mile Run at Shirlington Road | **24** | NVPDC#2 | 900 | | | 38.8431 | -77.0861 |
| 3) Lucky Run outfall at Four Mile Run | **23** | NVPDC#3 | 500 | | | 38.8456 | -77.0962 |
| 4) Upper Long Branch at Carlin Springs Road | **15** | NVPDC#4 | ≥1600 | | | 38.8587 | -77.1268 |
| 5) Lubber Run at Route 50 | **13** | NVPDC#5 | 500 | | | 38.8678 | -77.1201 |
| 6) Middle Four Mile Run, bike trail crossing just u/s of Rt. 50 | **10** | NVPDC#6 | 1600 | | | 38.8668 | -77.1242 |
| 7) Upper Four Mile Run at Falls Church line (Van Buren Street) | **1** | NVPDC#7 | 900 | | | 38.8825 | -77.1589 |
| 8) Doctors Run at Barcroft Park footbridge | **22** | NVPDC#8 | 900 | | | 38.8507 | -77.1028 |
| *9) Donaldson Run at Military Road (outside of study area)* | *n/a* | | *500* | | | *38.9111* | *-77.1134* |
| *10) Gulf Branch at Military Road (outside of study area)* | *n/a* | | *1600* | | | *38.9193* | *-77.1199* |
| | | | | | | | |
| | | | | | | | |
| **06-May-99** | | | | | | | |
| *Note: Drought conditions* | | | | | | | |
| 1) Ballston Beaver Pond, along open channel (Lubber Run) | **11** | Near LR112, Arlington | 900 | | | 38.8831 | -77.1190 |
| 2) Powhatan Run at N. Livingston Road, pristine site | **5** | u/s of FM300, Arlington | 50 | | | 38.8722 | -77.1408 |
| 3) Manchester Street 1.1 m (42") outfall (Glencarlyn Branch) | **6** | FM 330, Arlington | ≥1600 | | | 38.8675 | -77.1330 |
| 4) Four Mile Run at Shirlington Road | **24** | NVPDC#2 | 1600 | | | 38.8431 | -77.0861 |
| 5) Lucky Run outfall at Four Mile Run | **23** | NVPDC#3 | 500 | | | 38.8456 | -77.0962 |
| 6) Four Mile Run at Columbia Pike | **16** | 1AFOU004.22, Va. DEQ | 900 | | | 38.8561 | -77.1112 |
| | | | | | | | |

| **Table 3.** *(continued)* | | | Fecal Coliform, MPN | | | | |
|---|---|---|---|---|---|---|---|
| | I.D. | Alternate Station I.D. | Water | Water/ Sed. | Sedi- ment | Digital Latitude | Digital Longitude |
| **23-Nov-99** | | | | | | | |
| 1) Upper Long Branch downstream of Patrick Henry Drive | 14 | | 80 | 170 | 80 | 38.8669 | -77.1478 |
| 2) Upper Four Mile Run at Sycamore Street | 2 | | 30 | 300 | 30 | 38.8830 | -77.1561 |
| 3) Box culvert under Ballston just downstream of Beaver Pond | 12 | | 900 | 500 | | 38.8818 | -77.1185 |
| 4) Lubber Run at Route 50 | 13 | NVPDC#5 | 50 | 220 | 30 | 38.8678 | -77.1201 |
| 5) Four Mile Run at Columbia Pike | 16 | 1AFOU004.22, Va. DEQ | 240 | | 30 | 38.8561 | -77.1112 |
| 6) Doctors Run at Barcroft Park footbridge | 22 | NVPDC#8 | 80 | | 30 | 38.8507 | -77.1028 |
| 7) Lucky Run outfall at Four Mile Run | 23 | NVPDC#3 | 900 | | | 38.8456 | -77.0962 |
| 8) Four Mile Run at Shirlington Road | 24 | NVPDC#2 | 300 | | 22 | 38.8431 | -77.0861 |
| 9) Lower Long Branch in Arna Valley, 26th Street S. | 27 | NVPDC#1 | ≥1600 | | 33 | 38.8484 | -77.0748 |
| 10) Four Mile Run at George Washington Parkway | 31 | 1AFOU000.19, Va. DEQ | 130 | | | 38.8409 | -77.0478 |
| | | | | | | | |
| **22-Feb-00** | | | | | | | |
| 1) Ohio Street Branch at I-66, outfall | 3 | FM200 or FM210, Arlington | 50 | 900 | | 38.8822 | -77.1467 |
| 2) Westover Branch at I-66, outfall (twin box culvert to right of 2 m [78"] circular pipe) | 4 | FM230, Arlington | ≥1600 | ≥1600 | ≥1600 | 38.8810 | -77.1417 |
| 3) Powhatan Run at N. Livingston Road (pristine site) | 5 | u/s of FM300, Arlington | 23 | 280 | | 38.8722 | -77.1408 |
| 4) Manchester Street 1.1 m (42") outfall (Glencarlyn Branch) | 6 | FM 330, Arlington | 900 | ≥1600 | | 38.8675 | -77.1330 |
| 5) Baileys Branch at S. Frederick Street | 17 | FM350, Arlington | 80 | 300 | | 38.8536 | -77.1152 |
| 6) Four Mile Run at Columbia Pike | 16 | 1AFOU004.22, Va. DEQ | 130 | 500 | 80 | 38.8561 | -77.1112 |
| 7) Doctors Run at S. 6th Street & S. Quincy Street, biggest outfall | 18 | DB100, Arlington | 1600 | ≥1600 | | 38.8645 | -77.1014 |
| 8) Lucky Run outfall at Four Mile Run | 23 | NVPDC#3 | 500 | ≥1600 | | 38.8456 | -77.0962 |
| 9) Nauck Branch | 25 | FM450, Arlington | 500 | 1600 | 1600 | 38.8464 | -77.0832 |
| 10) Lower Long Branch at I-395 near 28th Street S., outfall--quad box culvert | 26 | 274 m (900') d/s of LL180, Arlington | 2 | 21 | 500 | 38.8506 | -77.0748 |
| 11) Arlington Sewage Treatment Plant outfall | 28 | FM545?, Arlington | 0 | | | 38.8438 | -77.0613 |
| 12) Four Mile Run at George Washington Parkway | 31 | 1AFOU000.19, Va. DEQ | 14 | 300 | | 38.8409 | -77.0478 |

| **Table 3.** *(continued)* | | | Fecal Coliform, MPN | | | | |
|---|---|---|---|---|---|---|---|
| | I.D. | Alternate Station I.D. | Water | Water/ Sed. | Sedi- ment | Digital Latitude | Digital Longitude |
| **19-Jun-00** | | | | | | | |
| *Note: Samples from June 19, 2000 at Stations 5 - 12 were taken at 5 minute intervals at all four stations approximately simultaneously (in late morning). DNA results for June 19 not available for this study.* | | | | | | | |
| 1) Alexandria trib behind Cora Kelly Community Center, CMP outfall | **30** | | 900 | | | 38.8383 | -77.0584 |
| 2) Alexandria trib behind Cora Kelly Community Center, upstream of outfall | **29** | | ≥1600 | | | 38.8383 | -77.0594 |
| 3) Arlington Sewage Treatment Plant outfall | **28** | FM545?, Arlington | 0 | | | 38.8438 | -77.0613 |
| 4) Four Mile Run at Columbia Pike | **16** | 1AFOU004.22, Va. DEQ | 1600 | | | 38.8561 | -77.1112 |
| 5) Doctors Run at S. 6th Street & S. Quincy Street, biggest outfall | **18** | DB100, Arlington | **≥1600, ≥1600, ≥1600** | | | 38.8645 | -77.1014 |
| 6) Doctors Run 61 m (200 ft) downstream of S. 6th Street & S. Quincy Street | **19** | d/s of DB100, Arlington | **900, ≥1600, 900** | | | 38.8640 | -77.1015 |
| 7) Doctors Run 122 (400 ft) d/s of S. 6th Street & S. Quincy Street | **20** | d/s of DB100, Arlington | **500, 900, 500** | | | 38.8635 | -77.1019 |
| 8) Doctors Run 183 (600 ft) d/s of S. 6th Street & S. Quincy Street | **21** | d/s of DB100, Arlington | **900, 300, 900** | | | 38.8630 | -77.1022 |
| 9) Manchester Street, 1.1 m (42 in) outfall | **22** | FM 330, Arlington | **900, 500, ≥1600** | | | 38.8675 | -77.1330 |
| 10) 46 m (150 ft) d/s of Manchester Street outfall | **23** | d/s of FM 330, Arlington | **≥1600, 1600, ≥1600** | | | 38.8677 | -77.1325 |
| 11) 91 m (300 ft) d/s of Manchester Street outfall | **24** | d/s of FM 330, Arlington | **1600, 1600, ≥1600** | | | 38.8680 | -77.1321 |
| 12) 137 m (450 ft) d/s of Manchester Street outfall | **25** | d/s of FM 330, Arlington | **1600, 900, ≥1600** | | | 38.8682 | -77.1317 |

## TABLE 4. Number of Isolates by DNA Match with Best Species

| | FIELD DATES | | | | |
|---|---|---|---|---|---|
| **Animal Species** | **28Aug98** | **6May99** | **23Nov99** | **22Feb00** | **TOTALS** |
| False Positives | 0 | 37 | 4 | 11 | 52 |
| No API Code | 3 | 1 | 31 | 2 | 37 |
| No Restriction | 3 | 3 | 3 | 2 | 11 |
| No Matches | 18 | 9 | 67 | 39 | 133 |
| | | | | | |
| Human | 9 | 11 | 11 | 15 | 46 |
| Raccoon | 4 | 5 | 22 | 11 | 42 |
| Canine | 1 | 0 | 10 | 13 | 24 |
| Deer | 10 | 0 | 1 | 18 | 29 |
| Bovine | 0 | 0 | 3 | 10 | 13 |
| Norway Rat | 10 | 0 | 0 | 1 | 11 |
| Feline | 0 | 0 | 3 | 0 | 3 |
| Opossum | 0 | 0 | 0 | 3 | 3 |
| Beaver | 0 | 0 | 1 | 0 | 1 |
| Muskrat | 0 | 0 | 1 | 0 | 1 |
| Herring Gull | 6 | 18 | 1 | 0 | 25 |
| Mallard Duck | 0 | 18 | 13 | 1 | 32 |
| Black Duck | 0 | 0 | 6 | 2 | 8 |
| Laughing Gull | 8 | 0 | 1 | 0 | 9 |
| Canada Goose | 8 | 0 | 8 | 3 | 19 |
| Black Back Gull | 5 | 0 | 1 | 0 | 6 |
| Tern | 0 | 0 | 3 | 3 | 6 |
| Undetermined | 4 | 8 | 8 | 8 | 28 |
| ***TOTALS*** | ***89*** | ***110*** | ***198*** | ***142*** | ***539*** |

**Isolates Analyzed:**

133 No Matching Records
52 False Positives
37 No API Code
11 Failed Restriction
28 Inconclusive Identification
278 Acceptable Matches
***539 Total Number of Isolates Considered***

**Acceptable Matches:**

| | |
|---|---|
| 46 | Human |
| 42 | Raccoon |
| 29 | Deer |
| 24 | Canine |
| 13 | Bovine |
| 11 | Norway Rat |
| 8 | Other Mammals |
| 105 | Waterfowl |
| ***278*** | ***Total*** |

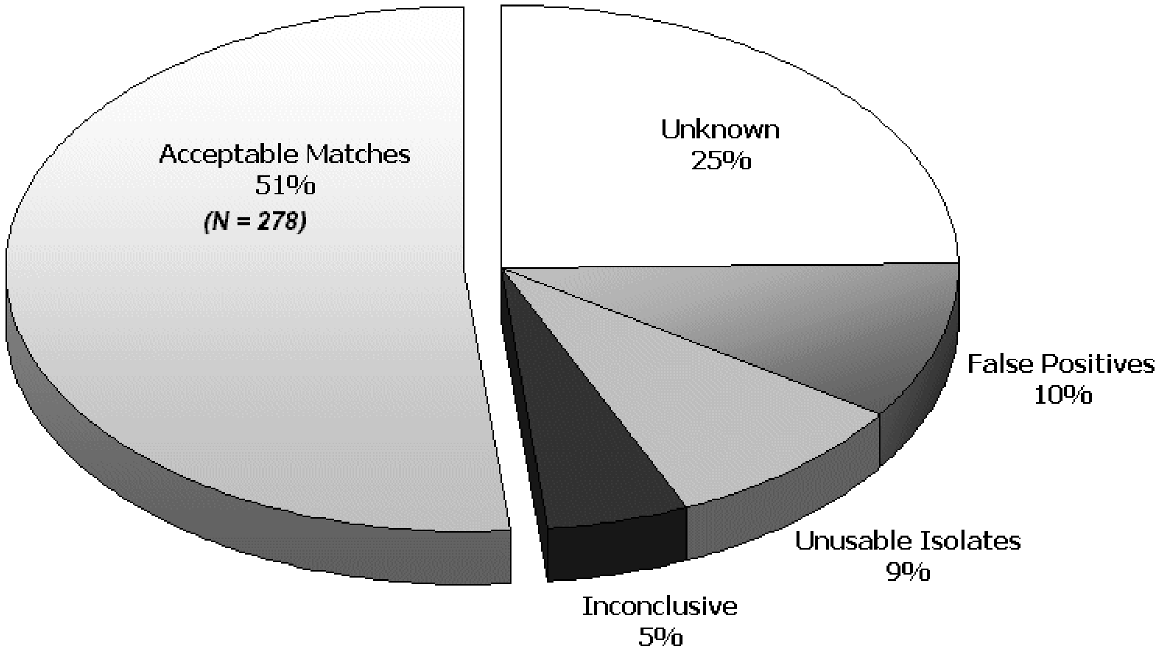



**Figure 2. Success of Isolate Matching, N = 539**

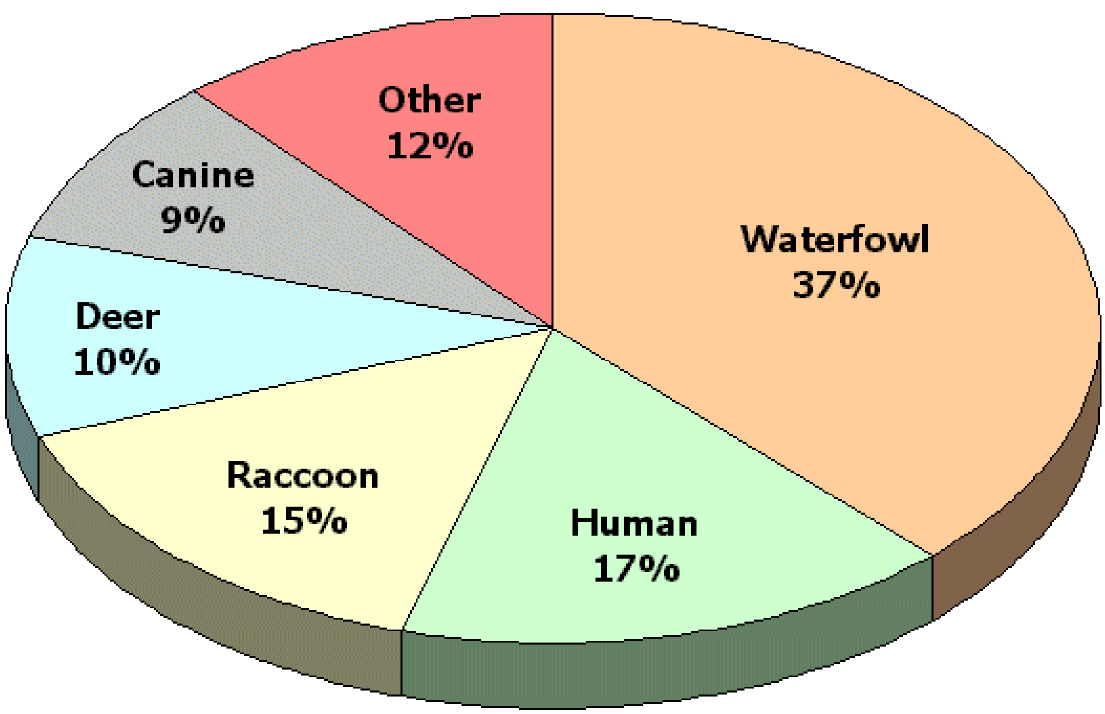



**Figure 3. Distribution of Acceptable Matches by Animal Group, N = 278**

## DISCUSSION

Is the major source of nonpoint fecal coliform contamination human or non -human in origin?

The data suggested, that on the basis of the 278 isolates which did show one or more matches with strains of *E. coli* from known sources, potential contribution from human sources wa s moderate. Forty-six (46) isolates (17%) were considered to be of human origin, whereas 232 isolates (83%) were considered to be of nonhuman origin. The potential contribution from human sources ranged between 13 -21% for all four seasonal sampling peri ods.

Is the human source localized?

The data suggested that possible contributions from human sources were localized. In particular, stations associated with Doctors Run (Feb '00, 13 isolates), Four Mile Run at Columbia Pike (Nov '99, 6 isolates), Donal dson Run at Military Road (Aug '98, 9 isolates), and Lucky Run (May '99, 11 isolates) suggested potential inputs of *E. coli* from human sources. Human signatures were not suggested at any of the other collecting sites.

Is the nonhuman source mammal or avi an in origin?

As stated above, 232 isolates were identified as being of nonhuman origin. Of this pool (232 isolates), the data suggested that 127 isolates (55%) were from a mammalian source and 105 isolates (45%) were from one or more species of waterfow l (geese, gulls, and ducks).

Is the major mammal contribution from domestic or wild animal species?

Several animals stand out in the mammal group. Of the 127 isolates attributed to nonhuman mammal sources, raccoon were the most dominant representative of the group with 42 isolates (33%) being represented; deer were second with a total of 42 isolates (33%) (assuming that the bovine isolates served as surrogates for deer, and for this reason deer are listed as the second group); canine isolates were third (24 isolates - 19%); and the Norway rat was fourth with 11 isolates (9%). Feline (3 isolates -2 %); opossum (3 isolates - 2%); beaver (1 isolate -1 %); and, muskrat (1 isolate -1 %) comprised the remaining matches. These data suggested that wild animal species, rather than domestic animal species, contributed the greater percentage of fecal coliform isolates to Four Mile Run and its tributaries.

The fact that deer signatures were much more frequent than would have been suspected can be explained in several ways. One explanation has to do with frequency of occurrence of isolates, and the other explanation deals with assignment to a particular source. In the August 1998 samples, all ten isolates at Station 7 had the same profile. Assignment was made to "deer" as a result of band-to-band comparisons, but herring gull was a strong second choice. In the Feb '00 samples, all 10 isolates from Station 4 showed the same identical profile and, again, band -to-band comparisons suggested a "deer" signature, but B lack Back Gull, raccoon, and canine were also possible choices. Stations 8 and 10 each had one isolate that suggested "deer," but muskrat and Canada goose were also reasonable choices. At Station 2, however, five isolates all had the same pattern, and "deer" was the only match suggested. Even if the other possible choices are considered, except in one case, the alternate choice is a wild animal source.

At the present time, the most limiting aspect of this research effort, aside from the modest size of the library, is the fact that canine and human *E. coli* populations cannot be separated statistically, despite this study's efforts to expand the source library for these two species. Whether this is a limitation of the size of the populations being tested, or if in fact *E. coli* do move freely between these two animal groups, remains to be investigated further. However, of the total pool of identifiable isolates, only 70 isolates (25%) could be assigned to human or canine and 208 (75%) isolates were assigned to wild animal sources.

The subject of urban wildlife ecology is still in its infancy and much still remains to be understood about the relationship of certain wildlife species to expanding urban environments (Murphy 1988).

The data **do not suggest** that there were more wildlife individuals in the watershed than canine or human individuals. The data **do suggest** that certain wildlife species have a greater, disproportionate, representation and effect on fecal coliform density in the watershed because of their direct contact and intimate association with the waterways. Furthermore, the frequency of occurrence of a wild animal species is not necessarily occur in direct relationship to the frequency of occurrence of their fecal coliform signature. Survival and regrowth of specific strains from a given animal also have to be considered as well as the specific time of collection.

The conclusion, suggested from the data in this study, that wildlife animal sources were a major contributor to the fecal coliform problem, has also been corroborated by fecal coliform studies in tidal creeks and estuaries in the southern Chesapeake Bay (Simmons, 1994; Simmons and Herbein, 1995; Simmons, et al, 1995; Herbein et al, 1996). Furthermore, the data are also consistent with other observations (anecdotal information from Simmons and coworkers). For example, waterfowl scat generally has less *E. coli* than mammal scat and strain diversity is generally lower. Also, sampling in small water bodies with large numbers of waterfowl present can show a low fecal coliform density (Simmons, unpublished data from research on Virginia's Eastern Shore, 1995-1997).

What is the role of sediments?

Two sampling periods (November 1999, and February 2000) focused on the contribution of water/sediment slurries and sediments to the fecal coliform problem. The MPN geometric mean in November for the fecal coliform densities in water was 149.3; for water/sediment slurries 239.7; and, for sediments 32.6. Estimates of sediment MPN density for this period consisted of adding 1 gm of sediment in 99 mls of buffered water, and the sediments consisted of very coarse sand and/or gravel. Some of the water/sediment slurries came from inside stormwater pipes and contained little/no sediment. These data suggested that the greatest number of fecal coliforms existed in the water column and as a microbial film attached to substrate.

This exercise was repeated in February 2000. At this time, the composition of the sediments and amounts added to buffered water was different than in the November exercise. In February, two samples of very fine sediments were collected at each stormwater outfall and 1.0 gm was added to 100 ml of buffered water. In two other samples, 6.0 and 15.0 gms of sediment were added to the buffered water because the sediments were so coarse that it was not possible to weigh out 1.0 gram exclusive of residual water in the syringe. The MPN geometric mean in February for the fecal coliform densities in water was 132.3; for water/sediment slurries 592.9; and for sediments 574.3.

The role of sediments as potential reservoirs has been documented by other researchers (Van Donsel and Geldreich, 1971; Gerba and McLoed, 1976; Hood and Ness, 1982; Stephenson and Rychert, 1982; Sherer, et al., 1992; Davies, et al., 1995; and, Reay, 2000). The February data showed that microbial films and sediments can serve as reservoirs and potentially contribute to the nonpoint fecal coliform problem in Four Mile Run. This contribution could be through the addition of cells to the water column from regrowth of either microbial films or from the sediments. Contributions through regrowth and subsequent sampling of clonal populations from the water column could explain the low strain diversity found by this investigation in many of the samples collected from stormwater outfalls.

What is the role of false positives?

False positives are those bacteria that also are characterized as part of the Enterobacteriaceae along with *E. coli*. False positive species not only inhabit the intestinal tract of animals along with *E. coli*, but also they may occur as free-living organisms in aquatic systems as well. In routine examination of freshwaters using gas formation as a method of identification, these other Enterobacteriaceae species may give a "false positive" reading. Therefore, in trying to determine nonpoint *E. coli* sources, detailed identification of isolates must be made to rule out the presence of non-*E. coli* species, or "false positives."

The role of false positives was not as significant in the final analysis of sources as originally believed, and the data suggested that false positives contributed only in a minor way to the overall nonpoint fecal coliform source question. However, in some cases and based on the number of isolates removed at random, the data suggested that false positives could be significant in isolated or localized situations. For example, at Station 3 in the May 6, 1999 sampling period, the 20 isolates removed for restriction analysis were all *Citrobacter freundii*. Likewise, on the same date at Station 6, 16 of the 20 isolates removed were *Enterobacter cloacae*. At Station 6 for the February 22, 2000 sampling, five of the 10 isolates removed were *C. freundii*. Even though the data suggested that false positives occurred at a low density level, they did contribute to the overall fecal coliform density.

Of the 539 isolates removed from samples for restriction analysis, 89 isolates (17%) fell into the category of "false positives" or "unidentified API profile." Of these 89 isolates, 55 isolates were identified with the API profile system to be *C. freundii*, *E. cloacae*, *Kluyvera*, spp, *Klebsiella pneumoniae*, or *K. ozaenae*. Of these taxonomic groups, *C. freundii* and *E. cloacae* comprised the greatest number of isolates (29 and 18, respectively) that were encountered in the "false positive" fecal coliform group.

Is there any seasonal variation?

No discernable pattern of seasonal variation among acceptable human or non-human matches was evident in this study. In fact, contrary to conventional wisdom, even the density of fecal coliforms seemed just as elevated during the winter sampling period as during the warmer months. This may point to a role for storm drains, which have been previously documented to moderate baseflow temperatures within Four Mile Run (NVRC, 1996b).

What is the effect of baseflow drainage through storm drains?

Two-thirds of the watershed's original stream network has been converted to underground drainage, primarily in its headwaters. The data collected from storm drains definitely suggested that drainage from these conduits during baseflow periods contributed significantly to the fecal coliform problem in Four Mile Run and its tributaries. For example, the MPN geometric mean of fecal coliform densities in open stretches of Four Mile Run and its tributaries was 231.1 (N=23); whereas, the MPN geometric mean of fecal coliform densities from stormwater outfalls during the same period was 400.2 (N=11).

In June 2000 a study was conducted at two stormwater outfalls (Doctors Run and Manchester Street) to determine the degree to which fecal coliform density from the outfalls diminished with distance downstream. The distance downstream from each outfall was approximately 100 meters. The fecal coliform density at the Doctors Run outfall was $\geq 1600$ and had decreased to a geometric mean of 624.0 at the downstream sampling point. At the Manchester Street outfall, the geometric mean of the fecal coliform density at the outfall was 914.5 but the density increased to a geometric mean of 1347.7 at the downstream sampling point. In the latter case, given the range of density associated with MPN values, the data demonstrate that there was little/no removal of fecal coliform density within the 100 meter stretch and that the open water portion of the stream was influenced by the discharge from the stormwater line. In the former case (Doctors Run), the data suggest that, while the stream had some filtration capacity to reduce fecal coliform densities, the density in the stream was also influenced by the stormwater discharge.

The influence of storm drains on the fecal coliform problem can be explained in two possible ways. First, the density of animal scat in the storm drains may provide a constant source of fecal coliforms as the water passes over the scat deposits. Second, and a more likely explanation, is that scat material is deposited in the storm drains, fecal coliforms are transported from the scat, become deposited in the storm drains, re-grow, and contribute to the microbial film found in the storm drains. Clonal populations lift-off, or are scoured by the moving water, and provide a continuous source, or inoculation, of fecal coliforms to the discharging water.

The importance of regrowth has been investigated by Simmons and his students (Carey and Simmons, 1995) in relation to discharge from a poultry processing plant on Virginia's Eastern Shore. Sediments are also important reservoirs for fecal coliform introduction to surface waters as noted by other investigators (cited above). Additional water chemistry data from Four Mile run and its tributaries (Northern Virginia Planning District Commission, 1996b) indicate that sufficient quantities of nutrients and carbon are available to support regrowth in the storm drains.

Additional information related to water quality in Four Mile Run (Harms and Southerland (1975); Randall, et al. (1978); and, Environmental Systems Analysis, Inc (1999) corroborates the importance to storm drains. In a study of the Upper South River Basin near Waynesboro, Virginia, Harms and Southerland (1975) documented the contribution of nutrient and sediment loading from urban runoff and concluded that controls were necessary to protect water quality in the river. Randall et al. (1978) working in the Occoquan watershed also noted the serious negative impact of urban stormwater runoff on stream water quality. Environmental Systems Analysis, Inc. (1999) completed a baseline macroinvertebrate assessment of Four Mile Run and found that the substrate at most sampling sites showed dominance of a few pollution-tolerant macroinvertebrates, and stations characterized by high levels of algal growth (evidence of nutrient loading), sedimentation, and erosive flows from high storm drain discharges during wet weather.

## SUMMARY

Based on the interpretation of DNA profile analyses of pulsed field gel electrophoresis patterns for those *E. coli* isolates from Four Mile Run and its tributaries that could be matched with *E. coli* strains from known sources in the Virginia Tech library; and, from fecal coliform densities of water, water/sediment slurries, and sediment, the data suggested the following:

1. without regard to specific host animals, *E. coli* bacteria seem to regrow, through cloning, within the storm drains and stream sediments, which in turn perpetuate elevated bacteria levels within the connected surface waters of Four Mile Run;
2. nonhuman species are the dominant sources of *E. coli* to Four Mile Run and its tributaries;
3. waterfowl contribute over one-third (37%) of those isolates that could be identified;
4. the presence of human *E. coli* is localized;
5. the nonhuman sources are wildlife species that have intimate association with the waterways;
6. the predominant nonhuman mammal contributors are raccoon, dog, deer, and Norway rat;
7. the combined human and canine contribution is approximately 25% of those isolates that could be identified;
8. the organisms contributing to the fecal coliform presence are those animals which would normally be expected in an urban watershed;
9. discharge from storm drains during baseflow play a significant role in the fecal coliform problem;

The data **do not suggest** there were more wildlife individuals in the watershed than canine or humans, but the data **do suggest** that certain wildlife species may have a greater, disproportionate, representation in the DNA profile analysis because of their direct contact and intimate association with the waterways. The DNA profile analysis is not a tool for estimating population density of any given species, but it may be an excellent method to identify those animals that have an impact on water quality.

It is neither desirable nor practical to eliminate wildlife animal species in the watershed. Ecologically speaking, the microbial community, including *E. coli*, is doing what heterotrophic microorganisms do – absorb nutrients and decompose organic compounds. The continued presence of *E. coli* suggests an ecosystem out of balance irrespective of the source.

While the citizens of Four Mile Run and those governmental agencies whose job it is to oversee and improve water quality in Four Mile Run deserve considerable credit for improving water quality in Four Mile Run and its tributaries, much remains to be done to reduce nutrient loading which may contribute to the regrowth of those *E. coli* which make their way into the waterways.

## REFERENCES

Adams, M.R., SM Grubb, A Hamer, and MN Clifford. 1990. Colorimetric enumeration of *Escherichia coli* based on β-glucuronidase activity. Appl. Environ. Microbiol 56:2021-2024.

American Public Health Association, American Water Works Association, and Water Environment Federation. 1992. Standard Methods for the Examination of Water and Wastewater. Eighteenth Edition. American Public Health Association, 1015 Fifteenth Street, NW, Washington, DC 20005.

Buchrieser, C, VV Gangar, RL Murphree, ML Tamplin, and CW Kaspar. 1995. Multiple *Vibrio vulnificus* strains in oysters as demonstrated by clamped homogenous electric field gel electrophoresis. Appl. and Environ. Microbiol. 61: 1163-1167.

Carey, J and GM Simmons, Jr. 1995. The use of DNA technology to predict nonpoint fecal coliform sources. Water Resources Research Conference, Richmond, Virginia.

Chang, G., J Brill, and R Lum. 1989. Proportion of β-D-glucuronidase-negative Escherichia coli in human fecal samples. Appl. Environ. Microbiol. 55:335 -339.

Davies, CM, JAH Long, M Donald, and NJ Ashbolt. 1995. Survival of fecal microorganisms in marine and freshwater sediments. Appl. Environ. Microbiol. 61: 1888-1896.

Edberg, SC, JE Patterson, and DB Smith. 1994. Differentiation of distribution systems, source water, and clinical coliforms by DNA analysis. J. Clin. Microbiol. 32:139 -142.

Environmental Systems Analysis, Inc. 1999. Appendix C – Baseline Monitoring (part of a survey of macrobenthic diversity in Four Mile Run).

Feng, P.C.S. and PA Hartman. 1982. Fluorogenic assays for immediate confirmation of *Escherichia coli*. Appl. Environ. Microbiol. 43: 1320 -1329.

Gerba, CP and JS McLoed. 1976. Effect of sediments on the survival of *Escherichia coli* in marine waters. Appl. Environ. Microbiol. 32: 114 -120.

Goering, R. 1993. Molecular epidemiology of nosocomical infection: Analysis of chromosomal restriction fragment patterns by pulsed -field gel electrophoresis. *Infect. Control Hosp. Epidemiol. 14: 595-600.*

Hadidian, J, DA Manski, and S Riley. 1991. Daytime resting site selection in an urban raccoon population, pp. 39-45. *In* LW Adams and DL Leedy (eds*), Wildlife Conservation in Metropolitan Environments* . National Institute for Urban Wildlife, Symposium, Ser.2, 10921 Trotting Ridge Way, Columbia, MD, 21044.

Hadidian, J, GR Hodge, and JW Grandy (eds). 1997. Wild Neighbors. Humane Society of the United States, Fulcrum Publishing, 350 Indiana Street, Suite 350, Golden, Colorado 80401 - 5093.

Harms, LL and EV Southerland. 1975. A case study of non -point source pollution in Virginia. Bull. 88. Virginia Water Resources Research Center, Virginia Polytechnic Institute and State University, Blacksburg, VA. 62 p.

Herbein, SL, GM Simmons,Jr. and SL Myers. 1996 Use of (PFGE) to investigate nonpoint fecal coliform sources to tidal inlets on the Eastern Shore of the Chesapeake Bay. Poster presented at the 1996 Annual Meeting of the American S ociety of Microbiology, New Orleans, LA. (Abstract with data available).

Hood, MA and GE Ness. 1982. Survival of *Vibrio cholerae* and *Escherichia coli* in estuarine waters and sediment. Appl. Environ. Microbiol. 43: 578 -584.

Levesque, B, P Broussequ, P Simard, E Dewailly, M Meisels, D Ramsay, and J Joly. 1993. Impact of the Ring-Billed gull (*Larus delawarensis*) on the microbiological quality of recreational water. Appl. Environ. Microbiol. 59:1228 -1230.

Maslow, JN, AM Slutsky, and RD Arbeit. 1993. Application of Pulsed-Field Gel Electrophoresis to Molecular Epidemiology, pp. 563 -572. *In* DH Persing, TF Smith, FC Tenover, and TJ White (eds.), *Diagnostic Molecular Biology Principles and Applications*. American Society of Microbiologists, Washington, DC.

Murphy, DD. 1988 Challenges to biological diversity in urban areas, pp. 71 -76. *In* EO Wilson and FM Peter (eds), *Biodiversity*. National Academy Press, Washington, DC.

Northern Virginia Planning District Commission. 1996(a). Staff Analysis of 199 0 U.S. Census data (unpublished). Annandale, Virginia 22003.

Northern Virginia Planning District Commission. 1996(b). Four Mile Run watershed in -stream water quality final report (April 1992 - March 1993, expanded October 1996).

Northern Virginia Planning District Commission. 1998. Bacteria source identification —a phased approach for meeting CWA goals (604 -b proposal to Virginia DEQ).

Randall, CW, TJ Gizzard, RC Hoehn. 1978. Impact of urban runoff on water quality in the Occoquan watershed. Bull. 80. Virginia Water Resources Research Center, Virginia Polytechnic Institute and State University, Blacksburg, VA. 77p.

Reay, WG. 2000. Influence of septic tank effluent on groundwater quality and nearshore sediment nutrient and fecal coliform bacteri a fluxes. Submitted to *Estuaries*.

Riley, SPD, J Hadidian, and DA Manski. 1998. Population density, survival, and rabies in raccoons in an urban national park. Can. J. Zool. 76: 1153 -1164.

Simmons, GM, Jr. 1994. Potential sources for nonpoint introd uction of *Escherichia coli* (*E. coli* ) to tidal inlets. Interstate Seafood Conference, Proceedings. Rehobeth Beach, Delaware.

Simmons, GM, Jr. and SA Herbein. 1995. The importance of marshes as potential reservoirs for fecal coliforms in coastal marin e environments. Final Report to the Virginia Department of Conservation and Historic Resources. Division of Soil and Water Conservation, Richmond, VA.

Simmons, GM, Jr., SA Herbein, and CA James. 1995. Managing nonpoint fecal coliform sources to tidal inlets. Water Res. Update. Issue 100: 64 -74.

Stephenson, GR and RC Rychert. 1982. Bottom sediment: a reservoir of *Escherichia coli* in rangeland streams. Jour. Range Management 35: 119 -124.

Sherer, BM, JR Miner, JA Moore, and JC Buckhouse. 1992. I ndicator bacterial survival in stream sediments. J. Environ. Qual. 21: 591 -595.

State Water Control Board. 1997. Water Quality Standards. Effective date, December 10, 1997.

Tynkkynen, S, R Satokari, M Saarela, T Matila -Sandholm, and M Saxelin. 1999. Comparison of ribotyping, randomly amplified polymorphic DNA analysis, and pulsed -field gel electrophoresis in typing of *Lactobacillus rhamnosus* and *L. casei* strains. Appl. Environ. Microbiol. 65: 3908 - 3914.

Van Donsel, DJ and EE Geldreich. 1971. Re lationships of *Salmonella* to fecal coliforms in bottom sediments. Water Research 5: 1079 -1087.

Virginia Department of Environmental Quality (DEQ) and Virginia Department of Conservation and Recreation (DCR). 1998. 303(d) Total Maximum Daily Load priori ty list report.

## Saturated Overland Flow in a Regularly Flooded Salt Marsh

Steven Turaski, Patricia Wiberg
Department of Environmental Sciences
University of Virginia
E-mail: sjt3b@virginia.edu

### ABSTRACT

Saturated overland flow occurs when the rainfall exceeds the storage capacity of the soil. In portions of a tidal marsh that are regularly flooded, the water table is close to the surface and the saturation deficit is small. Spatial and temporal patterns in runoff are determined by the formation of the saturation deficit and the timing of precipitation with respect to tidal flooding. The physical and biological controls on the saturation deficit have been investigated using a simple numerical model. The model results correlate well to field measurements of water table position obtained in a mainland salt marsh on the Eastern Shore of Virginia.

**Keywords:** erosion, sediment, salt marsh, unsaturated zone

### INTRODUCTION

Net sediment accretion or erosion is a determining factor in how a marsh will respond to sea level rise. Brinson and others (1995) describe state changes in marsh ecosystems based on topography and sedimentation dynamics. The occurrence of marsh surface erosion is suggested by earlier work that shows that short-term deposition rates are greater than long-term accumulation rates in Phillips Creek Marsh on the Eastern Shore of Virginia (Christiansen, 1998). Tidal flows cannot erode sediment. Vegetation dampens tidal flow velocity, promoting deposition and preventing resuspension of settled particles because the shear stress generated by tidal waters is insufficient to entrain sediment (Christiansen et al, 2000). Therefore, erosion during episodes of surface runoff is the most likely means of erosion. Sediment particles would be removed by overland flow upon disturbance by rain splatter, aided by the break-up of deposited sediment flocs by the freshwater.

There has been some dispute in the literature regarding the occurrence of erosion during runoff. Oertel (1976) noted an increase in suspended sediment in the estuarine and near-shore environment associated with rain when the marsh was exposed, but it was not determined if the sediment source was the marsh or the upland. Ward (1981) noted a similar increase in suspended sediment in a tidal creek from rain during tidal exposure. However, Letzsch and Frey (1981) observed runoff that did not carry sediment during heavy rains on an exposed salt marsh. Our work to date indicates that, at a minimum, sediment is mobilized on the surface during runoff events.

If indeed sediment is being eroded from the marsh surface during runoff events, there are several questions to investigate. What conditions are conducive to surface runoff? Where is runoff expected? How frequently does runoff occur? How does erosion occur; what are the physical mechanisms? What is the rate of erosion, both for an event and long-term?

These questions can be divided into two categories. The first includes characterizing the conditions for runoff and describing the erosion process. The second involves characterizing rates of runoff and erosion. The current work addresses the first set of questions. We have determined a mechanistic-conceptual model to describe the process of sediment erosion. The temporal and spatial patterns of runoff are being investigated, both on an event-scale and over a 10-year data set. Our work assumes both runoff and erosion occur whenever the conditions are favorable and does not measure erosion rates. Therefore predictions cannot be compared to actual measurements of erosion or runoff. Instead, the modeled saturation deficit has been compared to field measurements of water table position to verify model accuracy. Erosion rates are obviously very important in determining the significance of the process; however, that work would form a separate investigation.

Hillslope hydrologists have developed a conceptual model for surface runoff as saturated overland flow. The soil saturates as the water table rises or a perched water table forms during periods of precipitation or other water input. When the soil is saturated, no more water can infiltrate and any additional water added to the surface runs off. This model gives rise to the concept of a variable contributing area within a catchment. That is, not all of the catchment contributes surface runoff to the stormflow hydrograph. Rather, the source area changes based on the extent of soil saturation. (See Tischendorf, 1969; Dunne and Black, 1970; Hewlett and Nutter, 1970; Chorley, 1978).

An area of salt marsh can be viewed as analogous to a hillslope, being defined by the tidal creek at lower elevations and the forested upland at upper elevations. On a terrestrial hillslope, precipitation is the dominant input and lateral subsurface flow (drainage) is significant. Soil saturation is controlled by drainage, evapotranspiration, and precipitation. On regularly flooded areas within a salt marsh, soil saturation is controlled by tidal flooding, evapotranspiration, and precipitation. Tidal input is significant and the water table in these areas is near the surface. Salt marshes typically have a very small topographic gradient. Near tidal creeks lateral flow can be significant, but in the interior of the marsh evapotranspiration dominates pore water movement and subsurface flow is negligible (Nuttle, 1988).

To apply the saturated overland flow concept to a salt marsh requires consideration of the timing of precipitation with respect to tidal flooding. Tidal waters absorb rain falling onto a flooded marsh; therefore, rainfall must coincide with tidal exposure for runoff to occur. An unsaturated zone develops during exposure of the marsh surface after the tide recedes, when pore water is drawn out by evapotranspiration or drains laterally in the subsurface near creekbanks (Harvey et al, 1987). The position of the water table is of critical significance to surface runoff. The depth to the water table and soil moisture content above the water table determine the saturation deficit, the amount of water necessary to saturate the soil. The moisture content is typically at or near saturation for the fine-grained salt marsh sediments above the water table (Harvey et al, 1987; Hughes et al, 1998). Given the shallow depth to the water table in areas that receive frequent tidal inundation and a high water content, the saturation deficit is typically small. Saturated overland flow is expected when rainfall exceeds the saturation deficit.

Surface elevation and evapotranspiration are the major determinants of water table position in the marsh interior. Elevation is significant as a control on tidal inundation. Water loss from the marsh interior is almost entirely due to evapotranspiration. Nuttle (1988) showed that evapotranspiration accounted for 3.2 mm $day^{-1}$ of water and drainage for only 0.1 mm $day^{-1}$ when soil water was replaced daily by tidal inundation. During neap tides, when soil water was not replaced daily, drainage was still insignificant in the interior, accounting for 5 mm over 15 days. Sediment texture and precipitation also affect the water table position.

To summarize, the conditions favorable for overland flow are rainfall in excess of the saturation deficit at times when the tide is off the marsh. The soil will absorb rainfall until saturated, at which point saturation excess will begin to collect in the lowest parts of the microtopography. Infiltration excess overland flow could also occur infrequently. The infiltration rate is sufficient to absorb "typical" high intensity events, but the most intense rainfall rates exceed the infiltration rate. Intense rain events associated with a storm surge that inundates the marsh would not be effective in eroding the surface.

The microtopography of Phillips Creek Marsh is composed of sediment mounds and a network of trough-like depressions among the mounds. The crabs obviously create and maintain the microtopography, based on field observations. The mounds are on the order of 25-500 $cm^2$ and are created by crabs as they remove material while burrowing. The mounds are composed of loosened, easily erodable sediment. They are elevated above the saturation excess collecting in the depressions and are therefore exposed to the impact of raindrops. Sediment has been observed to be disturbed from the mounds during rain events and shed into adjacent depressions. Water in the depressions collects until a threshold depth is exceeded, at which point it begins to flow in rills down the slight topographic gradient. Less dense organic matter is more likely to be transported in the rills than mineral matter. Plants, while so significant as a control on the saturation deficit, do not directly affect the microtopography. Stem density is very high and the plants are randomly distributed, being found both associated with the mounds and in the depressions.

In this paper we address the spatial and temporal controls on runoff by saturated overland flow on a tidal salt marsh. That is, where do we expect runoff and under what conditions? To accomplish this we first characterize water table dynamics in the marsh interior on an hourly time-scale using field measurements of the water table along a topographic transect from the marsh to the upland over a period of about six months. We then identify and characterize the physical and biological controls on the water table. Physical controls (precipitation, tidal inundation, soil hydraulic properties) and biological controls (evapotranspiration, rooting depth) are investigated using a numerical model. The model results are compared to field measurements of water table elevation at positions along the transect. The data presented and discussed here are part of a larger field campaign initiated in July 1999 and scheduled for completion in spring 2001.

In the future we will model the occurrence of saturated overland flow across a "naturally delineated" plot (bound by the upland to the east, a headward erosion channel to the north, and to the south and west by low elevation areas adjacent to the tidal creek channel). Spatial patterns in runoff will vary based on differences in the length of time the marsh surface at each elevation is exposed, which corresponds to the formation of a saturation deficit. This is a function of the frequency and duration of tidal flooding at each elevation. Temporal variations in modeled runoff will be analyzed to determine what types of precipitation events are most effective at producing runoff . For example, coastal storms that generate precipitation but that also inundate the marsh with storm surges will not induce runoff; whereas convective thunderstorms are likely to be independent of tidal elevation.

## STUDY SITE

Field measurements were conducted as part of the Long-Term Ecological Research (LTER) effort at Phillips Creek Marsh in the Virginia Coast Reserve on the Eastern Shore of Virginia. This is a mainland marsh in the barrier-lagoon complex on the seaside of the Delmarva Peninsula (figure 1). The land is owned and maintained by the Nature Conservancy.

A 50m transect with 10 wells was installed in July 1999 to measure the position of the water table. The study site includes low marsh, high marsh, and a transition into the forested upland (Hmieleski, 1994). The low marsh is defined by the presence of intermediate height (~1.0 m) *Spartina alterniflora*. The high marsh is delineated by short form *S. alterniflora* grading into a mix of *S. patens* and *Distichlis spicata* at higher elevation. The transition into the forested upland in associated with shrubs (*Iva fructescens*, *Myrica cerifera, Baccharis halimifolia* ), grasses from the high marsh and *Panicum virgatum* , and standing dead trees. In terms of inundation frequency, the study site is in "low marsh" up to the upland transition. In the dichotomous classification of earlier studies in which an area was designated either "creekside" or "interior" (e.g., Nuttle, 1988; Blum, 1993), the study site is in the marsh interior.

The area (~1 ha) surrounding the transect of wells was surveyed using laser theodelite equipment and related to mean sea level. About 30% of maximum tidal elevations at high tide exceed 80 cm, which is approximately the maximum elevation of the high marsh. The tidal range at Redbank (~1.6 km from the site) is 2.25 m (Christiansen, 1998).

Mean temperature and total precipitation records from 1961 to 1990 and 1999 were used to characterize the climate of the study site (Owenby and Ezell, 1992). These data were collected at Painter, VA, which is approximately 15 km from the site, on the center of the Delmarva Peninsula. Monthly precipitation totals for 1999 were consistent with the 30-year average. Unusually high rainfall in September 1999 can be attributed to Hurricanes Dennis and Floyd. The annual precipitation total for 1999 was higher than average, but within the range of values for the 30-year period. Monthly average temperatures were also consistent with the 30-year average.

## METHODS

Wells were constructed of 5-cm (2-in.) diameter PVC pipe. The lower portion of each well was factory-slotted (~0.1 cm diameter). The upper portion was hand-drilled with a 0.6-cm (0.25-in.) bit along the entire length, including the above-ground portion. Wells were wrapped in nylon screening to keep sediment from clogging the holes. Wells were 210 cm (7 ft.) in length and were installed to an approximate depth of 150 cm.

Water table elevations were recorded along the entire transect approximately daily during low tide for 5 weeks in July and August 1999. Whole-transect hourly measurements throughout low tide were obtained in August 1999 and August and September 2000. These measurements were made by hand using either a salinity meter or a sonic well level indicator. "Solinst M5 leveloggers" were used to record continuous water table elevations in various wells from July 1999 until the present (15-60 minute interval, max. 3 wells at a time). Hand measurements were compared to logger measurements to verify agreement. Using the method of Harvey (1986), well tests were performed to ensure measured values were representative of the actual water table position in the marsh. The well at 23m from the creek did not pass and has been excluded from the analysis.

An hourly record of solar radiation, air temperature, and precipitation was obtained by the LTER meteorological station in the marsh (~100m from the transect). Average, minimum, and maximum temperature ($^{o}$C) are measured with a "Campbell Scientific Temperature/ Relative Humidity probe HMP35C". Hourly averages are derived from 60 readings at one-minute intervals. Solar radiation (kJ $m^{-2}$ $hr^{-1}$) is measured with a "Li-Cor 200S total radiation sensor". The hourly value is the sum of measurements taken at one-second intervals. Precipitation (mm $hr^{-1}$) is measured in a "Leopold tipping bucket". Relative humidity, wind speed and direction, and photosynthetically active radiation are also measured. All data are stored with a "Campbell Scientific 21x datalogger". The Phillips Creek meteorological dataset extends from 1991 to the present, though there are significant gaps. Preceding data from 1989-91 are available for a Brownsville meteorological station (Porter et al, 1999a).

Tides were recorded every 12 minutes at the Redbank tide gage operated by the LTER (~1.6 km from the site). The tide gauge is a "Stevens pressure transducer" and the data extend from 1992 to the present, also with gaps (Porter et al, 1999b). The tidal record from the NOAA station at Wachapreague (~20 km from the site) extends from 1978 to the present.

The microtopography was measured by leveling a 1-$m^2$ PVC frame above the surface and recording the depth to the surface from this datum at 5-cm intervals. Plant stems were cut to facilitate the measurements and to observe the surface. The locations of crab burrows and sediment mounds were recorded.

To model water table elevation, well records were used to determine periods of tidal flooding. Only inundation was determined with the well record; actual well levels were not used in the model. For periods when the marsh was flooded, the model sets the water table elevation equal to the surface elevation. Starting at the surface immediately following inundation, the water table is drawn down by evapotranspiration and is modeled as the saturation deficit divided by the specific yield of the soil. The saturation deficit is the sum of evapotranspirative losses between inundation events, minus precipitation inputs. A generic value for the specific yield of salt marsh soils is 0.1 (Nuttle and Hemond, 1988; Nuttle and Portnoy, 1992). Specific yield is a gravity-driven soil drainage parameter and is useful here for modeling water retention above the water table, even though the unsaturated zone is controlled by roots and not drainage.

The model employed the Priestly and Taylor (1972) method of estimating potential evapotranspiration. This method produces a rate (cm $hr^{-1}$) and requires solar radiation and temperature data. The marsh is a well-watered surface following inundation; therefore, an equation that models potential evapotranspiration, which assumes evapotranspiration is not limited by water availability, is justifiable in a regularly flooded salt marsh (Brutsaert, 1982).

## RESULTS

The temporal controls on runoff involve the timing of rain and the spatial controls involve the formation of the saturation deficit. Water table measurements from August 1999 demonstrate the combination of these factors to produce conditions conducive to saturated overland flow. On a day when there was rainfall during marsh surface exposure the water table remained at the surface (figure 2). The water table in the marsh was typically within 5 cm of the surface during the summer of 1999 (figure 3). During neap tides, when the marsh is not inundated for several days, the water table was drawn down to about 10-15 cm below the surface, beyond which it was not drawn down further. It appears that the water table reached a lower limit at these times. This was seen repeatedly at various wells and a similar limit of 10-15 cm was reached. Neap tides occur approximately monthly, but the water table does not always reach the lower limit during neap-tide drying periods.

The model does well at capturing the timing and magnitude of drawdown events (figure 4). The significant departure on days 262-264 is due to model over-prediction of the drawdown during a neap tide. The water table reaches its apparent limit, but the model continues to show drawdown. There are modest departures on days 264-270. The marsh on these days was exposed at night, and without solar radiation no drawdown was modeled. However, the water table measurements show some nighttime drawdown.

In the process of measuring the microtopography, it became apparent by observation that crab burrowing dominates the microtopography. Observed locations of burrows and sediment mounds correspond directly to measurements of the microtopography (figure 5). Burrow density decreased with distance from the tidal creek. In the low marsh, average burrow density was 40 $m^{-2}$; in the high marsh, 3 $m^{-2}$.

Runoff was observed moving sediment in April and July 2000. Saturation excess collected in micro-depressions and ran off in rills. Dye placed on a sediment mound was carried into the adjacent depression; dye placed in a depression was carried away in a rill. There was more runoff where there is more of a topographic gradient (i.e., 27 m from the tidal creek). Individual raindrops disturbed several grains that were then carried in the rill. A more intense rain with larger drops would cause greater disturbance of the sediment. Also, a greater volume of rain would increase the depth and amount of runoff. Turbulent runoff could keep entrained sediment in suspension and remove it further. Less dense organic matter was observed to be transported further than mineral matter.

## DISCUSSION

Model over-prediction of drawdown during neap tides indicates that, when the water table is drawn down to a lower limit, subsequent water loss by evapotranspiration is withdrawn from the soil above the water table, while the water table is not drawn down further. Given the low conductivity of fine-grained salt marsh soils, it is unlikely water below the root zone is pulled up rapidly enough to replace water drawn out by roots and draw down the water table. Several attempts to quantify this by measuring soil moisture content above the water table during these conditions were unsuccessful because the water table did not reach its lower limit during the sampling periods. The lower water table limit at 10-15 cm is consistent with the root zone of the marsh grasses (Blum, 1993; Tirrell, 1995). Root density, which is greatest 0-10 cm below the surface, determines this limit, and not maximum root extent, which is 25-35 cm below the surface.

The current study is concerned with the saturation deficit, not the actual position of the water table, per se. The measurements of the water table were used to test the model and illustrate the process of erosion. The test shows the model to be adequate except during the drying periods when the water table reaches the lower limit of the root zone. The saturation deficit during these periods is only 2 to 4 cm, an amount that could easily occur in a single rain event. It is reasonable to assume the saturation deficit is fairly accurate, even when the water table drawdown is over-predicted, because the moisture content above the water table is high and evapotranspiration would not be limited by water availability. Therefore the potential evapotranspiration may still be used to approximate the saturation deficit.

The model assumes soil specific yield, vegetation, radiation, air temperature, and rainfall are uniform on the plot. The effects of soil properties are combined into a single term, the specific yield. Using a single value for the specific yield, the model results are adequate for both summer and winter and at several positions on the transect. Within marsh zones, plant species composition is fairly homogenous, minimizing differences in root water uptake by different species. Any spatial differences in the saturation deficit due to the distribution of plants would be slight because of the high stem density. These differences would be negligible when considering widespread tidal or precipitation inputs. The study area is small enough (0.9 ha) that it is reasonable to assume uniform meteorological inputs.

The nocturnal rate of water uptake by *S. alterniflora* roots (Dacey and Howes, 1984) can be incorporated into the model. This rate is similar to measurements of nighttime drawdown at the study site. At locations 27 and 43 m from the creek the daily water table for summer 1999 was approximately 15 cm below the surface. The increased depth to the water table at these locations was not repeated in 2000 and therefore does not appear to be a permanent phenomenon. In 2000, the depth to the water table was slightly greater at 39 m from the tidal creek than at adjacent wells. These differences were not significant when the modeled drawdown was compared to measurements of the water table position.

### Further Work for this Research

Tidal inundation will be the basis for modeling saturated overland flow across the study plot. It is assumed any surface elevation less than the elevation of tidal flooding will be inundated. Using 5-cm elevation contours and assigning inundation for the contour only when the tide reaches the upper contour limit will minimize the effect of microtopographic differences. The saturation deficit will be modeled for each 5-cm contour. Precipitation inputs will be considered during tidal exposure. Precipitation in excess of the saturation deficit will be modeled as overland flow and the area of that contour as part of the contributing area.

A time series of the saturation deficit and the frequency of saturation excess overland flow can be calculated using historical meteorological and tidal data. The model may then be applied to predict changes in runoff associated with increased atmospheric temperature and sea level rise. For example, increased temperature is expected to increase evapotranspiration and the saturation deficit, making runoff less frequent. Increased tidal elevation is expected to increase inundation frequency and duration, decreasing tidal exposure at a given elevation and creating fewer opportunities for runoff.

Our model will not be applicable to areas with lateral discharge and so cannot be broadly applied to whole marshes. Also, the distance from a tidal creek can be as important as surface elevation as a control on the frequency of tidal inundation (Hmieleski, 1994). It should be kept in mind the objective of this study is to assess saturated overland flow and erosion. The state changes in marsh ecosystems is a complex issue and the effects of gradual changes in sea level and temperature cannot simply be inferred from water table response (Brinson et al, 1995).

## CONCLUSIONS

Runoff is expected when rainfall exceeds the saturation deficit of the soil. The saturation deficit is adequately modeled as a simple water balance with tidal inundation and precipitation as the inputs and evapotranspiration as the only output. During runoff events, sediment is shed from sediment mounds into saturation excess in micro-depressions and is transported when the depth in the depressions exceeds a threshold and begins to flow in rills. Short-term field measurements verified the numerical and conceptual models, which can now be applied to a long-term data set.

## ACKNOWLEDGEMENTS

Support for this project was provided by the Virginia Coast Reserve LTER project, which was funded by National Science Foundation grants BSR-8702333-06, DEB-9211772, and DEB-9411974. The Nature Conservancy provided access to the site.

## REFERENCES


Blum, L. K. 1993. *Spartina alterniflora* root dynamics in a Virginia marsh. Marine Ecology Progress Series 102: 169-178.

Brinson, M. M., R. R. Christian, and L.K. Blum. 1995. Multiple states in sea-level induced transition from terrestrial forest to estuary. Estuaries 18: 648-659.

Brutsaert, W. H. 1982. Evaporation into the Atmosphere: Theory, History, and Applications. D. Reidel Pub. Co., Boston, MA.

Chorley, R. J. 1978. The hillslope hydrological cycle. Chapter 1 in *Hillslope Hydrology* (Kirkby, M. J., ed.) John Wiley and Sons. 393 pp.

Christiansen, T. 1998. Sediment Deposition on a tidal salt marsh. Ph.D. dissertation. Univ. of Virginia, Charlottesville, VA, 134 pp.

Christiansen, T., P. L. Wiberg, and T. G. Milligan. 2000. Flow and sediment transport on a tidal salt marsh surface. Estuarine, Coastal and Shelf Science 50: 315-331.

Dacey, J. W. H., and B. L. Howes. 1984. Water uptake by roots controls water table movement and sediment oxidation in short *Spartina* marsh. Science 218: 487-489.

Dunne, T., and Black, R. D. 1970. Partial area contributions to storm runoff in a small New England watershed. Water Resources Research 6: 1296-1311.

Harvey, J. W. 1986. Subsurface hydrology in a regularly flooded *Spartina alterniflora* marsh. MS Thesis. Univ. of Virginia, Charlottesville, VA, 62 pp.

Harvey, J. W., P. F. Germann, and W. E. Odum. 1987. Geomorphological control of subsurface hydrology in the creekbank zone of tidal marshes. Estuarine Coastal and Shelf Science 25: 677-691.

Hewlett, J. D., and W. L. Nutter. 1970. The varying source area of streamflow from upland basins. *Symposium on Interdisciplinary aspects of watershed management*, Montana State Univ., Bozeman, MT. American Society of Civil Engineers. pp 65-83.

Hmieleski, J. I. 1994. High marsh – forest transitions in a brackish marsh: the effects of slope. MS Thesis. East Carolina Univ., Greenville, NC. 129 pp.

Hughes, C. E., P. Binning, and G. R. Willgoose, G. R. 1998. Characterisation of the hydrology of an estuarine wetland. Journal of Hydrology 211: 34-49.

Letzsch, W. S., and R. W. Frey. 1980. Deposition and erosion in a holocene salt marsh, Sapelo Island, Georgia. Journal of Sediment Petrology 50: 529-542.

Nuttle, W. K. 1988. The extent of lateral water movement in the sediments of a New England salt marsh. Water Resources Research 24: 2077-2085.

Nuttle, W. K., and H. F. Hemond. 1988. Salt marsh hydrology: implications for biogeochemical fluxes to the atmosphere and estuaries. Global Biogeochemical Cycles 2: 91-114.

Nuttle, W. K., and J. W. Portnoy. 1992. The effect of rising sea-level on runoff and groundwater discharge to coastal ecosystems. Estuarine Coastal and Shelf Science: 34: 203-212.

Oertel, G. F. 1976. Characteristics of suspended sediments in estuaries and nearshore waters of Georgia. Southeastern Geology 18: 107-118.

Owenby, J. R., and D. S. Ezell. 1992. Monthly station normals of temperature, precipitation, and heating and cooling degree days, 1961-1990. Climatography of the U.S., Publication No. 81. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Climate Data Center, Asheville, NC.

Porter, J. H., D. O. Krovetz, J. Spitler, and W. Nuttle. 1999a. Redbank tide data, VCR- LTER data set.

Porter, J. H., D. O. Krovetz, J. Spitler, and W. Nuttle. 1999b. Meteorological data, VCR- LTER data set.

Priestly, C. H. B., and R. J. Taylor. 1972. On the assessment of surface heat flux and evaporation using large-scale parameters. Monthly Weather Review 100: 81-92.

Tirrell, R. L. 1995. Response of the sediment microbial community to *Spartina alterniflora* roots in a Virginia salt marsh. MS Thesis. Univ. of Virginia, Charlottesville, VA, 161 pp.

Tischendorf, W. G. 1969. Tracing stormflow to varying source area in a small forested watershed in the southeastern Piedmont. Ph.D. dissertation , Univ. of Georgia, Athens, GA. 114 pp.

Ward, L. G. 1981. Suspended-material transport in marsh tidal channels, Kiawah Island, South Carolina. Marine Geology 40: 139-154.

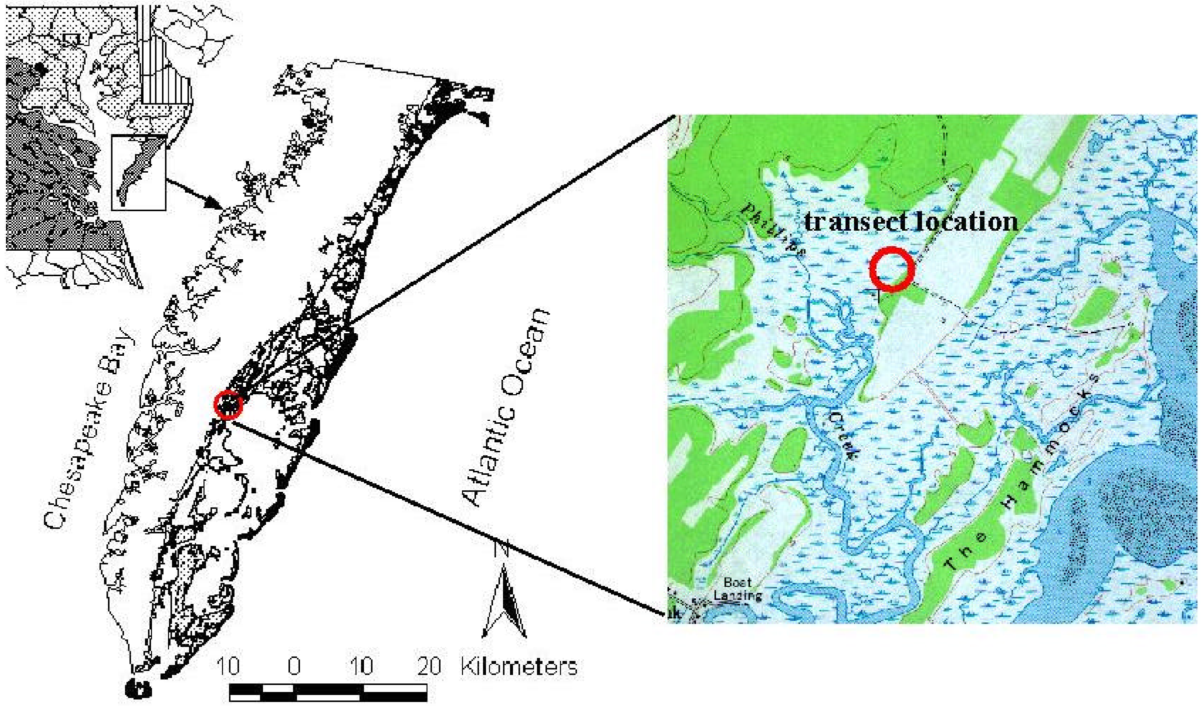



**Figure 1.** The study site is in a mainland-fringing marsh, forming the transition between the Delmarva Peninsula and the barrier island-lagoon complex. The transect of wells spans a topographic gradient from the low marsh to the forested upland. Figure created by Scott Dusterhoff.

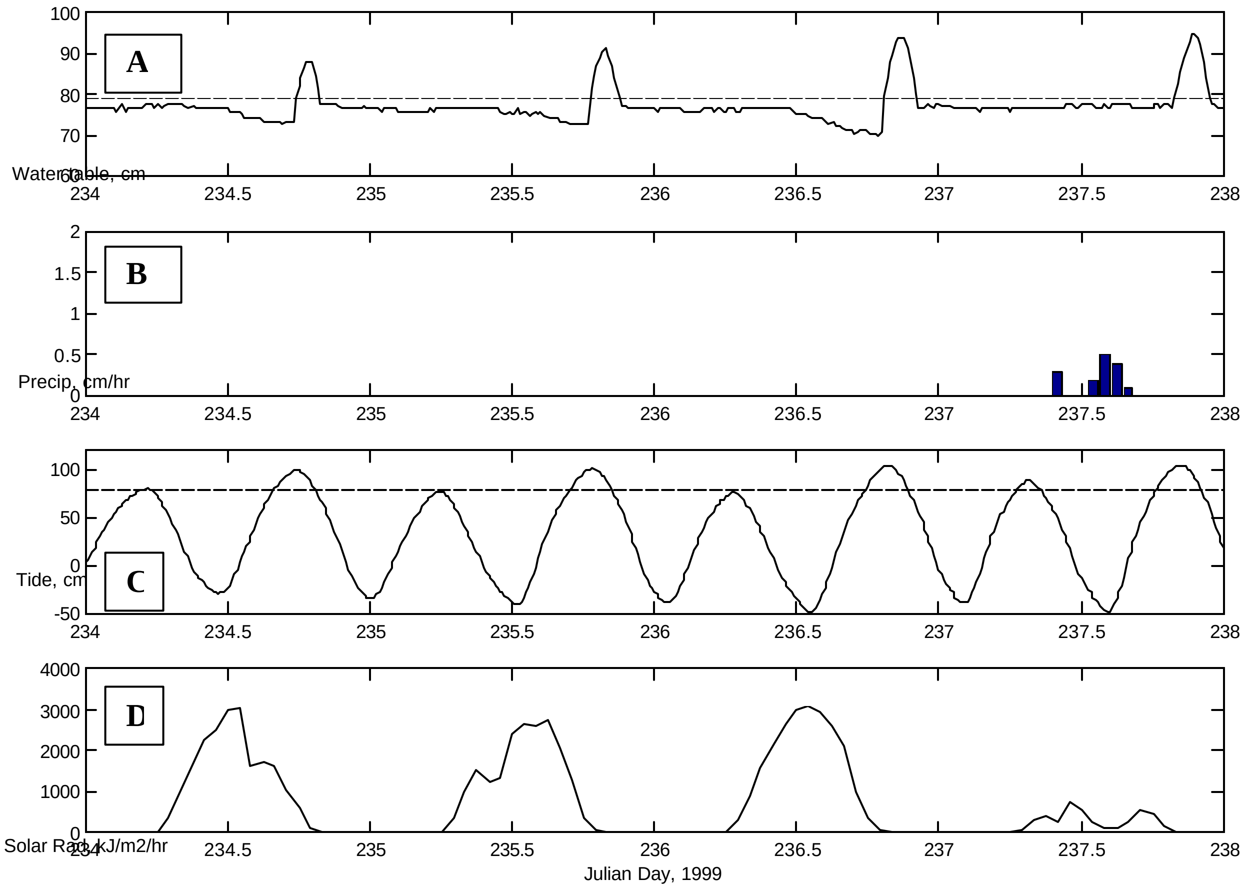



**Figure 2.** The dashed line in panels A and C represents the marsh surface elevation of 79 cm at 39m from the tidal creek channel. Water table drawdown occurs on days 234, 235, and 236, but not on 237, when precipitation coincided with low tide and solar radiation was minimal. The water table at this time remained very near the surface, illustrating conditions favorable for saturation overland flow.

A. Water table elevation 39m from the tidal creek channel (cm).
B. Hourly precipitation (cm $hr^{-1}$).
C. Tidal elevation (cm) using Redbank water levels converted to marsh water level according to Christiansen (1998). Note that the conversion is approximate because the tidal record shows that the tidal height exceeds the surface elevation twice on Day 237, but the well record shows only one period of inundation on that day.
D. Solar Radiation (kJ $m^{-2}$ $hr^{-1}$).

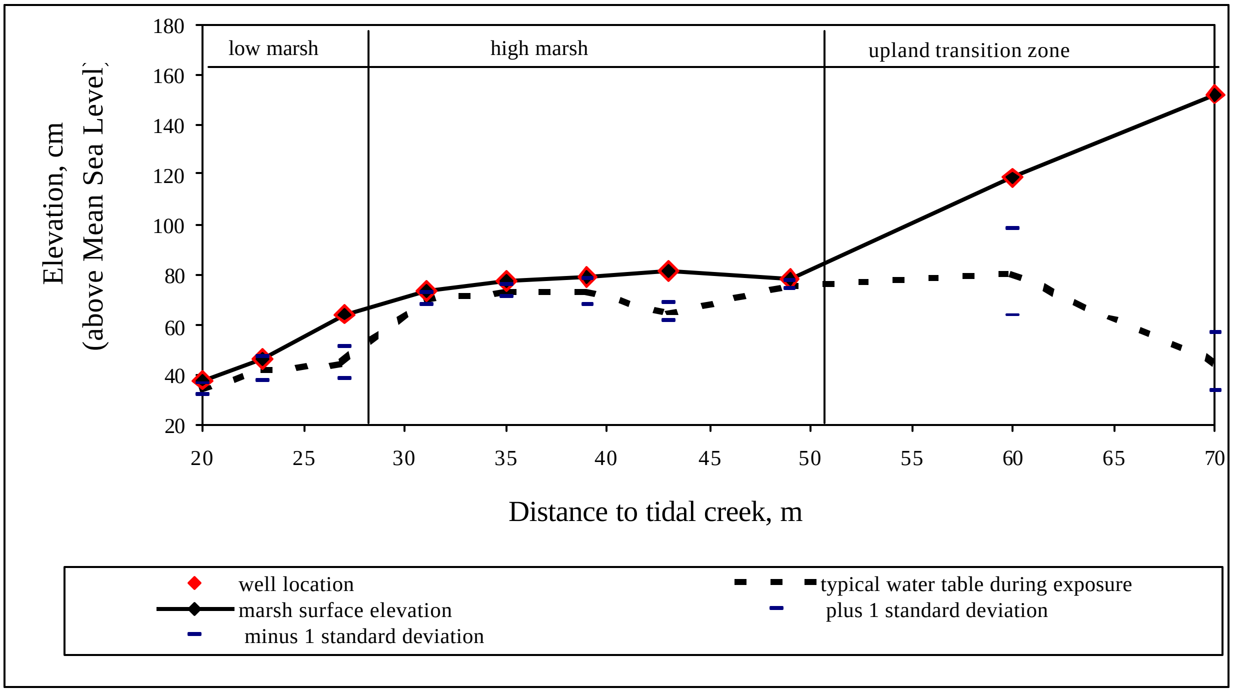



**Figure 3.** "Typical" water table elevations for the study site show the water table remains within 5-15 cm of the surface. Measurements were made during low tide in July and August 1999, but not necessarily at the lowest water table position. Therefore, the measurements are not an average or a low-tide minimum.

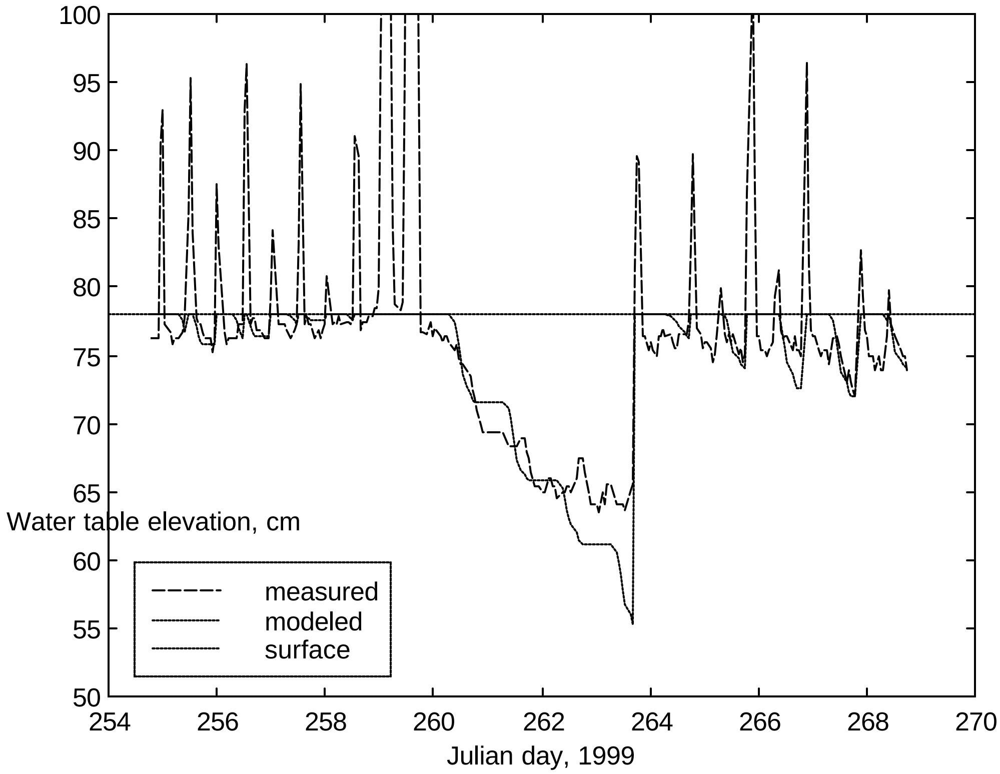



**Figure 4.** Measured water table elevations are from the well 35m from the tidal creek for a two-week period in September 1999. Note that the spikes in the measurements are periods of tidal inundation. When the marsh is inundated, the water table is at the surface. The model does well to capture the below ground water table dynamics and is not designed to simulate periods of inundation.

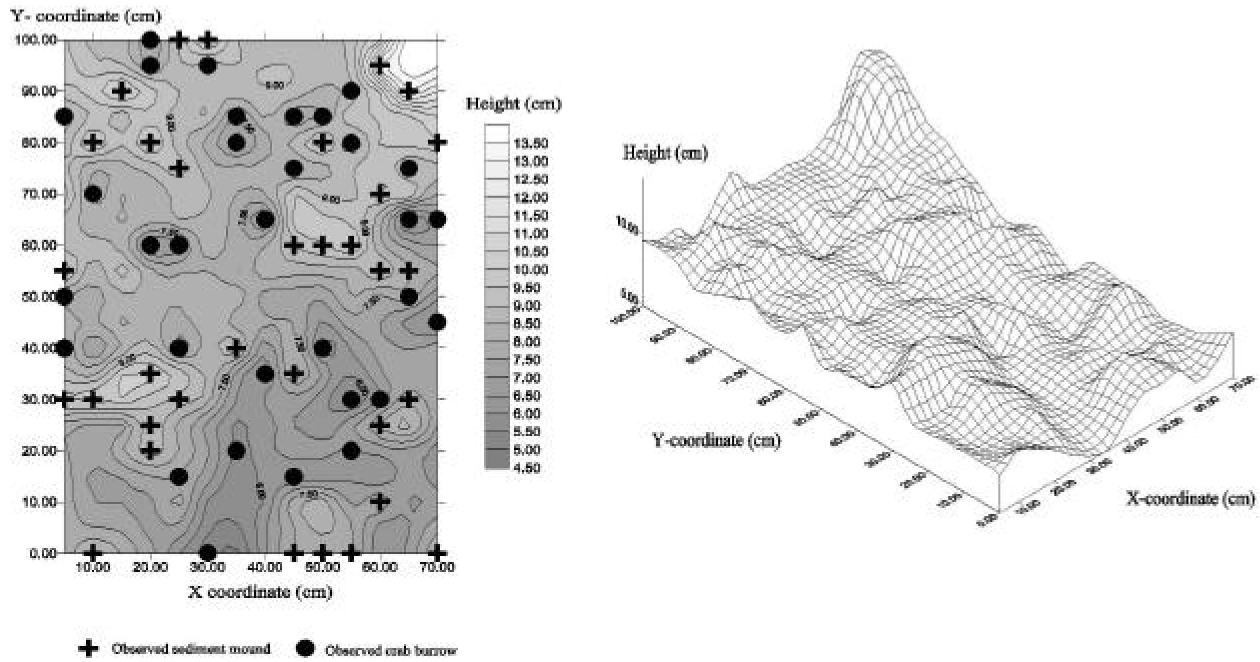



**Figure 5.** The microtopography corresponds directly to crab burrows and mounds of excavated sediment. The height was measured from an arbitrary datum.

# A New Tool for Tracing Human Sewage in Waterbodies: Optical Brightener Monitoring

Donald F. Waye, Senior Water Resources Planner
Northern Virginia Regional Commission
(formerly with the Northern Virginia Planning District Commission)

E-mail: dwaye@novaregion.org

## EXTENDED ABSTRACT

Municipal Separate Storm Sewer System (MS4) communities attempting to determine the extent and locations of any illicit discharges now have an effective and inexpensive option: optical brightener monitoring of storm sewer outfalls. Optical brighteners (OBs) are dyes added to most laundry detergents sold in the United States, including some labeled "dye-free" (look for "whiteners" or "brighteners" on the label's fine print).

Unlike more expensive tests like surfactants or caffeine, optical brightener monitoring (OBM) lends itself to wholesale surveys of storm drain outfalls within a given study area. Further, composite samples are taken, which offer a distinct advantage over the "hit-or-miss" nature of grab sampling. When the Northern Virginia Planning District Commission (NVPDC) used this technique on a 6,400 acre (10 square-mile) portion of the 12,600 acre (19.7 square-mile) ultra-urban Four Mile Run watershed in the summer of 1999, its total cost was less than $7,000, most of which was field costs. The survey took only six weeks to complete. In a 10 week period during the summer of 2000, the watershed was revisited, and every known storm drain outfall in the watershed was screened for optical brighteners—nearly 300 outfalls in all. Many outfalls were intentionally monitored twice, if investigators saw evidence that a problem may exist.

The novel approach hinges on the fact that laundry waste is a significant component of raw sewage. While these dyes are invisible to the naked eye, they appear as an easily detectable bright glow under an ordinary black light. The ploy is simple: monitor stream outfalls with OB "traps" (fabric sensitive to OB absorption), and if the trap fabric glows under black light, raw sewage may be leaking into the stream.

The 1999 investigation led to the discovery that two industrial-sized washing machines were discharging into a storm drain within Four Mile Run's Long Branch tributary in Fairfax County. Since the discharge consisted of intermittent laundry effluent only, the pollution was not obvious. The illicit connection was verified by a spectrofluorophotometer analysis. The investigation was turned over to Fairfax County, which confirmed the illicit connection with its own surfactants test. The County quickly traced the discharge to a hotel where the problem was promptly fixed.

Although OBM has promise for screening storm drain outfalls for sewage, it has limitations. It is not likely to detect sewage from most commercial buildings, which often lack laundry facilities. Also, centralized laundry rooms common to most multi-unit residential buildings add a complicating factor—for instance plumbing errors that do not involve laundry rooms may avoid being detected by OBM. Still, because of its relatively low cost and quick turn-around, applying OBM to storm drain outfalls should provide valuable information to MS4 communities.

During its second season of OBM in the Four Mile Run watershed, investigators from NVRC (formerly NVPDC) applied the lessons learned from the first season of monitoring. The pilot project benefited from two modifications:

1) A new type of OB trap was used. In the 1999 pilot, investigators used commercially available bulk cotton that had been cut into 10 cm (4 in) strips and held in place at each outfall by poultry netting and wire. Processed cotton (including sterile medical cotton) contains trace amounts of optical brighteners that interfere with low-to-moderate contamination levels of storm drain outfalls. While this speaks to the pervasiveness of this man-made dye as a good indicator of human activity, it limits the utility of commercial cotton to discovering only relatively high concentrations of OB in the field. Although unprocessed cotton picked straight from the field would be free of OB dyes, cottonseed oil is likely to interfere with the cotton's ability to absorb OBs. In this second season, NVRC employed fabric swatches use by the garment industry to test new dyes. The swatches are highly absorbent, free of OB dyes, and cost $1 each.

2) In the summer of 2000, each OB trap was left out in the field between 4 and 72 hours instead of the 7-10 day sampling period used in 1999. The objective of this modification was to maximize the composite nature of OB sampling while minimizing interference with wet weather events. Because of the heavy suspended sediment load inherent in urban stormwater, OBM has greatest utility during baseflow periods. Sediments (especially clay) carried by stormwater will coat the fibers in the trap fabric and mask any brighteners that may have been absorbed. Field personnel must pay close attention to weather forecasts and be ready to retrieve any traps when rains threaten. This modification resulted in a reduction of the percentage of traps lost in the field, despite the fact that there were considerably more storms during the second summer of monitoring.

Results from the second season of OBM revealed potential problems with 8 to 10 storm drain outfalls that may carry diluted sewage, although all levels detected during the second summer were well below the level found in the outfall that was revealed in 1999 to carry hotel laundry effluent. Follow-up monitoring is ongoing. Monitoring efforts during the summer of 2000 were severely hampered by an anomalous weather pattern of near-daily evening rain showers. Consequently, the OBM could have benefited from a longer composite sampling interval; a 24-hour minimum period is recommended, and 72 hours is optimal.

Additional information on NVRC's OBM project and related research is available online at <www.novaregion.org/4MileRun/obm.html>.


## ACKNOWLEDGEMENTS

This project was made possible by NVPDC/NVRC's dedicated interns: Kevin Johnson, Jessica Ross, and Marta Eckert-Mills. Financial assistance was provided by Virginia DEQ's mini-grants program for water quality monitoring, as well as the counties of Arlington and Fairfax and the cities of Alexandria and Falls Church through the regional Four Mile Run Watershed Management Program administered by NVRC.

**Influence of Topography on $pCO_2$ Dynamics in Forest Soils**

Daniel L. Welsch
George M. Hornberger

Department of Environmental Sciences, University of Virginia
PO Box 400123,Charlottesville, VA 22904-4123

E-mail: dwelsch@virginia.edu

## EXTENDED ABSTRACT

The spatial and temporal dynamics of the partial pressure of $CO_2$ ($pCO_2$) in soil air is poorly understood. Soil air $pCO_2$ controls the acid – base chemistry of soil water, having an significant impact on the chemistry of resulting stream water in small forested catchments. Research has given us an understanding of the important role $pCO_2$ plays in catchment acidification and base cation depletion (Reuss and Johnson 1985; David and Vance 1989; Pinol et al 1995). The value of $pCO_2$ in soil air is controlled primarily by microbial and root respiration. These functions are in turn controlled by substrate availability, soil moisture, and other minor factors. Recent work by Giardina and Ryan (2000) show that production of $CO_2$ from microbial decomposition of soil organic matter is not related to temperature. They determined that the rate of soil organic matter decomposition is a function of substrate availability. Substrate availability is related to hydrological flowpath which is related to topography. Soil moisture and substrate availability vary spatially and temporally in response to topographic and climatic gradients.

This proposed research investigates a new hypothesis. It is well understood that variations in soil $pCO_2$ influence stream chemistry. Water traveling via subsurface pathways to the stream encounters areas of different $pCO_2$, which is reflected in the chemistry of the water both in the soil and stream. The values of soil air $pCO_2$ have long been treated as static, even though evidence shows that variations in soil air $pCO_2$ can have significant effects on model predictions of stream chemistry (e.g. Neal and Whitehead 1988). However, little research has addressed this hypothesis**:** There is significant variation in the processes responsible for soil air $pCO_2$, resulting in large spatial differences in $pCO_2$ values. Topographic variation can be used a surrogate for process variation in explaining the spatial $pCO_2$ differences. If an understanding of how $pCO_2$ conditions vary in a catchment and how catchments respond to varying conditions is reached, the knowledge of spatial $pCO_2$ variations coupled with models of catchment hydrology (e.g. TOPMODEL) could yield new and powerful insights into the chemistry of waters within and leaving forested headwater catchments.

If the above hypothesis is correct, we will have demonstrated that: (1) not all areas of a watershed contribute alkalinity to the stream equally. (2) Relationships between flowpath and spatial variation of $pCO_2$ can influence stream chemistry. (3) Depletion of base cations may not be occurring at the same rate at all areas within a catchment (reflected in tree health?).

The study of $CO_2$ in soil has primarily been approached from an ecological or agronomic perspective, as an assessment of below-ground productivity (Ewel et al. 1987; Oberbauer et al. 1992). However, the hydrologic and biogeochemical impacts of soil air $pCO_2$ are large, driving carbonic acid weathering of silicate and carbonate minerals. It has been recognized that strong relationships exist between $pCO_2$ of soil solution and soil air and stream water chemistry (Reuss and Johnson 1985; Pinol et al. 1995). The dissolving of $CO_2$ into water results in the production of carbonic acid ($H_2CO_3$) which then dissociates to bicarbonate ($HCO_3^-$), and protons ($H^+$). Thus, the generation of alkalinity and the chemical weathering of

rocks is enhanced. Elevated $pCO_2$ values in soils are caused by the microbial decomposition of organic matter and root respiration, primarily controlled by soil moisture and substrate availability. At depleted soil air $pCO_2$ values, less $CO_2$ will dissolve in soil water, resulting in a higher pH and a solution less capable of chemical weathering.

Despite the importance of $pCO_2$ in controlling soil solution and stream chemistry, spatial and temporal variations of $pCO_2$ within a small catchment have only begun to be investigated. Neal and Whitehead (1988) showed that when working with models of watershed acidification (e.g. MAGIC, Cosby et al. 1996) variations in $pCO_2$ significantly altered model results. However, $pCO_2$ values are rarely measured and the distribution of this parameter is poorly understood. Results of this research could significantly alter our understanding of catchment acidification and base cation depletion through an enhanced knowledge of the processes controlling the spatial and temporal fluctuations of soil $pCO_2$.

Preliminary data were collected in the summer of 2000 from the Shenandoah National Park to identify the spatial trends in soil $pCO_2$. Soil temperature, soil moisture, soil pH, and soil $pCO_2$ values were measured in 18 locations throughout the park. Soil collected from these locations will be analyzed for total N, total C, cation exchange capacity, and exchangeable bases. These data will be analyzed to determine the parameters which most strongly influence $pCO_2$ values and will also serve to parameterize possible modeling efforts which will stem from this work.

The second phase of this research is to instrument and monitor a single catchment to determine the spatial and temporal variations in $pCO_2$ intensively. The goal will be to capture the spatial and temporal dynamics of $pCO_2$ throughout this watershed. This goal will be accomplished through the use of continuous and instantaneous measurements. Four parameters will be measured continuously: rainfall, soil temperature, soil moisture, and soil air $pCO_2$. Soil water will be sampled using tension and zero-tension lysimeters and stream water will be sampled using ISCO automated samplers. Water will be analyzed for pH, alkalinity, and major ions.

To augment the continuous measurements, a spatial network of sampling points will be established throughout the catchment in order to capture the topographic variability. Sampling locations will be identified through topographic analysis. At each sampling location, soil $pCO_2$ will be measured at two depths. Soil temperature and soil moisture will be measured as well. Sampling will be conducted to ensure a range of soil temperature and soil moisture conditions are captured. When this phase is complete, a record of spatial and temporal distributions of soil $pCO_2$, soil temperature, and soil moisture will have been obtained. It is hoped that correlations between the spatial but non-continuous data record can be made with the continuous temporal record through the use of spatial statistics to achieve a complete spatial data series. This will give a more complete understanding of the spatial and temporal variations of $pCO_2$ in soil air and permit the construction of more robust and accurate models of catchment water chemistry.

## REFERENCES


Cosby, B. J., G. M. Hornberger, J. N. Galloway, and R. F. Wright. 1986. Freshwater acidification from atmospheric deposition of sulfuric acid: a quantitative model, *Env. Sci. & Tech.*,19: 1144-1149.

David, M. B. and G. F. Vance. 1989. Generation of soil solution acid-neutralizing capacity by addition of dissolved inorganic carbon, *Env. Sci. and Tech.* 23(8), 1021-1024.

Ewel, K. C., W. P. Cropper, and H. L. Gholz. 1987. Soil $CO_2$ evolution in Florida slash pine plantations. I. Changes through time. *Can. J. For. Res.* 17, 325-329.

Giardina, C. P. and M. G. Ryan. 2000. Evidence that decomposition rates of organic carbon in mineral Soil do not vary with temperature. *Nature*, 404, 858-861.

Neal, C and P. G. Whitehead, 1988. The role of $CO_2$ in long term stream acidification processes: a modeling viewpoint. *Hydrol. Sci.* 33(1) 103-108.

Oberbauer, S. F., C. T. Gillespie, W. Cheng, R. Gebauer, A. Sala Serra, and J. D. Tenhunun. 1992. Environmental effects on $CO_2$ efflux from riparian tundra in the northern foothills of the Brooks Range, Alaska, USA, *Oecologia*, 92, 568-577.

Pinol, J., J. M. Alcaniz, and F. Roda. 1995. Carbon dioxide efflux and $pCO_2$ in soils of three *uercus ilex* montaine forests, *Biogeochem.* 30(3), 191-215.

Reuss, J. O. and D. W. Johnson, 1985. Effect of soil processes on the acidification of water by acid deposition, *J.Environ. Qual.* 14, 26-31.

# Water Quality Modelling and Pollution Control for the Zhang-Wei River Basin

Fawen Zheng
*Department of Civil and Environmental Engineering, South Dakota School of Mines and Technology, 501 E St Joseph Street, Rapid City, South Dakota 57701*
E-mail: fawen_zheng@yahoo.com

Jos van Gils
*Delft Hydraulics, P. O. Box 177, 2600 MH Delft, The Netherlands*

M. J. Hall
*International Institute for Infrastructural, Hydraulic, Environmental Engineering, P. O. Box 3015, 2601 DA Delft, The Netherlands*

## ABSTRACT

This study focused on basin-wide integrated management for pollution control in the Zhanghe and Weihe (ZW) river system located in northern China. The water quality modelling of organic matter and the pollution controls have been carried out for this river system. The cause-effect relations between pollution sources, river hydrology and water quality (DO and BOD) were simulated with the SOBEK model, a numerical water flow and water quality model. Water flow and water quality data for the years 1991 and 1988 were used for calibration and verification, respectively. The simulation results showed that the calibrated/verified model is acceptable for assessing the effectiveness of pollution control actions. Then the possible pollution control measures were developed. Projections of pollution source were made for 2000 and 2010. Four scenarios for pollution control were defined for improvement of water quality from upper boundary or/and point source pollution control. The effectiveness of these four scenarios on present and future situations was analyzed with the help of the calibrated/verified DO-BOD model. Based on the simulation results, it was demonstrated that the scenarios, which are the combination of point sources pollution control and improvement of water quality from upper boundary, largely improve the water quality to meet the pre-set water quality objectives.

**Keywords**: water quality, modelling, SOBEK model, pollution control, Zhang-Wei River Basin

## INTRODUCTION

Environmental problems related to water, air and soil have become manifest throughout the world in the last decades. In this respect, developing countries are of special concern (Baarse et al, 1990). In China, as the largest developing country, with rapid industrial and agricultural developments in the last 25 years, water resources in many river basins are polluted severely where essentially most of the waste material, both from industrial and domestic origin, is discharged into the surrounding water system without adequate treatment. The bulk of the waste production consists of organic wastes, but from a variety of primary small industries, a wide range of other, often highly toxic, substances are produced and discharged as well. The consequences are becoming quite obvious, especially in densely populated areas. Within city areas and downstream of cities, the surface waters may become anaerobic for much of the year. Micro-pollutants are often found in unacceptable concentrations in both surface waters and sediments, affecting human health and posing a serious threat to both the aquatic and terrestrial environment.

An investigation of the water pollution situation in China (Qian, 1994) has shown that organic pollution and ammonia-N have displayed on increasing trend. Industrial wastewater and city sewage are the main sources of water pollution in the China's river systems. At present, water pollution in these rivers has become a basin-wide issue. The poor water environment has become a barrier to national economic development. A lot of investment is prohibited due to the water quality problem.

Therefore, it is necessary and urgent to carry out integrated planning and management for those river basins. This study focused on these aspects of basin-wide integrated management for pollution control. The midstream river system of one river sub-basin (Zhang-Wei River and South Canal Basin, ZWS, a sub-basin of the Haihe River system) was chosen as the study area. The modelling of organic matter and the pollution controls has been carried out for this study river system.

## OBJECTIVES

The general objective of a water quality study to solve the above water pollution problems could be formulated as follows: to improve surface water quality up to a point where public health is ensured and a minimum set of functional requirements are fulfilled. More specific objectives and methodologies of this study can be summarized as following: (a) Quantitative analysis of the cause-effect relations between pollution sources, river hydrology and river water quality, aided by a numerical flow and water quality model (SOBEK); and (b) Development of pollution control measures. Different scenarios were defined, which were analyzed with the help of the numerical model developed under (a).

## METHODS

### 1. Study River System Description

Based on the data availability, a part of Haihe River was chosen as the study river system. The Haihe River is one of the seven largest rivers in China (Figure 1), and is located in northern China. Severity of water pollution in Haihe River Basin ranks second among the seven largest rivers in China (HRWCC, 1995).

The Zhanghe River, Weihe River and South Canal (ZWS) River Basin is located in the southern part of the Haihe River Basin (see Figure 1). For this study, mainly based on the data availability, the middle part of the ZWS Sub-river Basin was chosen as the study river system (see Figure 1). There are two main tributaries along main river reach, which are the Gongqi Canal (upstream) and the Zhanghe River (downstream). The main river reach starts at Qimen and ends at Linqing, the total length of which is 275.5 km. The first tributary, Gongqi Canal reach, starts at Liuzhuang, flowing into main river at Laoguanzhui. The length of Gongqi Canal reach is 44 km. The length of the second tributary, Zhanghe River reach, is 42 km, which starts at Caixiaozhuang and flows into the main river at Xuwanchang. Between Laoguanzhui and Xuwanchang, the Anyanghe Rriver flows into Weihe River. It was simply dealt with as a lateral discharge, not a branch (tributary), due to the data limitation in this river. The study river system mainly consists of the Zhanghe River and Weihe River, abbreviated to the ZW River system. Figure 2 shows the ZW River system. The ZW River system will be used throughout this paper. There are four cities located along the river: Huaxian city, Xunxian city, Daming city and Guantao city. The industrial wastewater and domestic sewage from these four cities are main pollution sources to the study river system.

Industry in ZWS is well developed. Point source pollution is the main problem in this area and non-point source pollution is neglected in this study. Since the wastewater and sewage are discharged into the river directly without treatment, the water in the river is seriously polluted. According to water quality assessment study in this river system, nearly all of the water there is seriously polluted and can not be used for irrigation and other benefits (Zheng, 1997).

## 2. Model Description and Validation

### *2.1 Model Description*

The SOBECK model used in this study was developed by Delft Hydraulics and the Institute for Inland Water Management and Wastewater Treatment of the Netherlands government. It is a one-dimensional open-channel dynamic numerical modelling system, equipped with a user shell which is capable of solving the equations that describe unsteady water flow, salt intrusion, sediment transport, morphology and water quality in two kinds of regions: river and estuary.

<u>Water flow model</u> The water flow in SOBEK is described by two equations (Delft Hydraulics & RIZA, 1996): the momentum equation and the continuity equation. The two equations are simultaneously solved numerically by a finite difference method based on the Preissmann four-point scheme. Within the branches of the model the double sweep method is applied to obtain the unknown water levels and discharges at the grid points as a function of the water levels and grid points in the nodes and boundaries.

<u>Water Quality model</u> SOBEK computes the concentration of substances in the model as a function of place and time by applying the advection-diffusion equation (Delft Hydraulics & RIZA, 1994):

$$\frac{\partial c}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial c}{\partial x} - uc\right) + \left(\frac{\partial c}{\partial t}\right)_{processes} + S \qquad (1)$$

in which: $c$ = Concentration of constituent [kg/m$^3$], $D$ = Dispersion coefficient [m$^2$/s], $S$ = Waste load(s) [kg/m/s], $\left(\frac{\partial c}{\partial t}\right)_{processes}$ = Water quality processes.

The water quality module of SOBEK works according to a finite volume approach. To define the computational elements (the volumes), the user indicates on the modelled network the bounds of each element (which are called "segments"). These segments are connected to each other at specified locations. Water and constituents are transported from one segment to the other. Each segment has a volume and a size, and the water in them is assumed be completely mixed.

Each segment has a water volume that changes in time during the simulation. The change of volume is determined by the water flow module. In addition to water, each segment contains a certain amount of each modelled substance. The concentration of the substance is computed by dividing the substance mass by the water mass. The water quality model identifies "active" and "inactive" substances. The active substances are transported with the water, whereas the inactive substances are not (for example, substances that are present in the bottom sediments).

### *2.2 Water Flow Modelling*

<u>River System Schematization</u> For the water flow modelling study, a detailed water flow model was constructed which contained 10 nodes, 9 branches, 188 grids, two major tributaries, four

pumping stations and five lateral discharges (see Figure 3). There were four and five boundary conditions for water inflow and outflow respectively.

Model Input Data and Calibrated Model Parameters Model input data includes: (1) Boundary conditions: The boundary condition upstream is the flow (node 1, node 9 and node 10) and downstream is the water level at the last node (node 8); (2) Lateral discharges: Lateral discharges include water entering and water leaving (e.g. for irrigation) along the river; (3) Cross section: 8 cross sections were applied to construct the model. They were: Qimen, Wuling, Yuanchunji, Longwangmiao, Nantao, Linqing, Liuzhuang and Caixiaozhuang; and (4) Slope and length: Every branch has a slope and a length, which were input into the model.

The roughness coefficient is the major parameter to be determined in water flow calibration and verification. The Chezy coefficient was used for the water flow model, and was considered as a function of flow discharge and branch.

Calibration and Verification Data for the years 1991 and 1988 were used for model calibration and verification respectively. In these two years, field data for three cross sections were available for each year. Both 1991 and 1988 were dry years whereas 1988 was relatively wetter than 1991. During some periods within one year, the river was partly or completely dry, mainly resulting from water extraction for irrigation; the longest period in some parts of river was more than one month. In this case a small discharge was set in order to make the model run continuously through one year.

Two statistical parameters were used to describe the quality of the simulation results. $F_1$ is the average of differences between measured and simulated values expressed as a percentage. $F_2$, which is the so-called coefficient of efficiency and is commonly used, is based upon the sum of the squares of the differences between observed and computed and has a very wide range, from 1.0 to $-\infty$ (Minns and Hall , 1996; Diskin and Simon, 1977).

$$F_1 = \frac{\frac{1}{m}\sum_1^m |O_i - S_i|}{\overline{O_i}} * 100 \qquad (2)$$

$$F_2 = 1 - \frac{\frac{1}{m}\sum (O_i - S_i)^2}{\frac{1}{m-1}\sum_1^m (O_i - \overline{O_i})^2} \qquad (3)$$

where $O_i$ are the observed values, $S_i$ are the simulated values, and $\overline{O_i}$ is the average of $O_i$, $i = 1, ..., m$.

The comparison was made between the simulated and measured data for the water level in three cross sections along the river system. The water levels simulated with the SOBEK model agree quite well with the observations (Figure 4 and Figure 5 show the simulated results at Nantao for calibration and verification respectively). The calculated $F_1$ average deviation and the efficiency coefficient $F_2$ are shown in Table 1 and Table 2.

### *2.3 Water Quality Modelling*

Water Quality Processes For water quality modelling, four processes were considered, which are: (1) BOD decay; (2) the reduction of BOD by sedimentation; (3) oxygen reaeration from the atmosphere; and (4) sediment oxygen demand, SOD.

(1) *BOD decay*

In the SOBEK model, BOD was expressed in carbon equivalents, which is often called BODC in the unit of $gC/m^3$. The decay flux of BOD is modelled as the sum of zero and first-order processes. If the water temperature drops below a critical value, only the zero-order flux remains. The first-order flux is corrected for water temperature and oxygen concentration. Below a critical oxygen concentration the oxygen function for BOD decay becomes equal to a user-defined level while for above optimal oxygen concentration these functions have a value of 1.0. Linear interpolation of the oxygen functions is valid for intermediate oxygen concentrations (Delft Hydraulics and RIZA, 1994). These processes were implemented by the following formulae:

If $T < T_c$ then $$\frac{dL}{dt} = -F_0 \quad (4)$$

otherwise $$\frac{dL}{dt} = -F_0 - (K_1)_{20}(q)^{(T-20)} Lf(C) \quad (5)$$

in which $L$ is remaining BOD at time $t$ (mg/l), $T_c$ is critical temperature (°C), $F_0$ is zero order mineralization flux (gC.m$^{-3}$.d$^{-1}$) and $f(C)$ is oxygen function for mineralization which was defined as the following:

$C = Max(C,0)$

if $C \geq C_{op}$, $f(C)$ =1, if $C < C_{cr}$, $f(C) = F_{cl}$; otherwise $f(C) = (1 - F_{cl})\frac{(C - C_{cr})}{(C_{op} - C_{cr})} + F_{cl}$

in which $C$ = oxygen concentration (g.m$^{-3}$); $C_{cr}$ = critical oxygen concentration (g.m$^{-3}$); $C_{op}$ = optimum oxygen concentration (g.m$^{-3}$); $F_{cl}$ = value of the oxygen function for oxygen levels below the critical oxygen concentration (-).

(2) *The Reduction of BOD by Sedimentation*

Part of the BOD can be removed by sedimentation without consumption of the dissolved oxygen, and that removal rate is directly proportional to the remaining BOD. In SOBEK, the sedimentation flux of a substance that contributes to the amount of dry matter in the water column was calculated by the sedimentation formulation that is based on a sedimentation velocity and a sedimentation probability. Sedimentation always results in an increase of sediment in the upper sediment layer. No more than the available amount of substance in the water column can settle in one model-time step (Delft Hydraulics and RIZA, 1994). The calculation formula is as follows:

$$\frac{dL}{dt} = -\frac{f_{sed}}{H} \quad (6)$$

$$f_{sed} = f_0 + V_{sed} L P_{sed} \quad (7)$$

in which $f_{sed}$ = sedimentation flux of substance per horizontal surface area(g.m$^{-2}$.d$^{-1}$), $f_0$ = zero order sedimentation rate of substance (g.m$^{-2}$.d$^{-1}$), $H$ = depth of a segment (m), $V_{sed}$ = sedimentation velocity of substance (m.d$^{-1}$), $P_{sed}$ = sedimentation probability substance based on Krone < 0,1 > (-). $f_{sed}$ was limited as following: $f_{sed} = Min(f_{sed}, L\frac{H}{\Delta})$. $P_{sed}$ was computed as the following: if $T_{bss}$ = -1.0, $P_{sed}$ = 1.0; otherwise $P_{sed} = Max(0, 1 - (\frac{T_{bss}}{T_{cbss}}))$

where $\Delta$ = Time step (d), $T_{bss}$ = Bottom shear stress (Pa), $T_{cbss}$ = Critical bottom shear stress for sedimentation substance (Pa), $Max(a,b)$ = Function takes the maximum of arguments a and b, $Min(a,b)$ = Function takes the minimum of arguments a and b.

(3) *Oxygen Reaeration*

Transfer of oxygen into the water is brought about by the "driving force" $C_s$-$C$, in which $C$ and $C_s$ are the oxygen concentration (mg/l) at time $t$, and oxygen saturation concentration in the water, respectively. The saturated oxygen concentration is calculated based on the water temperature and corrected for salinity. The reaeration rate constant can be supplied directly by the user or is calculated by the model. In the literature, there are an enormous number of empirical relations available. Several empirical relations representing the effect of depth and stream velocity on the reaeration constant are implemented as options (Delft Hydraulics et al., 1994). Oxygen reaeration processes are implemented as follows (Gils et al., 1993):

$$\frac{dC}{dt} = (K_2)_{20}(q)^{(T-20)}(C_s - C) \qquad (8)$$

where $T$ is water temperature (°C).

$$K_2 = \frac{K_L}{H} \qquad (9)$$

in which $K_2$ = Reaeration rate (day $^{-1}$); $K_L$ = Mass transport coefficient for reaeration (m.day$^{-1}$); $H$ = Water depth (m).

$$C_s = (14.652 - 0.41022 \times T + (0.089392 \times T)^2 - (0.042685 \times T)^3 \times (1 - (\frac{Cl}{10^5})) \qquad (10)$$

in which $Cl$ = chloride concentration (g.m$^{-3}$)

(4) *Sediment Oxygen Demand, SOD*

Some of the earliest work on the magnitude of the SOD was done by (Baity, 1938) and (Fair et al., 1941). A variety of other in situ measurements have been made for numerous water bodies in recent years (Thomann and Muller, 1987). Kelderman pointed out SOD will especially be important for shallow waters and slow-moving waters (Kelderman, 1994).

The demand of $gO_2/m^2 \cdot day$ is exerted from a bottom surface area. Thus, in SOBECK model the total flux is

$$S_B^{'} = \frac{S_B A_B}{V} = \frac{S_B}{H} \qquad (11)$$

where $S_B^{'}$ is the total flux of sediment oxygen demand [$M/L^3 \cdot T$], $S_B$ is the SOD [$M/L^2 \cdot T$], $A_B$ is the contribution bottom area[$L^2$], $V$ is the volume of the overlying water column[$L^3$], $H$ is the water depth[$L$] (Thomann and Muller, 1987).

<u>Definition of a Segment</u> In the water quality module of SOBEK, river systems are specified into a number of computational elements, which are also often called 'segments'. A segment is the numerical element for a water quality simulation. Usually, selection of different numerical elements for water flow and for water quality is desired. For that reason, the concept of segments is introduced. The concept of a segment is that of a tank of water with a certain depth and water surface area, and with a homogeneous water quality. A segment can be equal to one grid element, but can also stretch over more than one water flow branch. For the reason of computation time, the segments are often chosen to be as large as possible. For water quality modelling, 44 segments were defined in the ZW River system. They are shown in Figure 6. Comparing segments to the grids (Figure 3), they are different.

<u>Calibration and Verification</u> Similar to the water flow modelling, data for the years 1991 and 1988 were used for water quality model calibration and verification respectively. A time step of 6 hours for water quality modeling was chosen.

Mean value and root mean square (RMS) of the differences between measured value and simulated data were used to evaluate model accuracy.

$$F_3 = \frac{1}{m}\sum_1^m (O_i - S_i) \tag{12}$$

$$F_4 = \sqrt{\frac{1}{m}\sum_1^m (O_i - S_i)^2} \tag{13}$$

where $O_i$ are the observed values and $S_i$ are the simulated values, $i = 1,\ldots,m$

$F_3$ can give the information of model systematic error. If $F_3 < 0$, it means that the model overestimate the pollutant concentrations; if $F_3 > 0$ it means that the model underestimates the pollutant concentrations; if $F_3 = 0$ it shows there is no systematic error in the model. $F_4$ describes the deviation of the simulation values from the measured data. The larger $F_4$, the less accurate the model. The calculated $F_3$ and $F_4$ are shown in Table 3 to Table 6. Comparisons between simulated and measured data are plotted and some of them are given in: Figure 7a-7c show BOD calibration results at Caiwan, Guantao and Liqing; Figure 8a-8c show DO calibration results at Caiwan, Gauntao and Liqing; Figure 9a-9b show BOD verification results at Guantao and Linqing; Figure 10a-10b show DO verification results at Guantao and Linqing.

In the calibration year, the largest $F_3$ values are 2.04 for DO and 2.77 for BOD in Yinzhenchiao station. It showed the model slightly underestimated the pollutant concentrations in this station. The largest $F_4$ values are 3.07 for DO and 3.67 for BODC at the same station. In the verification year, the calculated $F_3$ and $F_4$ showed better results than in the calibration year except DO in Yanchunji station ($F_4 = 5.59$). Generally speaking, from the simulation results of calibration and verification, it is demonstrated that a good fit has been obtained between measured data and simulated values. The model had high accuracy and was thus acceptable for analysing a wide range of wastewater management alternatives to determine the water quality impacts associated with their implementation.

The model parameters finally used in the water quality modelling are summarised in Table 7. The initial parameter values were adopted from literature of the previous study in this river basin, such as BOD decay rate and oxygen reaeration rate from (Yu, 1995; Liu et al., 1995), and from other literature or default values proposed in the SOBEK model. Starting calibration and verification of the model with those initial model parameters, the final parameter values were obtained through trial-and-error method until the good fit between simulated values and observed data was achieved.

*2.4 Sensitivity Analysis of Model Parameters*

Through sensitivity analysis one can obtain an overview of the most sensitive components in the model (Drolc and Koncan, 1996). The objective of such an analysis is to gain an understanding of the model response to the major input variables and to provide a measure of the possible effect on simulated concentrations of a particular water quality constituent of uncertainty in the estimates of important input variables.

The sensitivity analysis was performed by varying one input parameter from the calibration value by a constant amount while holding all other input variables constant. In general, the variation selected was intended to represent the perceived uncertainty associated with that input parameter.

The sensitivity analysis was performed for the following model parameters or variables at each of the sampling stations on the ZW River system: (1) BOD decay rate ($K_1$); (2) mass transport coefficient for reaeration ($K_L$ ); (3) BOD sedimentation velocity (VBOD); (4) sediment oxygen demand (SOD); (5) BOD loads from upper boundary; and (6) BOD loads from relevant pollution sources.

All parameters but one were held constant, that one being increased or decreased by 50%. Some results of parameter sensitivity analysis are shown in Figure 11a($K_1$), Figure 11b($K_L$) and Figure 11c (VBOD).

A review of the sensitivity analysis results for dissolved oxygen concentrations showed the following: (a) Dissolved oxygen sensitivity to all associated input parameters increased in the downstream direction; (b) In general, simulated dissolved oxygen concentration was sensitive to the input parameters in the following order from most sensitive to least: mass transport coefficient for reaeration ($K_L$), BOD decay rate ($K_1$), sediment oxygen demand (SOD) rate, BOD concentration (BOD load) from upper boundary, other parameters; (c) The relatively low sensitivity of dissolved oxygen to velocity of BOD sedimentation.

The results of the dissolved oxygen sensitivity analysis, in addition to providing a perspective on the possible uncertainty of model results as a function of input uncertainty, provided a basis for estimating how changes in these parameters due to external influences could change dissolved oxygen concentration in the river. For example, changes in river channel configuration which tend to reduce the reaeration coefficient would have a potentially major impact on dissolved oxygen concentration in all reaches (Tishler et al., 1984).

Similar sensitivity analyses were performed on a simulated BOD. The results showed that:
(a) BOD sensitivity to $K_1, K_L$ and SOD increased significantly in the downstream direction, but BOD sensitivity to BOD load from upper boundary and pollution sources decreased in this direction; and (b) Generally, simulated BOD concentration was sensitive to the input parameters in the following order from most sensitive to least: BOD concentration (BOD loads) from the upper boundary, BOD load from pollution sources, BOD decay rate ($K_L$), other parameters.

The sensitivity of BOD to velocity of BOD sedimentation (VBOD), sediment oxygen demand (SOD), and mass transport coefficient for reaeration $K_L$ was relatively low. It shows BOD sedimentation is not an important sink under current condition in the ZW River system.

## 3. Development of Pollution Control Measures

### *3.1 Projection for Pollution Sources*

For projection of wastewater discharge and pollutant load, the following data were adopted (Stanley Associates Engineering Ltd. et al., 1995): (1) the annual population increase rate (1.2%); (2) domestic sewage discharge and non-residential sewage discharge rate (140 l/day/cap. in 2000 and 160 l/day/cap. in 2010); (3) the average annual industrial output value (IOV) increase rate (8%); (4) the annual wastewater discharge decrease rate (5%); (5) the annual amount of pollutant (COD, BOD) load per unit of IOV decrease rate (5%).

The method of industrial output value (IOV) was used to forecast the industrial wastewater discharge and to take into account the wastewater discharge and the pollutant concentration of a unit of IVO decreasing annually due to the development of water saving techniques, improvement of water re-use rate and industrial processes. The quota of daily water consumption per capita was used to forecast the domestic sewage discharge and also take into account the increasing water demand due to the growth of population and improvement of living standards.

The results of the projection of wastewater discharge and pollutant load are given in Table 8 and Table 9. From the projection results, although the pollutant concentration remains almost unchanged, the wastewater discharge and the amount of pollutant load increased about 2-fold from 1991 to 2010. It is obvious that without a comprehensive water quality control and management strategy, the water quality of the river basin will deteriorate further in view of the increase in pollutant loads generated from an increase in population, industrial and other activities.

### *3.2 Water Quality Objectives*

In order to improve and protect the ambient water quality in the ZW River system, it is important firstly to identify the appropriate water quality objectives in the system.

According to China Surface Water Environmental Quality Standards (SWEQS), surface water bodies are divided into five classes depending on their beneficialness. If water bodies provides multiple uses, it is classified according to the highest beneficial use. According to the water supply purpose in ZW River system, the water quality in Linqing needs to reach Class III. In the upstream of Linqing, the water quality in the upper part of Linqing needs to meet Class IV requirement (HRWCC, 1995). At present, the Zhanghe river water can reach Class II, but it is expected that the water in this region will deteriorate further in the future (Zheng, 1997).

### *3.3 Possible Pollution Control Measures*

Construction of wastewater treatment plants is the key pollution control measure to improve water quality status. Unfortunately, in the ZW River basin the wastewater treatment facilities are very poor. Most wastewater and domestic sewage are discharged directly into the river system without treatment (Wang, 1995). The details of options and realisation of possible measures are beyond the range of this study. It was just assumed that the proposed four scenarios would arise after those practical pollution control measures are carried out in the part or whole of the ZW River system region.

The four scenarios are developed as following:

1. the water quality from the upper boundary is to meet Class III according to the SWEQS (but no pollution control measure is carried in the study river system);
2. all the waste water discharged from the four pollution sources meet the national Integrated Wastewater Discharge Standards (IWDS); the water quality from upper boundary remains unchanged;
3. the water quality from the upper boundary reaches is raised to Class III and the waste discharge from the four pollution sources meets IWDS requirements;
4. the water quality from the upper boundary reaches meets the requirements of Class III; all the waste water will be treated by primary treatment plants to meet the irrigation requirement, then will be conducted to special areas for agriculture irrigation use or flow into the Bohai Sea by special canal directly, and not flow into river system any more.

### *3.4 Simulation of Effectiveness for the Scenarios*

The four scenarios discussed above are simulated with the calibration/verified SOBEK model. All the simulations were made based on the year 1991 data. The simulation results show the third scenario and fourth scenario improve the water quality in 1991. Therefore, these two practical scenarios are chosen to apply to the years 2000 and 2010. The results are summarised in Table 10.

From the simulation results of different scenario, the following general remarks can be drawn:
(1) The water quality in study river system is affected significantly by the pollutants from the upper boundary; (2) Each scenario causes more improvement to water quality in the winter season than that in the summer season; (3) DO has higher concentration in the winter season than that in the summer season and BOD decays more in the summer season; and (4) Both the third and the fourth scenarios cause water quality improvements that meet water quality objectives not only in the basic year (1991) but also in the future (2000 and 2010).

In each scenario, BOD loads in the water from the upstream boundary and/or from the pollution sources were changed. These changes normally will lead to the change of sediment oxygen demand (SOD). Due to many reasons, the relation between them is very difficult to be determined (at least special experiments need to be done for that). In this study, this relation was not involved. If this relation was considered, the simulated dissolved oxygen concentration for different scenarios will be higher (low BOD loads leads to low sediment oxygen demand).

## CONCLUSIONS

The main conclusions drawn from the study on the ZW River system can be summarised as follows: (1) The four water quality processes considered in this study can represent the behaviour of this water quality system; (2) The water quality in the ZW River system is affected very significantly by the pollutants coming from the upper boundary. If the water from upstream cannot be improved properly, it is impossible to reach water quality objectives in the ZW River system; (3) Dissolved oxygen (DO) was very sensitive to the mass transport coefficient for reaeration, BOD decay rate and sediment oxygen demand. BOD concentration was quite sensitive to BOD loads from the upper boundary and from pollution sources, and BOD decay rate; (4) In the third scenario, the water from the upper boundary meets Class III requirement according to the Surface Water Environmental Quality Standards (SWEQS) and the wastewater from pollution sources meets the national Integrated Wastewater Discharge Standards (IWDS). In the fourth scenario, besides the water from upper boundary meets SWEQS requirement, the wastewater from pollution sources will be treated to meet irrigation requirements and used for irrigation purposes, and the extra treated wastewater will be conducted into a special canal system and flow into the Bohai sea. Both the third and fourth scenarios are practical and can cause large improvements to the water quality and meet the water quality objectives in the ZW River system. Relatively, the fourth one is slightly better than the third one.

## ACKNOWLEDGEMENTS

The authors thank the following for providing data used in this study: Ministry of Water Resources, China; Haihe River Water Conservancy Commission; Zhang-Wei River Administration Bureau. This study was funded by “Integrated River and Lake Basin Management for Pollution Control” in International Institute for Infrastructural, Hydraulic, Environmental Engineering (IHE), Delft, The Netherlands.

## REFERENCES


Baarse, G. and Kerssens, P. J. M., 1990, *Water Quality Problems in China: Save Our Sources*, Paper presented at the 7th Conference of the Asia/Pacific Division of International Association for Hydraulics Research, Beijing.

Baity, H. G., 1983, *Some Factors Affecting the Aerobic Decomposition of Sewage Sludge Deposits*, Sewage WorksJ. 10:539-568.
Chow, V. T., 1959, *Open-Channel Hydraulics,* McGraw-Hill.

Delft Hydraulics & RIZA, 1996, *SOBEK - Technical Reference Guide*, Version 1.10.

Delft Hydraulics & RIZA, 1994, *SOBEK Introduction*.

Diskin, M.H. and Simon, E., 1977, *A Procedure for the Selection of Objective Functions for Hydrologic Simulation Models*, Journal of Hydrology, vol. 34, pp129-149.

Drolc A. and Koncan J. Z., 1996, *Water Quality Modelling of the River Sava, Slovenia*, Wat. Res. Vol. 30, No. 11, pp. 2587-2592.

Fair, G. M., E. W. Moore, and H. A. Thomas, 1941, *The Natural Purification of River Muds and Pollutional Sediments,* Sewage Works J. (13).

Gils, J. A. G., Ouboter, M. R. L., and De Rooij, N. M., 1993, *Modelling of Water and Sediment Quality in the Scheldt Estuary*, Netherlands Journal of aquatic Ecology 27(2-4) 257-265.

HRWCC (Haihe River Water Conservancy Commission), 1995, *Middle and long Term Water Resources Planning of Haihe River Basin*, Tianjin.

Kelderman, P., 1994, *Environmental Hydraulics and Modelling*, IHE Lecture notes, Infrastructural, Hydraulic, Environmental Engineering (IHE), Delft, The Netherlands.

Liu, D. and He., S., 1995, *Water Quality Forecast and Protective Countermeasures for East Route of South Water to North Project,* in Chinese, Wat. Res. Protection, pp.6-11, June.

Minns, A.W., and Hall, M.J., 1996, *Artificial Neural Networks as Rainfall Runoff Models*, Journal of Hydrological Sciences, vol. 41, pp399-417.

Qian, Z., Editor, 1994, *Water resources Development in China, Beijing,* China Water &Power.
Stanley Associates Engineering Ltd., Canada, et al, 1995, China: Haihe Basin EnvironmentalManagement and Planning Study, Volume 4 - Basin Water Quality Control and Management.

Thomann, R. V. and Muller, J. A., 1987, *Principles of Surface water Quality Modeling andControl,* Harper Collins.

Wang, Y., 1995, *Water Resource and present Situation of Water Environment in Haihe River andTentative Work Plan*, in Chinese, Wat. Res. Protection, pp.1-5, June.

Yu, W., 1995, *Effect of Polluted Water in Weihe River on Water Quality of East Route Water t to*

*North Project,* in Chinese, Wat. Res. Protection, pp.21-24, June.

Zheng, F., 1997, *Modelling and Pollution Control for the Zhang-Wei River Basin*, MSc thesis, International Institute for Infrastructural, Hydraulic, Environmental Engineering (IHE), Delft, The Netherlands.

**Table 1. The calculated $F_1$ and $F_2$ for water level at three cross sections along the river calibration, 1991)**

| Factor | Wuling (node3) | Yuanchunji (node4) | Guantao (node 6) |
|---|---|---|---|
| $F_1(\%)$ | 30.14 | 16.38 | 10.1 |
| $F_2$ | 0.40 | 0.76 | 0.93 |

**Table 2. The calculated $F_1$ and $F_2$ for water level at three cross sections along the river (verification, 1988)**

| Factor | Wuling (node3) | Yuanchunji (node4) | Guantao (node 6) |
|---|---|---|---|
| $F_1(\%)$ | 12.67 | 23.67 | 28.18 |
| $F_2$ | 0.88 | 0.88 | 0.70 |

**Table 3. $F_3$ and $F_4$ of DO (Calibration, 1991)**

| | Cai-wan | Xunxian-chiao | Yinzhen-chiao | Guantao | Linqing |
|---|---|---|---|---|---|
| $F_3$ | 0.06 | -0.44 | 2.04 | -0.87 | -0.82 |
| $F_4$ | 0.27 | 0.53 | 3.07 | 1.26 | 1.48 |

**Table 4. $F_3$ and $F_4$ of BODC (Calibration, 1991)**

| | Cai-wan | Xunxian-chiao | Yinzhen-chiao | Guantao | Linqing |
|---|---|---|---|---|---|
| $F_3$ | 0.42 | 2.21 | 2.77 | 0.55 | -0.79 |
| $F_4$ | 1.28 | 2.96 | 3.67 | 2.25 | 1.96 |

**Table 5. $F_3$ and $F_4$ of DO (verification, 1988)**

| | Yuanchunji | Guantao | Linqing |
|---|---|---|---|
| $F_3$ | -0.96 | 1.5 | 0.87 |
| $F_4$ | 5.59 | 1.97 | 2.71 |

**Table 6. $F_3$ and $F_4$ of BODC (verification, 1988)**

| | Yuanchunji | Guantao | Linqing |
|---|---|---|---|
| $F_3$ | -0.66 | -0.9 | -1.14 |
| $F_4$ | 1.77 | 1.27 | 1.77 |

**Table 7. Water quality model parameters**

| Water quality process | Parameters | Value |
|---|---|---|
| Decay of BOD | zero-order minerliaztion flux, $F_0$ (gC.m$^{-3}$.d$^{-1}$) | 0 |
| | first-order mineralization rate, $K_1$(day$^{-1}$) | 0.5 |
| | temperature coefficient,$q$ (-) | 1.047 |
| | optimum oxygen concentration, $C_{op}$ (g/m$^{-3}$) | 6 |
| | critical oxygen concentration, $C_{cr}$ (g/m$^{-3}$) | 2 |
| | critical temperature, $T_c$ (°C) | 2 |
| | oxygen function, $F_{cl}$ (-) | 0.3 |
| BOD sedimentation | zero-order sedimentation rate, $f_0$(g.m$^{-2}$.d$^{-1}$) | 0 |
| | sedimentation velocity of BOD, $V_{sed}$ (m.d$^{-1}$) | 0.1 |
| | critical shear stress, $T_{cbss}$(Pa) | 0.1 |
| Reaeration of oxygen | mass transport coefficient for reaeration, $K_L$ (m.d$^{-1}$) | 0.3 |
| | reaeration temperature coefficient, $q$ (-) | 1.016 |
| Sediment oxygen demand | zero-order sediment oxygen demand, $S_B$($gO_2 / m^2 / d$) | 1 |

**Table 8. Projection of wastewater discharge**

| | 1991 wastewater discharge ($\times 10^3$m$^3$ / day) | | | 1990 wastewater discharge ($\times 10^3$m$^3$ / day) | | | 1990 wastewater discharge ($\times 10^3$m$^3$ / day) | | |
|---|---|---|---|---|---|---|---|---|---|
| **City** | **domestic** | **Industry** | **total** | **domestic** | **industry** | **total** | **domestic** | **industry** | **total** |
| Huaxia | 0.5 | 7.9 | 8.4 | 1.58 | 10.20 | 11.78 | 2.03 | 13.19 | 15.22 |
| Xunxia | 3.1 | 13.6 | 16.7 | 9.42 | 17.58 | 27.00 | 12.13 | 22.73 | 34.85 |
| Daming | 1.8 | 8.3 | 10.1 | 4.95 | 10.73 | 15.69 | 6.38 | 13.88 | 20.25 |
| Guantao | 1 | 3.3 | 4.3 | 4.13 | 4.27 | 8.40 | 5.32 | 5.52 | 10.84 |

**Table 9. Projection of concentration of pollutants (COD, BOD)**

| | 1991 | | | 2000 | | | 2010 | | |
|---|---|---|---|---|---|---|---|---|---|
| City | Q ($m^3$/s) | COD (mg/l) | BOD (mg/l) | Q ($m^3$/s) | COD (mg/l) | BOD (mg/l) | Q ($m^3$/s) | COD (mg/l) | BOD (mg/l) |
| Huaxia | 0.10 | 2203.57 | 222.83 | 0.14 | 2022.80 | 204.55 | 0.18 | 2016.47 | 203.91 |
| Xunxia | 0.19 | 1548.50 | 156.59 | 0.31 | 1216.13 | 122.98 | 0.40 | 1198.56 | 121.20 |
| Daming | 0.12 | 1331.68 | 134.66 | 0.18 | 1094.02 | 110.63 | 0.23 | 1082.89 | 109.51 |
| Guantao | 0.05 | 2655.81 | 268.57 | 0.10 | 1741.59 | 176.12 | 0.13 | 1731.97 | 175.14 |

**Table 10. Summary of the effectiveness for four scenarios**

| Scenario | 1991 | 2000 | 2010 |
|---|---|---|---|
| The first scenario | No | | |
| The second scenario | No | | |
| The third scenario | Yes | Yes | Yes |
| The fourth scenario | Yes | Yes | Yes |

No : the scenario can not cause large improvement to water quality to meet the water quality objectives

Yes : the scenario can cause large improvement to water quality to meet the water quality objectives

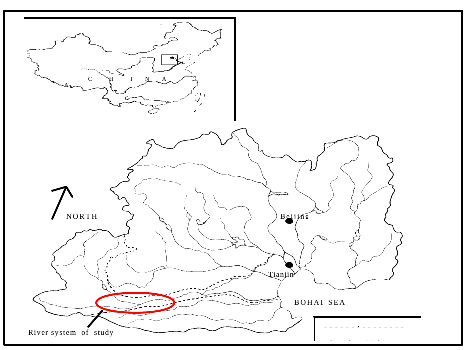



**Figure 1 Haihe River Basin, ZWS Sub-river Basin location and study area position**

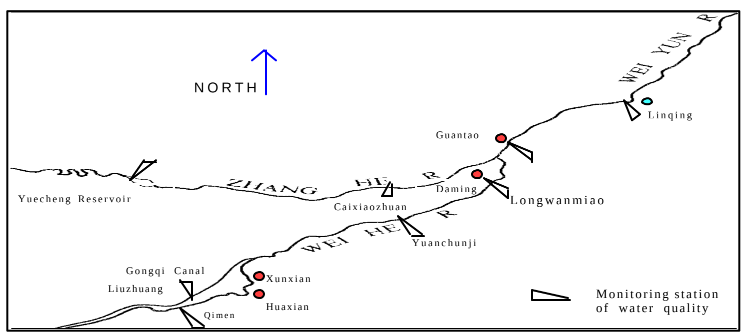



**Figure 2 Study river system and water quality**

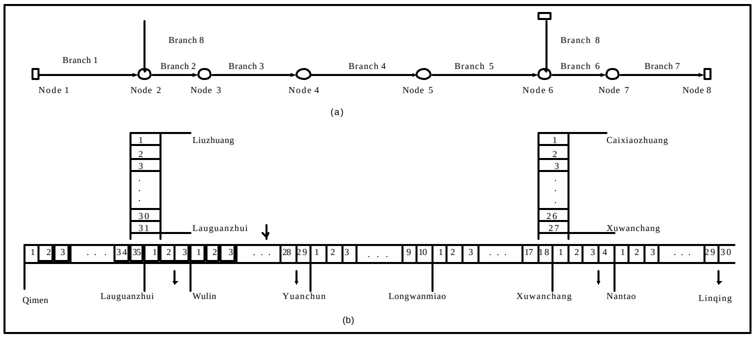


**Figure 3 Nodes, branches and grids for water flow model: (a) nodes and branches of river system; (b) Grids, computational elements (between grids and node) of river system**

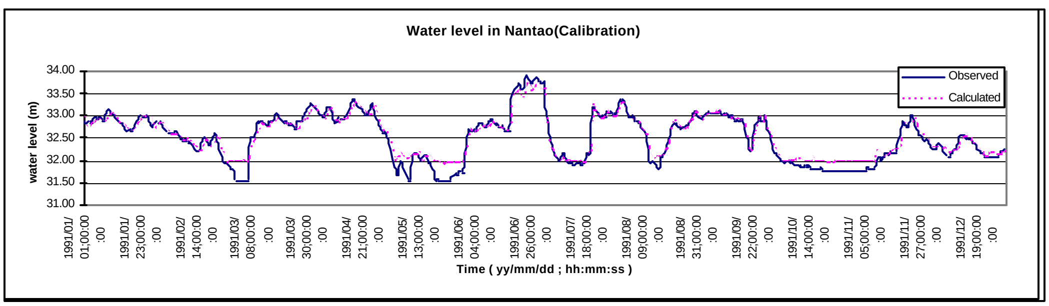


**Figure 4 Simulated water level at Nantao (Calibration)**

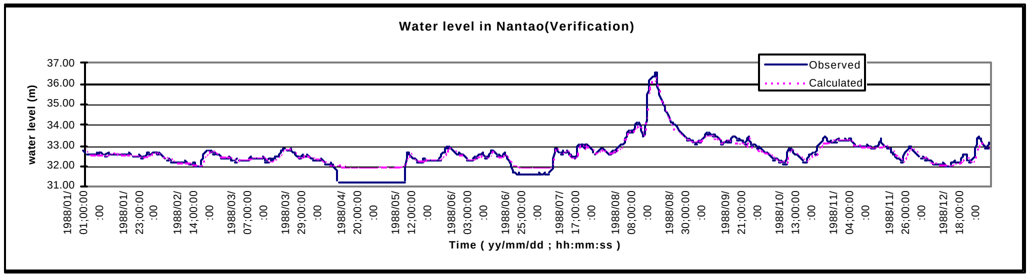



**Figure 5 Simulated water level at Nantao (verification)**

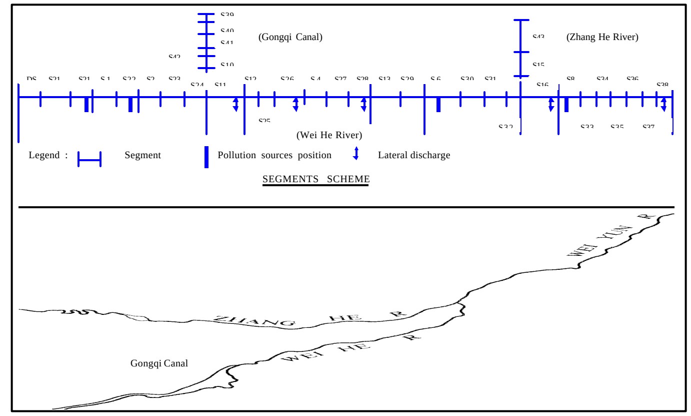



**Figure 6 Definition of segment in the ZW River system**

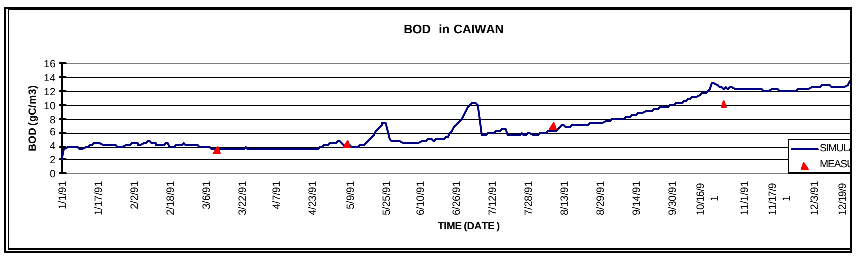



**Figure 7a**

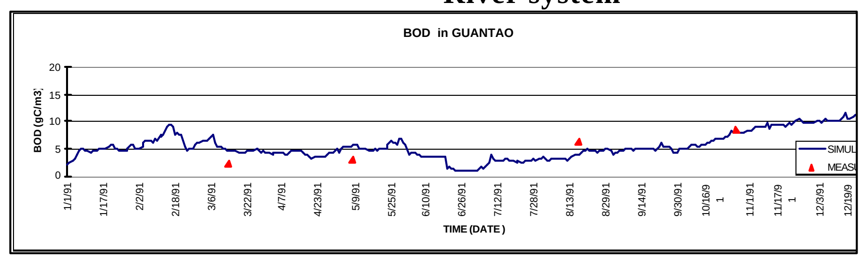



**Figure 7b**

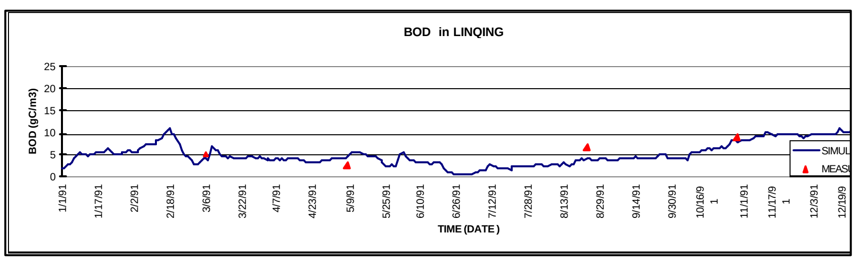



**Figure 7c**

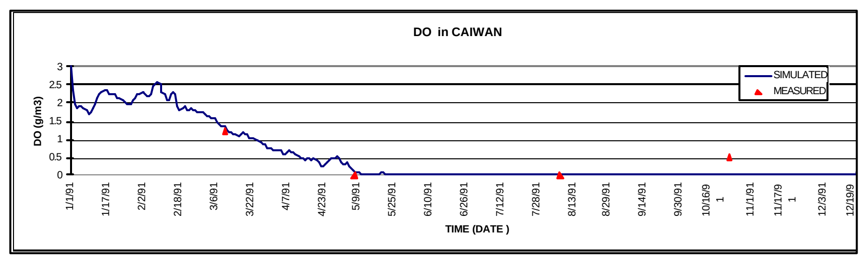



**Figure 8a**

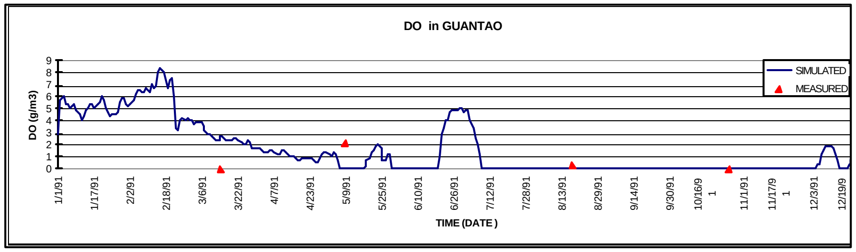



Figure 8b

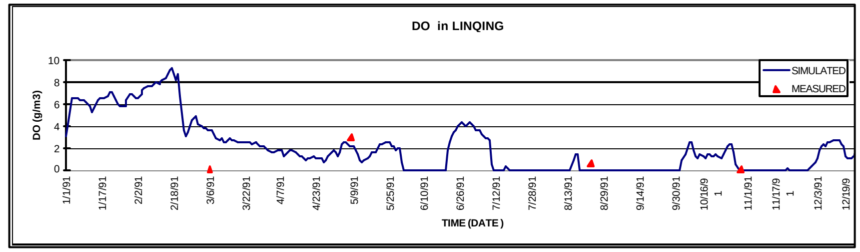



Figure 8c

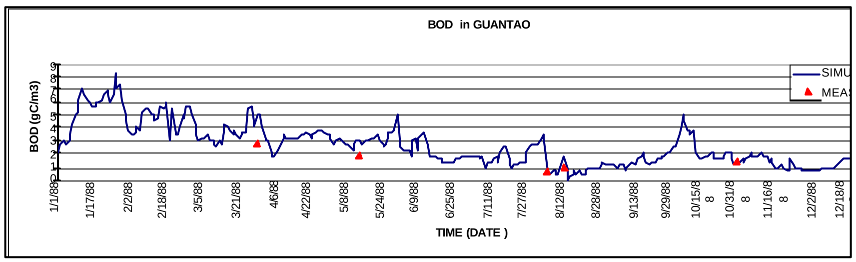



Figure 9a

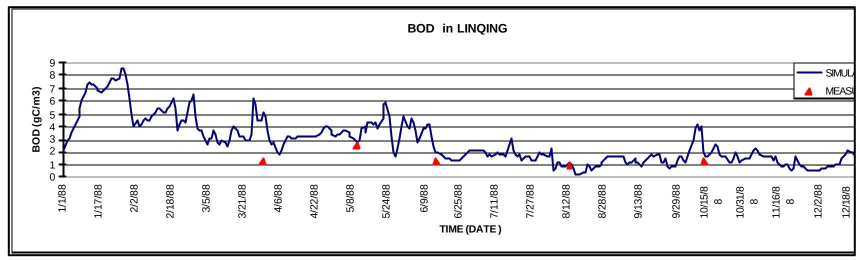



Figure 9b

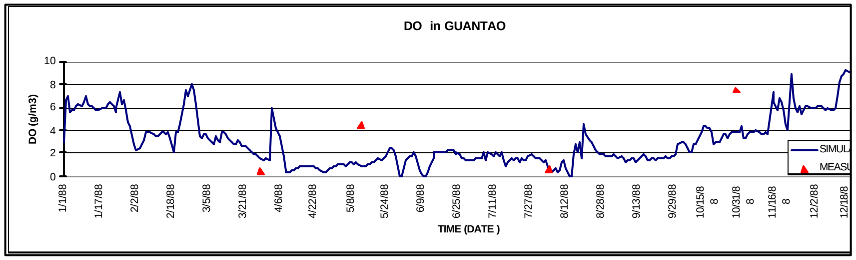



Figure 10a

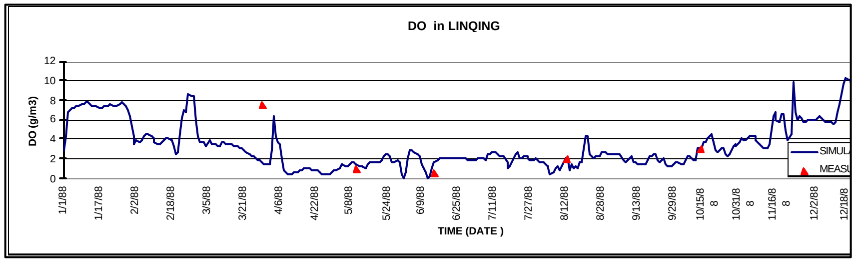



Figure 10b

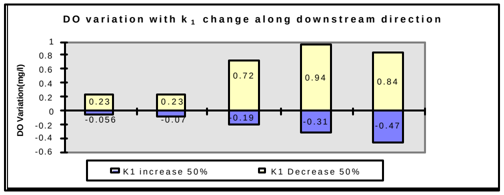



Figure 11a

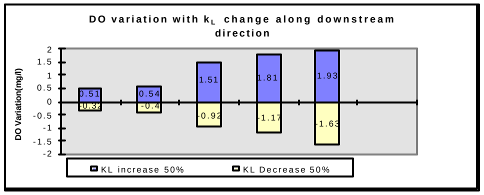



Figure 11b

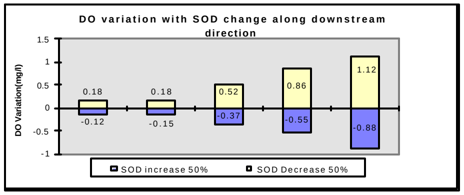



Figure 11c

# A Study of Water-Quality Trends and Watershed Characteristics

C.E. Zipper
Assistant Professor, Department of Crop and Soil Environmental Sciences

G.I. Holtzman
Associate Professor, Department of Statistics

J.J. Gildea and M. Schneider
Former Graduate Research Assistants, Dept of Crop and Soil Environmental Sciences

E-mail: czip@vt.edu

## EXTENDED ABSTRACT

The purpose of this research was to investigate statistical associations between water-quality and variables representing watershed characteristics associated with economic and land-use activity.

## OBJECTIVES

1. Obtain data on population, land use, economic activity, and point-source discharges for political jurisdictions and geographic areas of Virginia. Organize that information within a geographic database framework.
2. Identify watershed areas above each monitoring station for which trend data were analyzed; quantify economic and land-use factors for each watershed area.
3. Investigate the presence of statistical relationships between economic and land-use factors and water-quality change.

## METHODS

Water Quality Variables:

Data representing the 1978 - 1995 period from 180 Virginia water-quality monitoring stations maintained by Virginia Department of Environmental Quality and 7 Virginia monitoring stations maintained by US Geological Survey were analyzed for trend using a seasonal Kendall Tau rank correlation test. Kendall's tau is a statistic that represents the strength or consistency of water-quality trends. Values of tau vary between +1 and -1. Positive values indicate the presence of increasing trends, while negative tau values indicate the presence of decreasing trends and tau-values close to zero indicate a strong likelihood that there is no trend. In data analysis, tau was used to represent trend. Monitoring stations were selected for analysis based on availability of long-term monitoring data and the goal of achieving statewide distribution.

Data for nine water-quality characteristics were analyzed: dissolved oxygen saturation (DO), biochemical oxygen demand (BOD), pH, total residue (TR), non-filterable residue (NFR - represents suspended solids), nitrate-nitrite nitrogen (NN), total Kjeldahl nitrogen (TKN), total phosphorus (TP), and fecal coliform (FC). Values for Kendall's tau and median values were generated for each of the nine water-quality variables at each of the 187 water-quality monitoring stations for which sufficient data were available. The resulting values of Kendall's tau were screened, and Kendall's tau values characterized by low precision and/or poor representation of the full 18-year time period were eliminated from the subsequent analyses. Nineteen monitoring stations were also eliminated from further analysis for reasons that include insufficient water-quality data and watershed areas smaller than 7 square miles. Median values of the nine water quality characteristics were also determined at each monitoring station.

Land Use Variables:

U.S. Census data from 1990 were used to generate 3 socioeconomic variables (population density, per-capita income, 1990 unsewered household densities) for 151 monitoring-station watersheds with more than 80-percent in-state area.

Variables representing agricultural intensity were generated for 151 monitoring-station watersheds with more than 80 percent in-state area. The variables represented potential fertilizer applications to agricultural land areas and estimated manure production within the watershed. Separate variables were generated to represent nitrogen, phosphorous, and total-nutrients. The variables were calculated on both per-watershed-acre and per-agricultural acre bases to represent potential 1991-92 applications. The primary data sources were the Census of Agriculture which tabulates agricultural data for city-county geographic units, and the Virginia Department of Agriculture and Consumer Services fertilizer sales data.

Five variables representing land-use and land-use change were generated at 168 monitoring stations from an early 1990s statewide land-use coverages. Variables representing major land uses (urban, forest, pasture, cropland, and agriculture) were generated, with the "agriculture" land-use variables representing combined pasture and cropland. Land-use variables were expressed as a percentage of total watershed area.

In order to investigate relationships between watershed variables and water quality trends, correlation analyses were performed. In interpreting correlation results, population density and per-capita income were considered as associated with urban land uses, while animal-manure production and fertilizer applications were considered to be associated with agricultural land uses. Conclusions were drawn only when the results of these analyses revealed consistent patterns of watershed – water quality relationships.

Presence or absence of point sources:

Monitoring station watersheds were coded based on the presence or absence of major point sources discharging to stream segments located upstream from the monitoring station, and discharging to stream segments located upstream and within 10 stream-miles of the monitoring station. Separate analyses were conducted for point sources (regardless of type), and for municipal and industrial point sources separately. For each of these six point-source locations and 18 water-quality taus and medians, analyses were conducted to determine whether or not statistically significant differences between the mean tau/median values are present.

## RESULTS

Over the 1978 – 1995 period, water quality data at the monitoring stations used in this study exhibited general tendency toward improvement with respect to BOD, FC, NFR, and TP. There was little change, on average, in pH and DO, while TR, TKN, and NN exhibited predominance of increasing trends. A large number of TP observations at many monitoring stations were found to have been recorded as detection-limited levels, limiting the sensitivity of TP analysis.

Forest land uses were found to be associated consistently with good and improving water quality. Forested land use was found to be positively correlated with pH-tau and pH–median, and with DO-tau and –median; and negatively associated with BOD-tau and BOD–median, FC-tau, NFR-median, NN-median, TKN-median, TR-tau, and TR–median.

Urban land use, and socioeconomic characteristics associated with urban uses, were found to be associated with water of poorer quality, on average, than that typifying forested areas, and with either water-quality deterioration or failure to exhibit water-quality improvements to the same degree as did forested areas. Variables representing urban land uses were found to be positively correlated with BOD-tau, BOD-median, FC-tau, NFR-median, NN-median, TKN-median, and TR-tau, and negatively correlated with pH-tau.

In addition to potential land use effects, the data regarding urban and forest land uses also reflect Virginia's landscape and terrain. In Virginia, geographic biases affect both land use and water quality, as the states' western mountains that serve as headwaters for its major rivers are heavily forested relative to coastal areas where major population centers are located. Water in headwater streams tends to of higher quality, generally, than that downstream as a natural consequence of watershed processes. BOD-, NFR-, TKN-, and TR- median levels tend to be higher in the eastern Virginia, while DO- and pH-median levels tend to be higher in the western Virginia.

Animal agriculture (as represented by livestock manure-production potential and pasture) was found to be associated with good water quality, relative to most of the non-nutrient medians and TKN, while cropland agriculture was found to be associated with water of poorer quality than that the state's forested areas with respect to all of these variables. However, these patterns are consistent with the geographic biases affecting Virginia's water-quality, as discussed above and may not be reflective of land-use effects.

Results of trend analysis indicate cause for concern regarding surface-water nitrogen, as both NN and TKN exhibited positive tau values at more locations than those where negative values of tau are evident. The analysis showed strong and consistent relationships between NN-tau, NN-median, TKN-tau, and TKN-median with variables representing agricultural intensity and agricultural land uses. These relationships exhibited regional differences, tending to be strongest in western Virginia. Both TKN-median and TKN-tau also exhibit geographic biases, as both tend to be higher in the eastern part of the state. TKN-median also exhibits strong, positive correlations with variables representing urban land uses; like TKN-median, these variables are heavily biased towards the east.

Monitoring stations below major point sources exhibited fecal coliform medians that were lower, on average, than other monitoring stations. Monitoring stations located below and within 10 miles of major municipal point sources exhibited BOD-medians, NN-medians, and TP-medians that were higher, on average, than other stations studied, and were characterized by NFR-taus that were more negative, and TKN-taus that were less positive, than other stations studied.

Although the results of analyses are generally consistent with one another, and when taken in aggregate appear to provide intelligible patterns, they are not strong indicating that many factors other than those studied influence water quality. In most of the statistical relationships observed, watershed characteristics account for only a small percentage of the total variation in the water-quality variables studied. Several of the water-quality variables studied -- including BOD-tau, FC-median, pH-median, and TKN-median -- exhibited relationships with longitude that are as strong or stronger than any of the relationships with the watershed characteristics. Correlation analysis results should not be interpreted as indicating causality.

## SUMMARY

In Virginia, land use patterns tend to be strongly affected by geography and terrain; together, these factors demonstrate a number of relationships to median water quality. At the monitoring stations studied, DO- and pH-medians tended to be higher, on average, in Virginia's forested and mountainous regions, while higher values of BOD-, TKN-, TR-, and NFR-medians tended to occur in eastern Virginia and to be associated with the state's more urbanized regions.

Over the 1978 - 1995 period at the monitoring stations studied, trends representing water-quality improvement were more numerous than those representing water-quality deterioration for BOD, FC, NFR, and TP, while increasing trends outnumbered declining trends for NN, TKN, and TR. Forested land uses and low per-capita incomes were found to be associated with declining BOD and FC trends, while urban land use and socioeconomic characteristics associated with urban use were found to be associated with either water-quality deterioration, or failure to exhibit water-quality improvements to the same degree as did forested areas, with respect to FC and TR. Positive associations were found to occur between variables representing agricultural land uses and water-quality nitrogen medians and trends (both NN and TKN), but analysis of these results revealed strong regional differences.