

VIRGINIA WATER RESOURCES RESEARCH CENTER

**VIRGINIA
WATER RESEARCH
SYMPOSIUM
2004**

**Assessing and Managing
Virginia's Water Resources**



Virginia
WATER RESOURCES
Research Center

PROCEEDINGS



**VIRGINIA POLYTECHNIC INSTITUTE AND STATE UNIVERSITY
BLACKSBURG, VIRGINIA**

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PROCEEDINGS

Virginia Water Research Symposium 2004

Assessing and Managing Virginia's Water Resources

October 4-6, 2004

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Virginia Tech
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RESERVOIR DYNAMICS: A COMPARISON OF SMITH MOUNTAIN LAKE AND CLAYTOR LAKE

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KEY WORDS: trophic state, total phosphorus, chlorophyll-a, Secchi depth, nutrient dynamics

ABSTRACT

The dynamics and trophic states of Smith Mountain Lake and Claytor Lake are compared with each other and with other southern reservoirs studied by William Walker in his work for the Army Corps of Engineers (Walker 1996).

The Ferrum College Water Quality Lab has collaborated with the Smith Mountain Lake Association (SMLA) since 1987 and with the Friends of Claytor Lake (FOCL) since 1996 to monitor the trophic status of Smith Mountain Lake and Claytor Lake.

Both reservoirs are hydroelectric facilities operated by American Electric Power (AEP), but are very different impoundments. Smith Mountain Lake has an average hydraulic residence time (AHRT) of about a year, while Claytor Lake's AHRT is about a month. SML receives most of its water from two rivers, the Roanoke and the Blackwater, that form the two main channels of the lake. The two channels are nearly perpendicular before their confluence produces the lake's "main basin". Claytor Lake is a "run of the river" reservoir on the New River. In addition to the differences in AHRT and geometric configuration, hydroelectric functions also produce differences in circulation patterns and water quality dynamics. Both reservoirs produce hydroelectric power but Smith Mountain Lake is a "pump-back" facility. The Pigg River joins the Roanoke River just below the dam so that water from the Pigg River is pumped back to mix with Smith Mountain Lake water near the dam. The pump-back system also leads to more frequent variation in the water level in Smith Mountain Lake, which increases lake mixing and destabilizes the shoreline.

Smith Mountain Lake and Claytor Lake will be compared with each other and with other southeastern reservoirs in terms of trophic state parameters (total phosphorus, chlorophyll-*a*, and Secchi depth), non-algal turbidity, and trophic state index. Additional comparisons will be made between Smith Mountain Lake and Claytor Lake by considering the rate of change in trophic status and variation in water quality with distance from the dam. The presentation concludes by considering how the differences might affect strategies for managing water quality in reservoirs.

INTRODUCTION

With the assistance of citizen volunteers and student interns, scientists from the Ferrum College Water Quality Lab have been monitoring the trophic status of Smith Mountain Lake since 1987 and of Claytor Lake since 1996. Both water bodies are reservoirs, owned and operated by American Electric Power (AEP) and were constructed for flood control, generation of hydroelectric power, and recreation. Claytor Lake, on the New River, dates back to the late 1930's, and is a "run-of-the-river" reservoir with an average hydraulic residence time (AHRT) of about one month. Smith Mountain Lake dates back to the late 1960's and has an AHRT of about a year and receives most of its water from two rivers, the Roanoke and the Blackwater, that form the two main channels of the lake. The two channels are nearly perpendicular before their confluence produces the lake's "main basin". In addition to the differences in AHRT and geometric configuration, hydroelectric functions also produce differences in circulation patterns and water quality dynamics. Both reservoirs produce hydroelectric power but Smith Mountain Lake is a "pump-back" facility. The Pigg River joins the Roanoke River just below the dam and water from that river is pumped back to mix with SML water in the main basin and the pump-back system also leads to more frequent variation in the water level in Smith Mountain Lake, which increases lake mixing and destabilizes the shoreline.

The trophic state of a lake depends on the degree of nutrient enrichment. As nutrients accumulate in the lake, algal production increases and, in turn, water clarity decreases. The biomass produced by algae settles in the lake, causing the depletion of dissolved oxygen (DO) in the hypolimnion, even though algal photosynthesis produces oxygen in the epilimnion during the day. The diurnal DO swing becomes more severe because increased algal populations produce more oxygen during the day and consume more oxygen at night.

Trophic status is evaluated by measuring the typical suite of trophic state indicators; total phosphorus, chlorophyll-*a*, and Secchi depth. At both reservoirs, the college has worked cooperatively with the local lake association, *i.e.*, the Smith Mountain Lake Association (SMLA) and the Friends of Claytor Lake (FOCL). Samples are collected at permanently designated stations each two weeks during the period between Memorial Day and Labor Day. Trained volunteer monitors collect the samples and measure water clarity from a boat during a one-week sampling window and student interns analyze the samples at the Ferrum College Water Quality Lab.

Bob Carlson developed algorithms to calculate a trophic state index based on algal biomass (Carlson 1977). However, Carlson's Trophic State Index (TSI) can be calculated from the seasonal average value of any one of the three trophic state parameters; total phosphorus concentration (TP) as the indicator of nutrient enrichment, chlorophyll-*a* concentration (CHA) as the indicator of algal biomass, or Secchi depth (SD), as the indicator of water clarity. If a lake or reservoir were functioning in classic fashion, the three TSI values (TSI-TP, TSI-CHA, and TSI-SD), calculated from each of the three trophic state parameters, would be similar. The average of the three TSI values is the combined trophic state index, TSI-C.

More recently, William Walker studied 41 southern reservoirs in his work developing a reservoir model (BATHTUB) for the Army Corps of Engineers (Walker 1996). He found that insights

into non-classical behavior could be gained by comparing relative values of TP, CHA, and SD in the study set.

The full trophic state data set from Smith Mountain Lake (1987-2004) and Claytor Lake (1996-2004) has been summarized, compared with each other and with average values for the 41 southern reservoirs included in the Walker study.

METHODS

Field Procedures

Volunteer monitors measure water clarity with a Secchi disk and collect integrated samples of the photic zone. The photic zone is operationally defined as twice the Secchi depth (~ 95% light extinction) and the integrated sample is collected with a rubber hose that has been conditioned in lake water, marked at one-meter intervals, and fitted with a rope and diver's weight. The water sample is mixed in a 4-L polyethylene bucket and an aliquot is placed in a 60 mL polyethylene bottle for total phosphorus analysis. A second 100 mL aliquot is filtered through a type-A glass filter and the filter is analyzed for chlorophyll-*a*. The procedures used by the volunteer monitors are described in detail in the *Smith Mountain Lake/Claytor Lake Volunteer Monitoring Manual* (Thomas and Johnson 2003).

Laboratory Procedures

Analytical methods are adapted from *Standard Methods for the Examination of Water and Wastewater* (APHA 1995). Total phosphorus is measured spectrophotometrically after persulfate digestion and chlorophyll-*a* is measured fluorometrically after acetone extraction. The detailed methods are described in the *Ferrum Water Quality Lab Procedures Manual* (Johnson and Thomas 2004).

RESULTS AND DISCUSSION

Comparison of Smith Mountain Lake and Claytor Lake to the Reservoirs in Walker's Study

Overall average values for total phosphorus, chlorophyll-*a*, and Secchi depth are displayed in Table 1 for Smith Mountain Lake (SML), Claytor Lake (CL), and the Walker study (WS). Trophic state data for SML and CL is grouped by zone, with each zone representing a 5-mile length of reservoir. Zone 1 is from 0 – 5 miles from the dam and so on; Smith Mountain Lake has 6 zones and Claytor Lake has 4 zones. In Table 1, the average value for SML and CL is the average for all zones, the minimum value is the average value for the zone with the lowest average, and the maximum is the average value for the zone with the highest average. Both local reservoirs have lower concentrations of total phosphorus and chlorophyll-*a* and greater Secchi depths than the average of the 41 reservoirs included in the Walker study. As expected, the range of values in the 41 reservoirs in Walker's study is larger than the range found in SML and CL.

Table 1. Average values for trophic state parameters for Smith Mountain Lake, Claytor Lake, and the Walker study.

Total Phosphorus (ppb)				Chlorophyll-a (ppb)				Secchi Depth (m)			
	min	max	avg		min	max	avg		min	max	avg
Smith Mtn Lake	20	64	38	Smith Mtn Lake	1.8	11.3	5.3	Smith Mtn Lake	3.2	1.3	2.0
Claytor Lake	30	57	44	Claytor Lake	1.9	8.4	6.5	Claytor Lake	1.0	1.6	1.3
Walker Study	10	274	48	Walker Study	2.0	64.0	9.4	Walker Study	0.2	4.6	1.1

As part of the model development, Walker examined several relationships among the three trophic state parameters and three of the relationships are displayed in Figure 2 with brief interpretations in the boxes below. Average values calculated for the diagnostics for Smith Mountain Lake and Claytor Lake are neither “high” nor “low”, indicating classical behavior “on average”. However, the minimum values of CHA*SD and CHA/TP in SML and CL indicate a low response to nutrients in some zones of SML and CL.

Table 2. Diagnostic variables used in the Walker study with values for Smith Mountain Lake and Claytor Lake.

Non-algal Turbidity (1/m)				CHA*SD (mg/m ²)				CHA/TP			
	min	max	avg		min	max	avg		min	max	avg
Smith Mtn Lake	0.27	0.52	0.42	Smith Mtn Lake	5.6	14.0	8.7	Smith Mtn Lake	0.09	0.17	0.13
Claytor Lake	0.40	1.00	0.60	Claytor Lake	1.9	13.6	9.2	Claytor Lake	0.03	0.25	0.2
Walker Study	0.13	5.2	0.61	Walker Study	1.8	31.0	10.0	Walker Study	0.04	0.6	0.2

<p>NAT = 1/SD - 0.025*CHA Inverse SD corrected for light extinction by CHA Low: < 0.4; allocthanous PM unimportant? high response to nutrients High: >1; allocthanous PM important? low response to nutrients</p>	<p>mean CHA * mean SD (mg/m²) Light extinction; algal & non-algal turidity Low: < 6; nonalgal turbidity dominates expect low nutrient response High: > 16; algal turbidity dominates expect high nutrient response</p>	<p>mean CHA/mean TP Algal use of phosphorus supply Low: < 0.13; low phosphorus response algae limited by N, light, or flushing rate High: > 0.40; high phosphorus response algae limited by phosphorus</p>
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Temporal Variation of Trophic State Parameters

The trophic states of Claytor Lake and Smith Mountain Lake have not changed significantly since the Ferrum Water Quality Lab began the monitoring programs. Figure 1 displays the combined trophic state index for Claytor Lake.

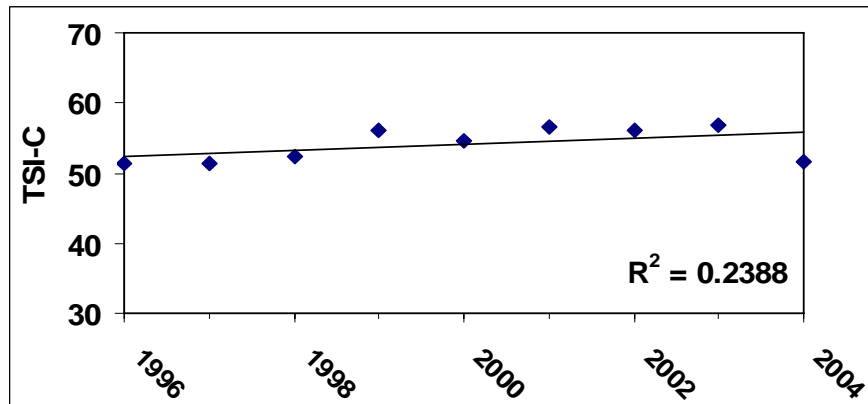


Figure 1. The combined trophic state index for Claytor Lake (1996 – 2004).

Trophic status is evaluated by zone in Smith Mountain Lake because water quality changes significantly as it moves down the long Blackwater and Roanoke channels. Figure 2 displays the trophic state data for Smith Mountain Lake by year for each of the six zones.

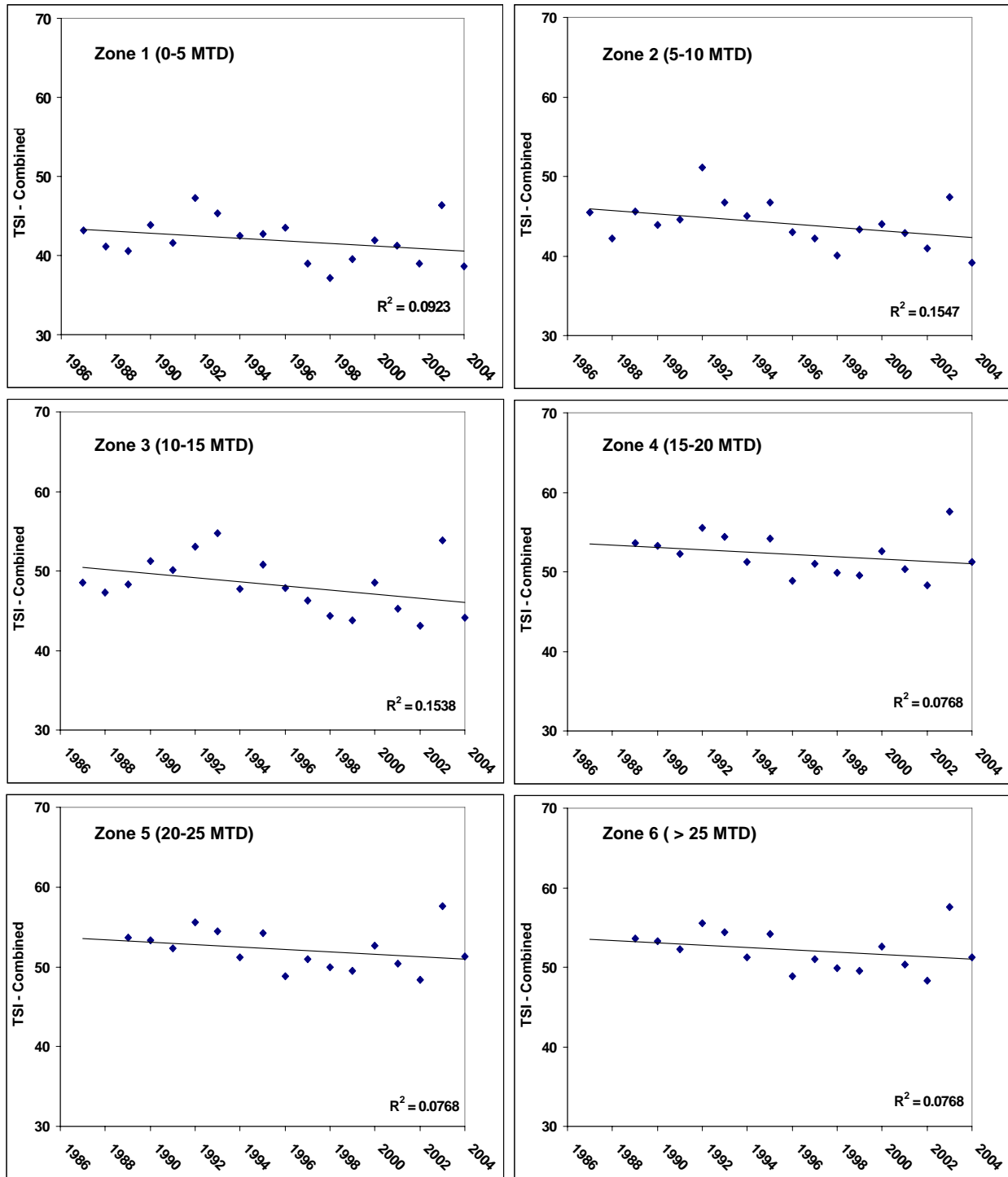


Figure 2. Combined trophic state index for Smith Mountain Lake by zone (1987-2004).

Spatial Variation of Trophic State Parameters

Variation of the three trophic state parameters by zone in Smith Mountain Lake is shown in figures 3, 4, and 5. In each case, the pattern in 2004 is similar to the pattern over the period from 1987 – 2004. The increase in SD towards the dam can be explained by settling of silt in the upper channels of the lake. Phosphate strongly adsorbs to clay particles in the silt and the removal of phosphorus, in turn, leads to reduced CHA. The improved curve fit seen with the second-order regression is consistent with a settling process following first order kinetics.

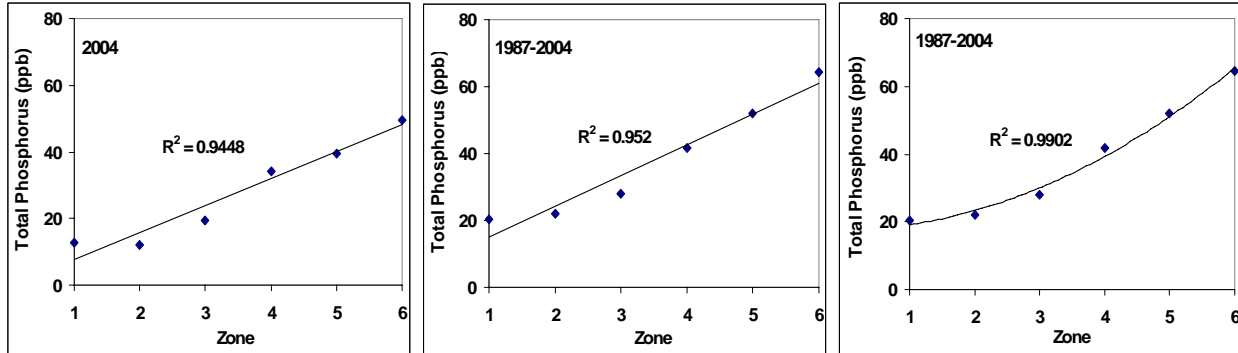


Figure 3. Total phosphorus by zone in Smith Mountain Lake.

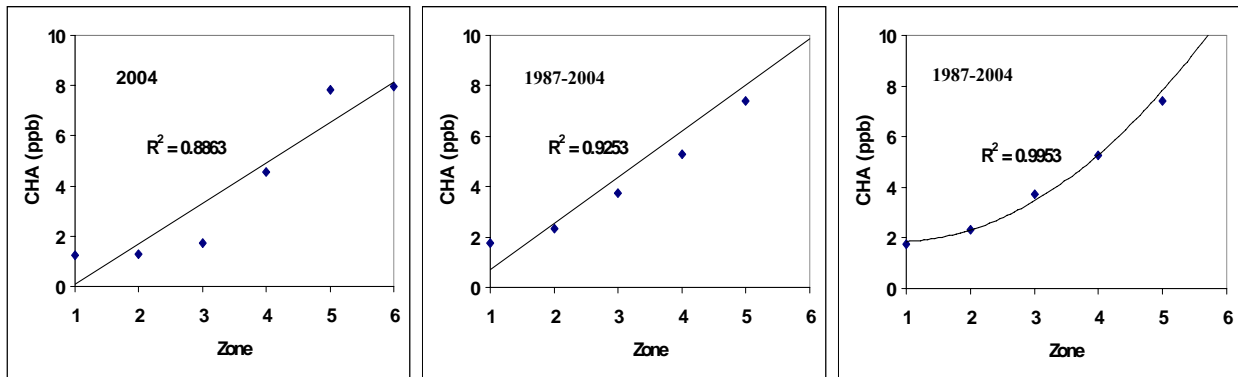


Figure 4. Chlorophyll-a by zone in Smith Mountain Lake.

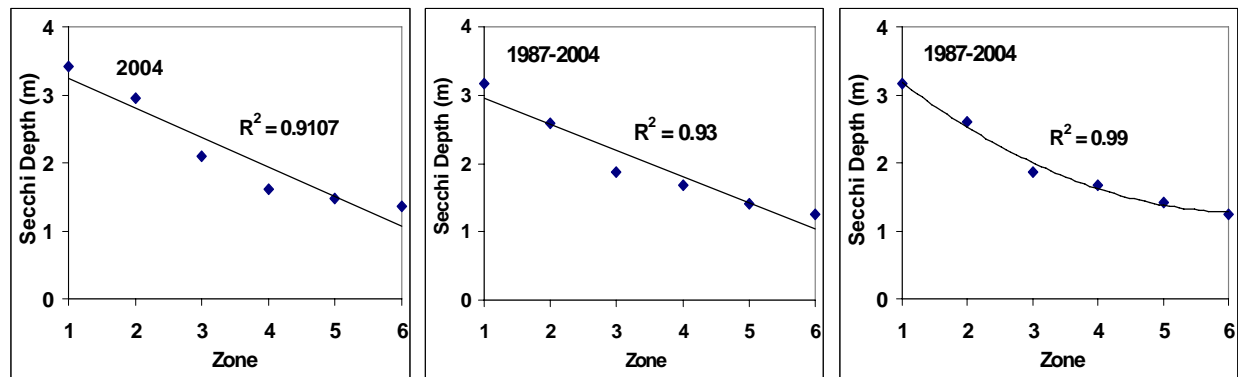


Figure 5. Secchi depth by zone in Smith Mountain Lake.

Figure 6 shows the variation of trophic state parameters with distance from the dam in Claytor Lake. Each data point in the figures below represents one of the 12 sampling sites on the lake. As in Smith Mountain Lake, settling of silt lowers the trophic status of the water near the dam. However, the extremely turbid headwaters of Claytor Lake inhibit algal growth. The three stations furthest from the dam have non-algal turbidities above 1 and much lower CHA levels than would be expected given the high TP concentration at those sites.

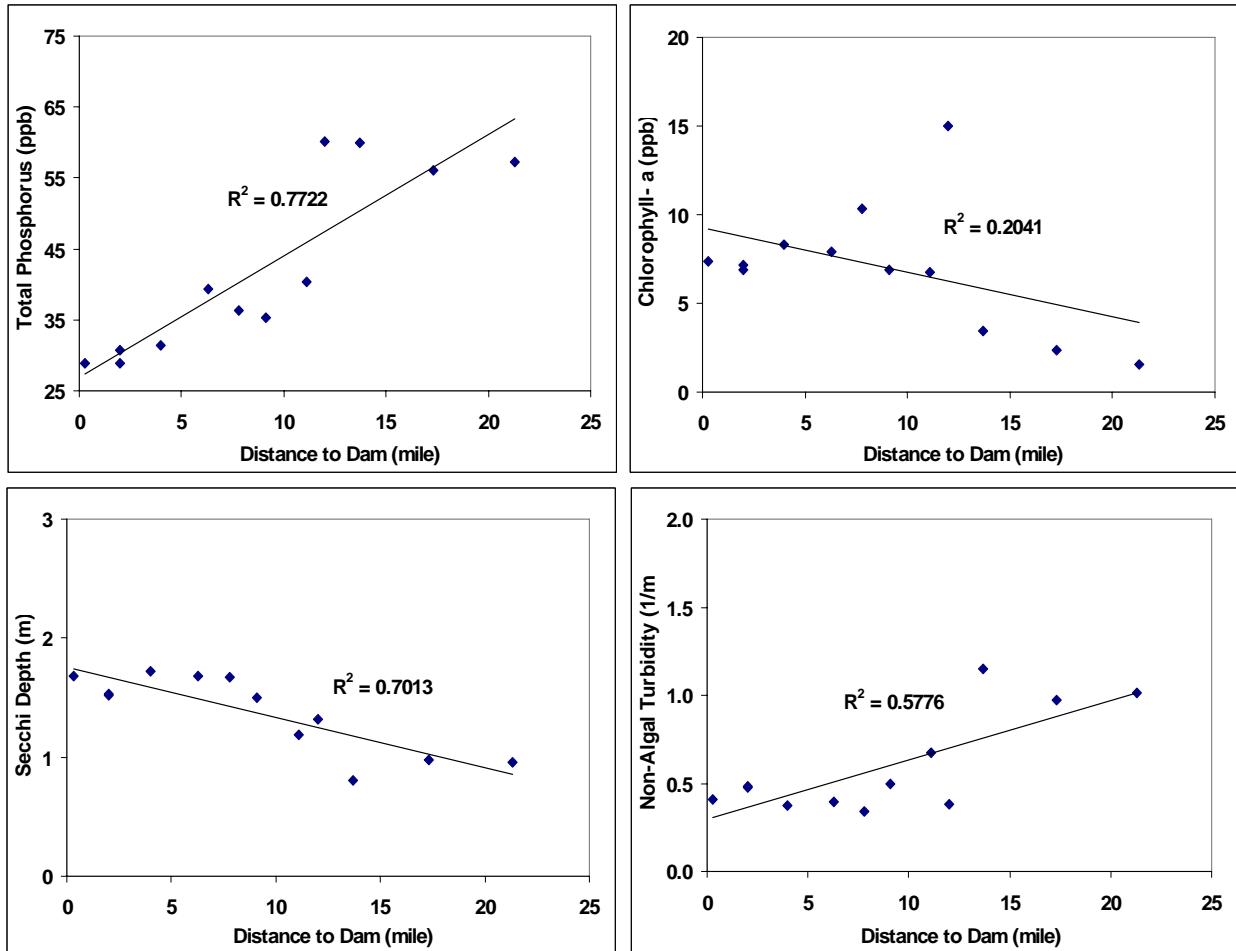


Figure 6. Variation of trophic state parameter with distance from dam in Claytor Lake.

CONCLUSIONS

Smith Mountain Lake and Claytor Lake have the long narrow channels typical of reservoirs. As a result, the water is not as homogeneous as found in classical lakes. As Virginia develops assessment criteria and methodologies for lakes and reservoirs, there is a need to consider the down-channel point at which the riverine channel becomes the lake and should meet state water quality criteria. AHRT of the reservoir should also be considered because “run-of-the-river” reservoirs such as Claytor Lake do not develop the internal nutrient dynamics at work in larger reservoirs such as Smith Mountain Lake.

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Ferrum College
Franklin County
Bedford County
Pittsylvania County
Pulaski County
Smith Mountain Lake Association
Virginia Department of Environmental Quality
Virginia Department of Game and Inland Fisheries
Virginia Environmental Endowment

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STATUS AND TRENDS OF *MICROCYSTIS AERUGINOSA* IN TIDAL FRESH JAMES RIVER, VIRGINIA

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KEY WORDS: algal blooms, James River, *Microcystis*, tidal fresh

ABSTRACT

Over fifteen years of phytoplankton data and environmental parameters were analyzed in the tidal fresh James River. Average abundance of a common, bloom forming Cyanobacteria species, *Microcystis aeruginosa*, was over 10×10^6 cells liter⁻¹ during the summer months. This abundance was five times greater than concentrations reported in the tidal fresh Rappahannock River and over thirty times higher than levels in the tidal fresh York River. While there was an upward trend in Cyanobacteria from 1985 through 2002 in James River, peak abundance of *M. aeruginosa* was during 1985 through 1989, but declined significantly through the 1990s. More recent reports indicate that *M. aeruginosa* abundance reappeared in 2000 through 2002. Both biologically available nitrogen and phosphorus were in excess of limiting thresholds in this region since 1985. While the tidal fresh James experienced higher N:P ratios throughout most of the 1990s, low N:P ratios were observed at least 50% of the time between 1985 to 1988 and from 2000 to 2002. Rapid advective transport in this river could also be regulating this species.

INTRODUCTION

Microcystis aeruginos is a common species of Cyanobacteria (often called blue-green algae). This particular species is colonial. Single cells can join together to form colonies floating near the water surface. Colony sizes may vary from a few to hundreds of cells, but this species is a common bloom-forming algae. Under bloom conditions colonies may exceed 10,000 cells milliliter⁻¹. Blooms usually occur in mid to late summer primarily in nutrient enriched fresh and low salinity waters of the Chesapeake and Coastal Bays. *Microcystis* was first reported in the Potomac River below Washington D.C. during the 1960's when massive blooms were observed. During the 1970's and early 1980's, this area of the Potomac and Upper Bay experienced varying degrees of similar blooms. Then during August and September 2000, larger than normal blooms were reported in the Potomac River. Maryland Department of Natural Resources sampled the river to discover concentrations > 2 million cells milliliter⁻¹, the highest counts observed during any monitoring in the state. Again, the blooms were reported in 2004 caused by *Microcystis aeruginosa*.

The tidal fresh James River contains a diverse and abundant phytoplankton flora. Of over 270 taxa identified in this region (Marshall and Burchardt 1998), *Microcystis aeruginosa* is one of

the 36 species of blue-greens reported. While present in Virginia's tidal waters, blooms have not been observed at the scale of those reported in the Potomac and other upper Bay tributaries. The objectives of this study were to: 1) present the status and trends of this species in the tidal fresh James River, 2) compare abundance of this blue-green in Virginia's three tidal fresh estuaries, and 3) discuss potential controlling factors regulating abundance of this bloom producer in the lower Bay tidal fresh waters.

METHODS

Phytoplankton and water quality data were collected from the Virginia Chesapeake Bay Tributary Monitoring Program from 1984 to 2002. Phytoplankton composition and abundance were determined from monthly composite samples usually obtained from depth-integrated samples above the pycnocline or upper layer. Samples were taken at Station TF5.5 in James River as part of the Chesapeake Bay Plankton Monitoring Program. For this study, two seasons were examined: spring (March, April, May) and summer (July, August, September). June was excluded from the analysis since it represented a transitional period for phytoplankton assemblages. Nutrient samples were collected at various depths from surface to bottom and selected to match the corresponding phytoplankton data by date. Refer to Marshall and Alden (1990), Marshall (1994) and Buchanan *et al.* (2004) for detailed descriptions of these two monitoring programs. Nitrogen and phosphorus limitation was determined using the "Redfield ratio" of 7.2:1 on a mass basis for dissolved inorganic nitrogen (DIN) and phosphorus (PO₄) (Thomann *et al.* 1994).

RESULTS

Cyanobacteria dominate the summer surface waters in the tidal fresh James (Table 1). These algae represent about 72% of the total abundance whereas diatom abundance, a more desirable algae was only 15%. The total biomass for this undesirable species was around ten percent (10.6%), comparable to that found in the Rappahannock, but much higher than observed in York River. Based on this continuous data set, there is an upward trend in Cyanobacteria during the period from 1985 through 2002 (Figure 1).

Table 1. Comparison of phytoplankton groups in Virginia's three tidal regions during summer, presented as a % of total abundance.

Tributary (tidal fresh)	Major phytoplankton groups as a percentage (%)					
	Abundance					
Summer	Chlorophyte	Chrysophytes	Cryptophytes	Cyanophytes	Diatoms	Others
Rappahannock	7.9%	0.2%	4.5%	59.5%	26.9%	1.0%
York	13.3%	0.0%	8.4%	39.5%	38.0%	0.7%
James	9.3%	0.9%	1.3%	72.3%	15.7%	0.4%
Biomass						
Summer	Chlorophyte	Chrysophytes	Cryptophytes	Cyanophytes	Diatoms	Others
Rappahannock	20.0%	3.2%	4.4%	9.7%	52.8%	9.9%
York	51.8%	0.0%	3.5%	2.3%	38.8%	3.6%
James	26.8%	5.7%	1.1%	10.6%	53.7%	2.1%

Other forms include chrysophytes, cryptophytes, and other less abundant forms.

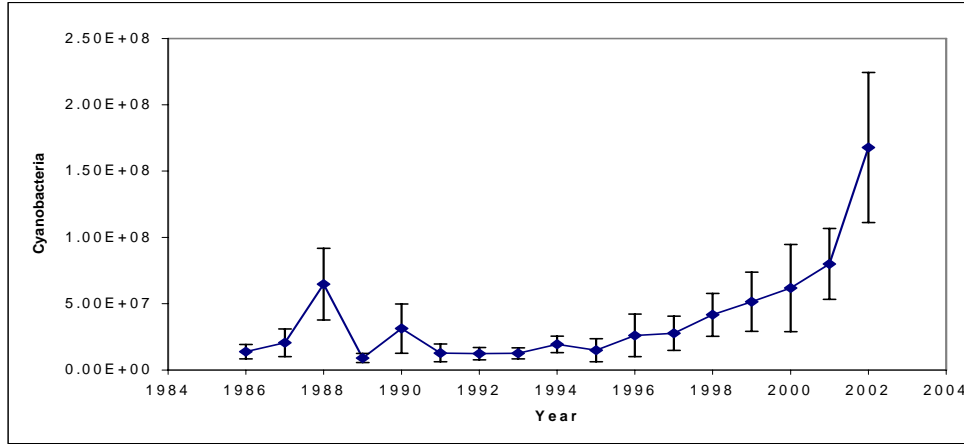


Figure 1. Time-series Cyanobacteria in summer tidal fresh James at station TF5.5.

Microcystis aeruginosa concentrations were over 10×10^6 cells liter⁻¹ during the summer months (Table 2). Average abundance of this bloom producer was significantly higher than levels found in tidal fresh regions of either the Rappahannock or York Rivers.

Table 2. Average abundance of *Microcystis* (cells per liter⁻¹).

Tributary	Abundance	Ratio
Rappahannock	2,446,000	8
York	318,000	1
James	10,947,000	34

Peak abundance of *M. aeruginosa* was reported during 1985 through 1989, but declined significantly through the 1990s (Figure 2). Abundance reappeared in 2000 through 2002. Mean concentrations of biologically available nitrogen as DIN were 0.507 mg liter⁻¹ (+/- 0.349) (N=305) and phosphorus as PO₄ were 0.0589 mg liter⁻¹ (+/- 0.0067) (N=471) during the summer.

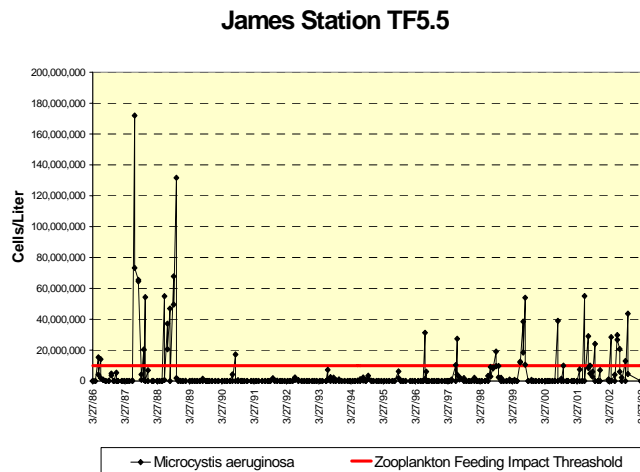


Figure 2. Time-series of *M. aeruginosa* abundance.

DISCUSSION

Many processes are known to regulate the growth and accumulation of algae in aquatic systems (Fisher and Butt 1994). While light and nutrients limit algal growth rates, other processes such as advective transport, grazing, sinking, and cell death help mediate biomass accumulation. The environmental factor most directly related to controlling *M. aeruginosa* seems to be biologically available nutrients in excess of algal growth requirements during the summer months (Paerl 1983). If nutrient concentrations remain in excess during bloom development, then other physical factors including sunlight, low flows contributing to stagnation and salinity may dictate the magnitude of such blooms.

Environmentally controlled experiments by Fisher and Gustafson (2003) demonstrated that concentrations of 0.07 mg DIN liter⁻¹ and 0.007 mg PO₄ liter⁻¹ were limiting thresholds in nutrient bioassays of phytoplankton populations. Based on these threshold levels, the tidal fresh James is highly eutrophic thus satisfying the primary environmental factor controlling this species.

The monitoring data show an increasing trend in *M. aeruginosa* abundance since the lows reported during the 1990s. But concentrations haven't reached peak abundance reported in the 1980s. While flows have been implicated in regulating this bloom producer, advective transport in the tidal fresh James is highest among the Bay's major tributaries. Based on the depth and median residence time for tidal waters, James River was the shallowest (3.3 m) with the fastest residence time (31 days) (Table 3). Therefore, other factors could be regulating this species.

Table 3. Mean depth and freshwater residence time.

Basin	Mean Depth (m)	Median Residence Time (days)
Potomac	7.2	66
Rappahannock	4.8	53
York	4.3	35
James	3.3	31

(Hagy and Boynton 2000)

While the tidal fresh James experienced high N:P ratios throughout most of the 1990s, low N:P ratios were observed at least 50% of the time between 1985 to 1988 and from 2000 to 2002 (Figure 3). This relationship is reported elsewhere (Jacoby *et al.* 2000). Because *Microcystis aeruginosa* can be a contributor to water quality impairments including toxicity to living resources, this particular undesirable species warrants continued surveillance. Aesthetically, it is quite striking as it floats and forms a surface scum. The water may show the appearance of blue-green paint floating or billowing near the surface. Ingestion of waters containing high concentrations of *Microcystis* can cause abdominal stress in humans leading to precautionary beach closings as was done at Colonial Beach (VA) in July of 2004.

In conclusion, elevated nutrient levels continue to support and nourish undesirable, nuisance aquatic plant life in the tidal fresh James River. The dominance of *M. aeruginosa* and the

increasing trend of Cyanobacteria create an imbalance in the indigenous population of phytoplankton in this section of the river as compared to Virginia's other tidal fresh tributaries. Factors regulating *M. aeruginosa* abundance in the tidal fresh James River include high N:P ratios ($> 7.2:1$) during 1985 and 2002 coupled with rapid advective transport. Elevated concentrations of this species were observed when low N:P ($< 7.2:1$) was observed at least fifty percent of the season. However, blooms also appeared to be regulated by moderate to high flows in that section of the river. Low flows tend to reduce mixing and increase light condition, both favoring planktonic blooms. Since 1984, 2002 was the only dry summer on record, but was preceded by a moderate spring flow (Olson 2003). Due to shallow depths coupled with moderate to high flows, waters in this section of the river experience mixing caused by rapid advective transport to higher salinity waters less favorable to these blue-green algae. While the tidal fresh is strongly light limited (Hass and Webb 1998), it may not impact the dynamics of this species since it is a surface dwelling colonial form.

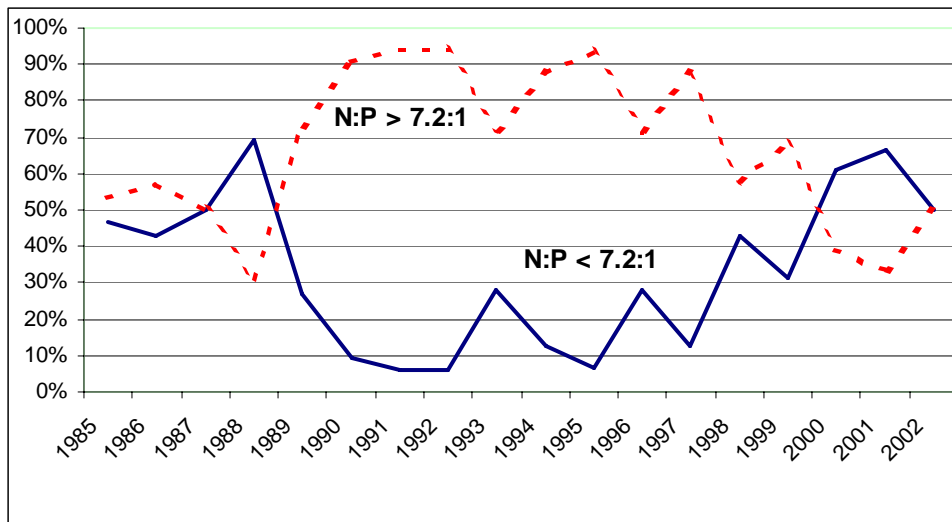


Figure 3. Percent of time nutrient limitation was observed in tidal fresh James River during summer months (1985-2002) based on DIN:PO4 ratio.

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NUTRITIONAL FACTORS PROMOTING HARMFUL ALGAL BLOOMS: THE ADVANTAGES OF MIXOTROPHY

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KEY WORDS: harmful algal blooms, nutrients, mixotrophy, dinoflagellates

ABSTRACT

Mixotrophic species may have a competitive advantage over strictly autotrophic species that can only fix carbon during the day through photosynthesis or are limited by light. Under a variety of conditions heterotrophic uptake of dissolved organic carbon (DOC) provides a significant source of carbon and energy for population growth. The Elizabeth and Lafayette Rivers, sub-tributaries of the lower Chesapeake Bay, experience seasonal dinoflagellate blooms including a variety of mixotrophic dinoflagellates. *Heterocapsa triquetra* blooms occur in the late winter and early spring, *Prorocentrum minimum* blooms in the spring and *Akashiwo sanguinea* blooms in the early summer and *Scrippsiella trochoidea* and *Cochlodinium heterolobatum* bloom later in the summer and fall. We determined that these species have the capacity to take up organic N and C and that both contribute substantially to their growth, often exceeding C uptake from photosynthesis. Results from diel studies suggest that rates of these processes vary on seasonal and diel time scales and as blooms initiate and develop. The ability to assimilate dissolved organic carbon may offer these species a competitive advantage when light is limiting, however, it puts them in direct competition with bacteria. We have found that many bloom-forming phytoplankton can effectively compete with bacteria for organic compounds on relevant timescales and that bacterial productivity may be overestimated when phytoplankton mixotrophs are present. What often inhibits the uptake of dissolved organic compounds is their size. We have found that many of the bloom-forming dinoflagellates are capable of mobilizing organic compounds via peptide hydrolysis and amino acid oxidation and that many of these species can take up compounds larger than previously thought. In general, species that form harmful algal blooms in the lower Chesapeake Bay have flexible metabolisms that allow them to take advantage of a changing nutrient and light environment.

INTRODUCTION

Harmful algal blooms (HABs) appear to be increasing in their geographical extent and frequency and have resulted in severe economic and public health impacts around the world, including the United States and Virginia. Harmful species can elicit adverse effects either directly, by the production of potent toxins that kill or harm fish (or people), or indirectly, through the production of excessive biomass and eutrophication. Effects of excessive production of algal biomass include: shading of submerged vegetation, disruption of food web dynamics and structure, and oxygen depletion as blooms decay. Species that form HABs represent a variety of

taxa that occur naturally in the environment and may “bloom” when environmental conditions are altered to give them a competitive advantage over other co-occurring taxa.

The late spring/early summer plankton communities of the lower Chesapeake Bay (Virginia) contain a number of potentially harmful algal bloom species (*e.g.*, the dinoflagellates *Pfiesteria piscicida*, *Prorocentrum minimum*, *Gyrodinium galatheanum*, *Katodinium rotundatum*, *Cochlodinium heterolobatum*, and *Gymnodinium* spp.) that can reach bloom densities (Marshall 1995). Increases in the occurrence of HABs, including dinoflagellate species in the lower Chesapeake Bay, have been attributed to changes in water quality in a variety of locations; specifically, the enrichment of estuarine waters with dissolved organic material (DOM) relative to inorganic nutrients. Elevated DOM concentrations can result from direct inputs of organic wastes, such as runoff from animal farms or human waste from septic fields and/or inefficient sewage treatment, or indirectly, as a result of nutrient recycling. However, with the exception of *Pfiesteria* research programs, there have been few studies undertaken to understand the processes and interactions whereby water quality parameters influence algal abundance and the occurrence of harmful algal blooms in Virginia waters.

Increases in the occurrence of harmful algal blooms (HABs) including *Aureococcus anophagefferens*, *Pfiesteria piscicida*, *Prorocentrum minimum*, and *Gyrodinium galatheanum*, have been attributed to changes in water quality in a variety of locations; specifically, the enrichment of estuarine waters with dissolved organic material (DOM) relative to inorganic nutrients (Paerl 1988, Lewitus *et al.* 1999, Glibert *et al.* 2001). Many of the organisms responsible for these blooms can use dissolved organic nitrogen (DON) sources to meet at least a portion of their nitrogen (N) demand for growth (Berg *et al.* 1997, Lewitus *et al.* 1999, Mulholland *et al.* 2002). It is thought that this capability offers a competitive advantage over species that can only use inorganic nitrogenous compounds for growth. Consequently, increases in DON-inputs are thought to promote blooms of undesirable species, especially in coastal areas where flushing rates are low and there are seed stocks of organisms capable of utilizing organic compounds for growth.

Dissolved organic material also contains carbon (C). Certain organisms are mixotrophic, *i.e.*, has the capacity to supplement photosynthetic carbon fixation by feeding on other organisms or taking up dissolved organic C compounds (*e.g.*, Paerl 1988, Stoecker *et al.* 1997). Organisms capable of mixotrophy may have a competitive advantage when inorganic nutrients are low or when organic nutrients are present in high concentrations. There is evidence that a number of potentially harmful algal species are able to use both N and C from dissolved free amino acids (Mulholland *et al.* 2002, 2003). The capacity to grow using inorganic nutrients and light energy (*e.g.*, as an autotroph) during the day AND to grow using organic substrates for both nutrients and energy (*e.g.*, as a heterotroph) would give such mixotrophs a competitive advantage by enabling them to: (1) access a larger nutrient pool over a longer time period (*e.g.*, the entire 24-hour diurnal cycle); and (2) grow at high densities when other autotrophs are liable to self-shade.

The late spring/early summer plankton communities of the Chesapeake Bay and the Virginia coastal waterways contain a number of potentially harmful algal bloom species (*e.g.*, *Pfiesteria piscicida*, *Prorocentrum minimum*, *Gyrodinium galatheanum*, *Katodinium rotundatum*, *Cochlodinium heterolobatum*, *Gymnodinium* spp., and *Aureococcus anophagefferens*) that can

reach bloom densities and result in ecological and economic impacts. Most of these species are dinoflagellates and are known to be mixotrophic to some extent. Mixotrophy may play a key role in the ecological success of bloom species in particular environments.

To determine what nutritional factors promote the growth of bloom-forming species in the lower Chesapeake Bay, we examined the relative importance of autotrophic growth and DOM uptake to the nutrition of bloom-forming species that occur in this region.

METHODS

Using highly enriched (96-99%) stable isotopes as tracers and fluorescently labeled substrates, we traced both C and N uptake and examined pathways of organic material mobilization by different size-fractions in natural populations and by individual species and groups of species in cultures. Field sites adjacent to Old Dominion University's campus in the Lafayette and Elizabeth Rivers (near their confluence with the James River and the Chesapeake Bay) were sampled on diel to interannual timescales to allow us to make specific comparisons between the seasonal cycle of nutrient availability and the daily metabolic cycle of photosynthetic organisms.

Nutrient concentrations were measured using standard colorimetric analyses either manually (Parsons *et al.* 1984) or using an Astoria Pacific automated nutrient analyzer. Dissolved free amino acid (DFAA) concentrations were measured by high performance liquid chromatography (Lindroth and Mopper 1979). Stable isotopes of N and C were used to trace the uptake of inorganic and organic N and C (ammonium (NH₄⁺), nitrate (NO₃⁻), urea, amino acids, glucose (glu) and bicarbonate) (Mulholland *et al.* 2002). Extracellular enzyme activity was measured using fluorescently-labeled organic compounds (Mulholland *et al.* 1998, 2002). Isotope samples were analyzed on a Europa 20/20 isotope ratio mass spectrometer equipped with an elemental analyzer to measure the mass of carbon and nitrogen in particulate samples.

RESULTS

Contrary to expectations, nitrate was most often the dominant form of nitrogen (N) in the tributaries during most of the year, including during blooms of the dinoflagellates, *Heterocapsa triquetra*, *Prorocentrum minimum*, *Akashiwo sanguinea* and *Cochlodinium* sp. (Table 1).

Table 1. Seasonal distribution of nutrients at bloom sites in 2003.

Month	NH ₄ ⁺ (μM)	NO ₃ ⁻ (μM)	Urea (μM)	DFAA (μM)	Chl <i>a</i> (μg l ⁻¹)	Cells (cells ml ⁻¹)	Species
Feb	1.95	0.2	1.4	0.17	17.8	2000	<i>H. triquetra</i>
April	0.03 – 2.9	3.7 – 4.8	0.9 – 1.8	0.17-0.27	18 – 191	25,000- 238,000	<i>P. minimum</i>
June	2.6 – 5.7	3.0 – 5.3	1.0	0.21-1.01	16 – 110	10,000	<i>A. sanguinea</i>
Sept	1.7	6.1	1.7	0.29	17.9	10,000	<i>Cochlodinium</i> sp.

Despite the nutrient distribution, blooms became more heterotrophic as the seasons progressed. N uptake became increasingly dominated by organic N compounds (urea and DFAA) (Figure 1).

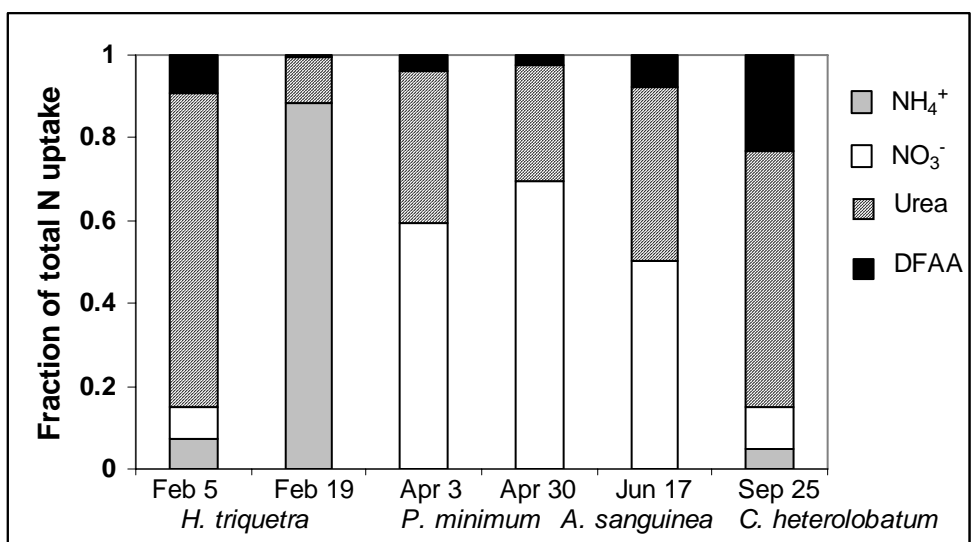


Figure 1. Fraction of total measured N uptake represented by various N pools.

C uptake became increasingly dominated by organic C compounds both seasonally and as blooms progressed (Figure 2).

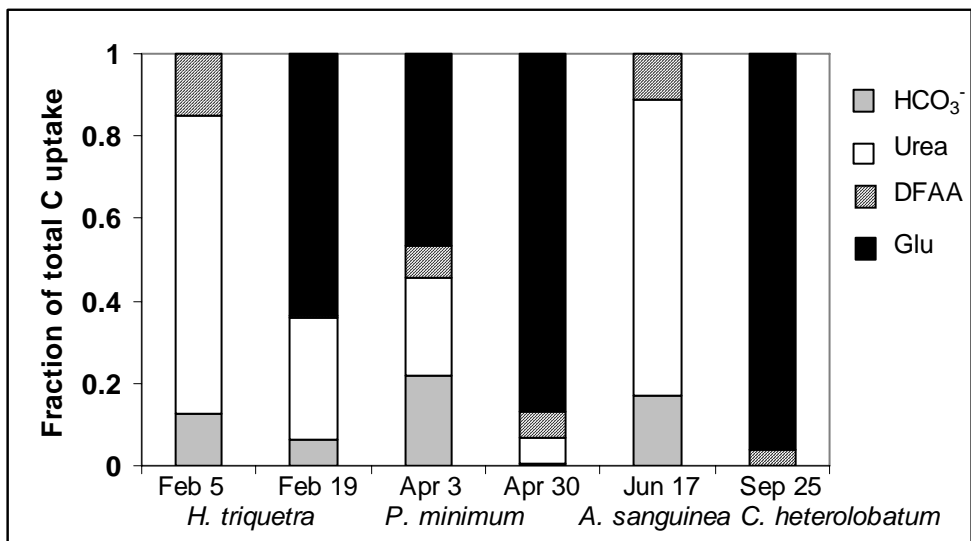


Figure 2. Fraction of total measured C uptake represented by various C pools.

When we examined the C and N nutrition of *P. minimum* on diel timescales, the abundance of cells and chlorophyll concentrations varied dramatically over the diurnal cycle (Table 2) and nutrient concentrations also exhibited diurnal differences.

Table 2. Ambient nutrient concentrations and cell abundance over a diel cycle spanning from April 30-May 1, 2003 during a *P. minimum* bloom (> 99% of all species present).

Time	Temp °C	NH ₄ ⁺ (μM)	NO ₃ ⁻ (μM)	Urea (μM)	DFAA (μM)	Chl <i>a</i> (μg l ⁻¹)	Cells (cells ml ⁻¹)
1100	19	0.77 (0.11)	0.88 (0.17)	0.24 (0.03)	0.16 (0.01)	23.5	11,880
1600	22	0.50 (0.29)	0.65 (0.16)	0.32 (0.02)	0.23 (0.02)	192	238,000
2000	20	0.62	0.93 (0.17)	0.79 (0.02)	0.14 (0.01)	11.5	7,100
2400	19.8	1.52 (0.35)	0.66 (0.04)	0.56 (0.01)	0.24 (0.01)	4.8	682
400	19.8	1.07 (0.43)	0.04 (0.02)	0.70 (0.04)	0.34 (0.03)	24.5	9,240
800	19	1.05 (0.37)	0.12 (0.02)	0.96 (0.14)	0.37 (0.01)	16.1	5,400

Uptake of C and N compounds also varied over the diurnal cycle (Figure 3).

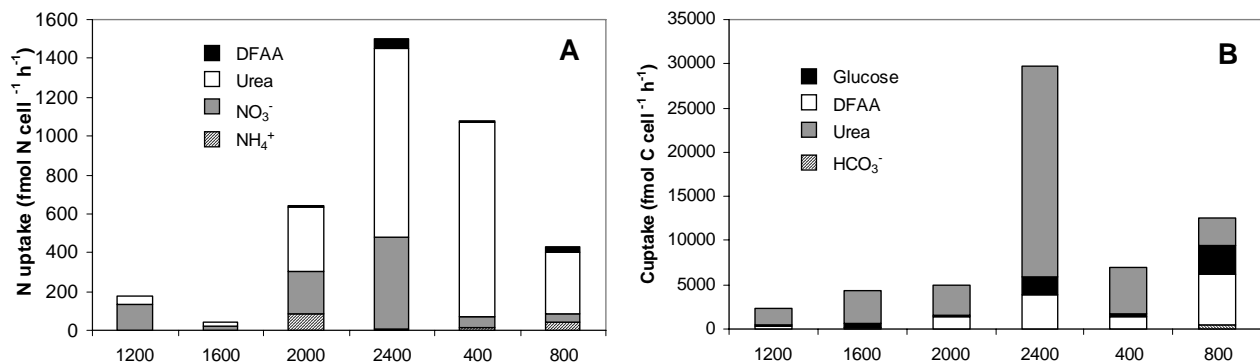


Figure 3. Uptake of N (A) and C (B) over a diel cycle during a bloom of *P. minimum* during 2003. Results are normalized on a per cell basis to account for varying cell density (Table 2). Total daily C:N uptake was 5.8 although hourly C:N uptake ratios ranged from 1.9 to 23.6.

When we examined amino uptake by phytoplankton relative to bacteria, we found that phytoplankton were good competitors with bacteria for amino acids such as leucine, a compound often used to estimate bacterial growth in natural systems. For example, the *P. minimum* size fraction was responsible for the bulk of leucine uptake over the diel cycle (Figure 4) suggesting they can readily compete with bacteria for amino acids traditionally used to estimate bacterial productivity.

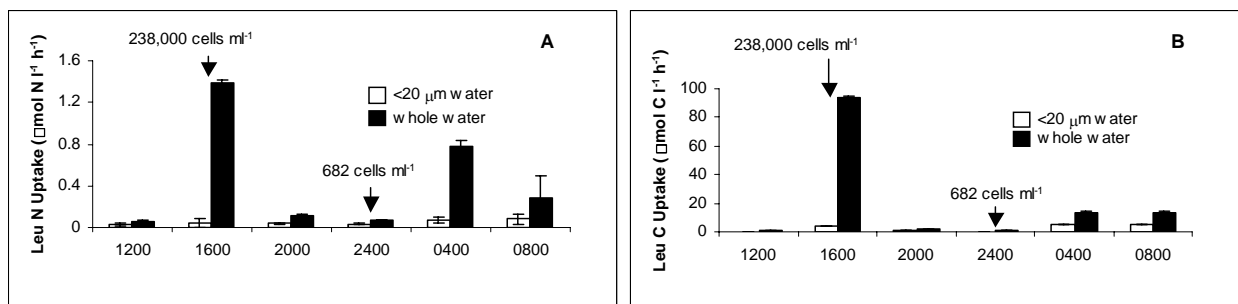


Figure 4. N uptake (A) and C uptake (B) from leucine during a *P. minimum* bloom sampled over a diurnal cycle.

DISCUSSION

Uptake of dissolved organic N (DON) has been explored in recent years as a contributing factor in harmful algal blooms (Glibert *et al.* 2001). Although many of the species that form these blooms are known to be mixotrophic (*e.g.*, can acquire carbon both autotrophically and heterotrophically), the uptake of associated C has not been characterized. One of the reasons for this is that phytoplankton are thought to be poor competitors with bacteria, traditionally the primary consumers of DOM in nature. We demonstrate that, year-round, dinoflagellate blooms are fueled by organic N and C and that many of these species are excellent competitors with bacteria for amino acids. In addition, many of these dinoflagellates are capable of hydrolyzing peptides (Mulholland *et al.* 2002 and 2003, Stoecker and Gustafson 2003, Mulholland and Watson unpublished data) and oxidizing amino acids (Palenik and Morel 1990, Mulholland *et al.* 1998). This may allow organisms to mobilize organic compounds and generate smaller, utilizable compounds such as amino acids and NH_4^+ .

These studies demonstrate that many bloom-forming organisms have versatile metabolisms that contribute to their success in nature. It is unclear whether this metabolic flexibility offers them competitive or even evolutionary advantages over species that are restricted to a narrower range of nutritional choices (*e.g.*, strict photoautotrophy or heterotrophy). In addition, these data argue that it is insufficient to draw general conclusions regarding nutritional physiology based on *ad hoc* studies of N and C uptake made on inappropriate time scales.

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INTERANNUAL DIFFERENCES IN BROWN TIDE (*AUREOCOCCUS ANOPHAGEFFERENS*) BLOOM NUTRITIONAL PHYSIOLOGY.

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KEY WORDS: brown tides, *Aureococcus anophagefferens*, nutrients, harmful algal blooms

ABSTRACT

Blooms of *Aureococcus anophagefferens* have been reported in coastal bays along the East Coast for two decades. In 2003, we recorded the first bloom in Virginia's coastal waters. During this bloom, cell densities reached 0.5 million cells ml⁻¹. Blooms appear constrained to shallow bays that have low flushing rates, little freshwater input and high salinities. Brown tides are also associated with low dissolved inorganic N concentrations and occur subsequent to nutrient drawdown by the preceding algal communities. *A. anophagefferens* can use a variety of organic compounds to meet N demands for growth, but it is unclear whether C from organic compounds subsidizes photosynthetic C uptake, especially during dense blooms when cells self-shade. In this study, N and C dynamics during brown tide blooms in Chincoteague Bay were examined to determine nutritional causes promoting blooms. During 2002, there was a bloom of 1.2 million cells ml⁻¹ in Maryland. In 2003, blooms were less dense but occurred in Maryland and Virginia waters. During 2002, the mid-Atlantic coast experienced drought conditions while 2003 was the wettest year on record. Dissolved inorganic N concentrations were more depleted and dissolved organic C and N concentrations were higher during the 2002 bloom than during the 2003 blooms. Urea was the dominant source of N for blooms during both years, but the source of C fueling blooms likely varied, as indicated by the very different isotopic signatures of particulate material between years. In 2002, more C came from uptake of organic C compounds while in 2003, more of the C demand was met by photosynthesis. There remains a deficit in the cellular C budget suggesting that cells are using compounds not identified. We conclude that *A. anophagefferens* can exploit a variety of nutrients and that nutrient inputs, biotic interactions and the dominant recycling pathways determine which compounds are available.

INTRODUCTION

Harmful algal blooms (HABs) appear to be increasing in their geographical extent and frequency and have resulted in severe economic and public health impacts around the world, including the United States and Virginia. *Aureococcus anophagefferens* occurs in the inland waterways from at least Massachusetts south to Florida (Popels *et al.* 2003) and blooms of this species have been

occurring in estuaries of the mid-Atlantic region for nearly two decades (Casper *et al.* 1990, Bricelj and Lonsdale 1997). Since their first appearance in NY and RI in the mid-1980's, "brown tide" blooms have been observed in NJ, DE, MD and now VA (Mulholland *et al.* in press). During blooms, cell densities can reach over 10^6 cells ml^{-1} (Casper *et al.* 1990, Mulholland *et al.* 2002).

A variety of physical, chemical and biological factors contribute to the formation of harmful algal blooms, including brown tides, in coastal waters (ECO HAB 1995, GEO HAB 2001). While physical factors (*e.g.*, stratification, upwelling, water residence time) appear to pose a first order constraint on the accumulation of phytoplankton biomass in many aquatic systems, nutrient enrichment of estuarine waters has been linked to increased occurrences of harmful algal blooms (HABs) including, *Aureococcus anophagefferens*, in a variety of locations (Paerl 1988, Berg *et al.* 1997, Glibert *et al.* 2001). The coastal bays and waterways of VA receive agricultural nutrient inputs but are also receiving increases in urban inputs, particularly in the north, leading to a number of these waters having been listed as "impaired" (VA DEQ 2004).

In addition to being able to use dissolved inorganic N (DIN) sources, *A. anophagefferens* can use dissolved organic N (DON) to grow. DOM may provide phytoplankton with more than just N for growth. Certain mixotrophic organisms, *i.e.*, osmotrophs, may have the capacity to supplement photosynthetically derived carbon (C) uptake with organic C uptake and this may provide them with a competitive advantage over other phytoplankton in organically-enriched and/or light-limited systems. There is evidence that a number of potentially harmful algal species, including *A. anophagefferens*, are able to use both N and C from DOM compounds (Dzurica *et al.* 1989, Mulholland *et al.* 2002 and in press, Watson *et al.* unpublished data). Organisms capable of mixotrophy may have a competitive advantage when inorganic nutrients are depleted or when organic nutrients are present in high concentrations. The capacity to grow using inorganic nutrients and light energy (as an autotroph) during the day and to grow using organic substrates for both nutrients and energy (as a heterotroph) during both the day and night would give such mixotrophs a competitive advantage by enabling them to: (1) access a larger nutrient pool over a longer time period (*e.g.*, the entire 24-hour diurnal cycle); and (2) grow at high densities when other autotrophs are liable to self-shade. The disadvantage of this type of mixotrophy is that phytoplankton become competitors with bacteria.

Elevated DOM concentrations can result from direct inputs or indirectly, as a result of nutrient recycling. For example, increases in nitrate (NO_3^-) inputs due to agricultural runoff and/or groundwater can stimulate micro- or macroalgal production. Elevated DOM concentrations can occur during and subsequent to these blooms as a result of degradation of plant biomass, grazing or through direct release of assimilated N. Reducing total nutrient inputs and/or removing or safely sequestering nutrients once they have entered a sensitive aquatic system may be a key to avoiding blooms of undesirable or harmful algae. The coastal bays of VA experience strong seasonal shifts in runoff, groundwater inputs, macro- and microalgae production, and nutrient and DOM concentrations. Many blooms occur subsequent to macroalgal blooms in early spring and it has been hypothesized that regeneration of DOM from macroalgae could fuel brown tides (Bob Nuzzi, Cathy Wazniak pers. comm.). High concentrations of brown tide cells were also found near Fisherman's Island during a 2002 survey that included Virginia's coastal bays (Popels *et al.* 2003). This area has extensive benthic mats and so may be a site of extensive

benthic N₂ fixation (Noffke, pers. comm.). Regeneration of DOM from these benthic populations may play an important role in fueling water column algal blooms during May and June and so it is critical that we understand the dominant recycling pathways.

In this study, we examined N and C dynamics during brown tide blooms in Chincoteague Bay to determine nutritional causes promoting blooms.

METHODS

Using highly enriched (96-99%) stable isotopes as tracers and fluorescently labeled substrates, we traced both C and N uptake and examined pathways of organic material mobilization in natural populations collected from Chincoteague Bay, Maryland and Virginia during 2002 and 2003. We compare uptake of inorganic and organic C and N compounds over diel to interannual timescales in the context of the dominant phytoplankton species.

Nutrient concentrations were measured using standard colorometric analyses either manually (Parsons *et al.* 1984) or using an Astoria Pacific automated nutrient analyzer. Dissolved free amino acid concentrations were measured by high performance liquid chromatography (Lindroth and Mopper 1979). Stable isotopes of N and C were used to trace the uptake of inorganic and organic N and C (ammonium (NH₄⁺), nitrate (NO₃⁻), urea, amino acids, glucose and bicarbonate) (Mulholland *et al.* 2002). Extracellular enzyme activity was measured using fluorescently labeled organic compounds (Mulholland *et al.* 1998, 2002). Isotope samples were analyzed on a Europa 20/20 isotope ratio mass spectrometer equipped with an elemental analyzer to measure the mass of carbon and nitrogen in particulate samples.

RESULTS

We found that nutrient and DOM quantity and quality varied more interannually than it did between sites that did and did not have blooms during the same year (Table 1). During the 2003 brown tide blooms, inorganic N was never depleted and, while urea was the primary form of N taken up, there did not appear to be significant uptake of urea or other organic carbon, even at night (Figure 1). This was in contrast to results from a Long Island, NY study where amino acids and NH₄⁺ were the primary sources of N during a brown tide bloom (Mulholland *et al.* 2002 & in press).

Table 1. Salinity, nutrients and chlorophyll concentrations in Chincoteague Bay at Public Landing, MD (PL) and Greenbackville, VA (GB) in 2002 and 2003. There was no bloom at GB in 2002. There were blooms of *A. anophagefferens* at GB in 2002 and at PL in both years.

Site	Salinity	NO ₃ ⁻ (μM)	NH ₄ ⁺ (μM)	Urea (μM)	Chl <i>a</i> (μg l ⁻¹)
PL 2002	32.6	0	0	4.24	19.5
PL 2003	25.9	0.25	1.04	0.60	10.6
GB 2002	33.7	0.30	1.17	2.72	10.0
GB 2003	25.7	0.26	0.71	1.04	16.1

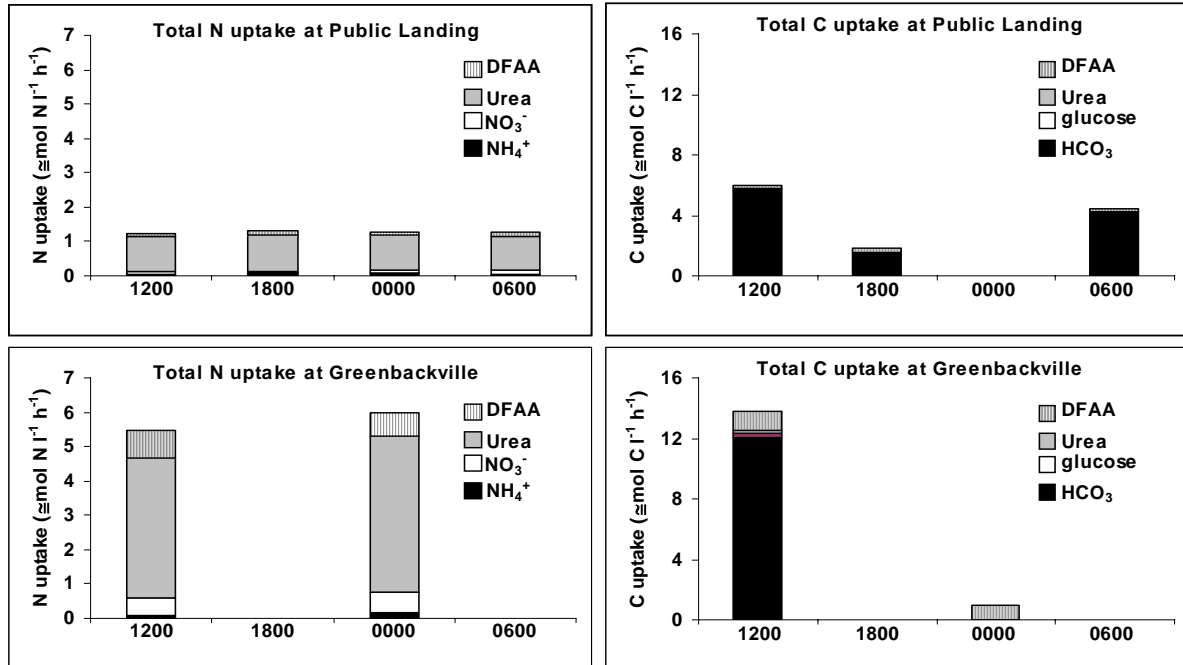


Figure 1. Diel N and C uptake ($\mu\text{mol N}$ or $\text{C l}^{-1} \text{h}^{-1}$) during blooms in Chincoteague Bay during 2003. *A. anophagefferens* concentrations reached 4.91×10^5 (GB) and 3.58×10^5 cells ml^{-1} (PL).

Urea was the primary source of N and bicarbonate the primary source of C during both years although amino acids could supply both N and C to cells. While photosynthetic C uptake was the dominant source of C measured during both years, photosynthesis was insufficient to meet cellular C demand and so other C sources that we were not measuring must have been exploited. In 2003, we observed that, where the brown tide bloom was well developed (Greenbackville), there was more total N uptake (mainly organic N) and more organic C uptake than at Public Landing, where the bloom was just initiating at the time of sampling (Figure 1). This suggested to us that organic carbon uptake may be related to the physiological state of cells. Initially, when blooms are beginning, *A. anophagefferens* may be relying solely on photosynthesis. When densities increase and nutrients decrease organic carbon uptake may become more important. During 2003, cell densities were not as high as those observed during a 2002 bloom (1.2×10^6 cells ml^{-1}) and so self-shading and consequent uptake of organic C may have been less important during the 2003 blooms.

These findings lead us to believe that *A. anophagefferens* has a flexible metabolism and can exploit a variety of nutritional sources that may vary on interannual timescales. The adaptability of *A. anophagefferens* to the use of different nutrient substrates is consistent with previous hypotheses regarding their competitive advantage during conditions of inorganic nutrient limitation and/or light limitation (e.g., Gobler and Sañudo-Wilhelmy, 2001a; 2001b).

The primary sources of N and C available to phytoplankton are likely determined by exogenous inputs from runoff and the dominant *in situ* producers and recycling pathways affecting the degradation of that material. During the previous two-year study, the amount of rainfall varied dramatically between years and that may have played an important role in the relative dominance of exogenous versus regenerated nutrient sources between study years.

Finally, purely by accident, we discovered that the natural abundance of ^{13}C in particulate and dissolved material varied significantly between years during our previous project suggesting the source of C varied between years (Table 2). Blooms occurred during both years suggesting that there are multiple C cycling pathways or multiple C sources that can satisfy the demands of this particular bloom-forming organism.

Table 2. Measured $\delta^{13}\text{C}$ signatures of dissolved and particulate pools.

Size fraction	Bloom	Non-bloom	Post-bloom
<i>2002:</i>			
POM	-24.5	-21.8	-20.2
HMW-DOM	-21.0	-20.2	-22.2
<i>2003:</i>			
POM	-37.5		
HMW-DOM	-23.4/-26.8		-20.8

The observed interannual differences argue for a longer-term study in which potential sources of N and C, both autochthonous and allochthonous, are identified and described.

DISCUSSION

Of the largely uncharacterized bulk DON pool, urea and amino acids are probably the most widely studied compounds because they are relatively easy to quantify and have simple chemical structures (Antia *et al.* 1991, Bronk 2002). *A. anophagefferens* has a high affinity for urea (Lomas *et al.* 1996) as well as a high potential uptake capacity for this form of N (Berg *et al.* 1997). Of the N sources examined, urea was the dominant source of N during a brown tide bloom in Shinnecock Bay, NY (Berg *et al.* 1997) as well as during the blooms we report on in Chincoteague Bay while NH_4^+ and DFAA were the dominant N sources during blooms in Quantuck Bay, NY (Mulholland *et al.* 2002). Dzurica *et al.* (1989) found that urea uptake was dependant on the nutritional history of cultured cells. Similarly, peptide hydrolysis and amino acid oxidation were less important in Chincoteague Bay than in Quantuck Bay and this may reflect the nutrient history of cells (*e.g.*, Mulholland *et al.* 2002 and in press). Treatment incubations support this idea since additions of inorganic and organic nutrients caused decreases in peptide hydrolysis relative to controls (Mulholland *et al.* in press).

Despite a growing body of work aimed at understanding the role of organic matter in the nutrition of *A. anophagefferens*, it appears that there is no smoking gun implicating any one compound in the formation of brown tide blooms. Urea, amino acids and NH_4^+ can all contribute to the N nutrition of *A. anophagefferens* and it appears that a variety of others can as well.

When uptake from all of the labeled C compounds tested in this study (bicarbonate, urea, glucose, and DFAA) were summed, there was still a significant deficit in C uptake relative to N uptake. Disparities between C and N uptake can arise on diel or timescales other than those examined. In addition, other organic carbon species not yet “modeled” by the tracer molecules may be responsible for subsidizing brown tide C nutrition or, alternatively, uptake of organic C

may be sporadic throughout the day. In nature, the types of compounds available may be site specific, depending on direct inputs, biotic interactions and the dominant recycling pathways. The adaptability of *A. anophagefferens* to the use of such different substrates has been hypothesized to give the species a competitive advantage during conditions of inorganic nutrient limitation and/or light limitation (*e.g.*, Gobler and Sañudo-Wilhelmy, 2001a; 2001b).

DOM (including DON) in coastal systems has several allochthonous and autochthonous sources. Allochthonous DOM inputs result from riverine/stream inputs, runoff, and atmospheric deposition to the water surface or watershed. Autochthonous sources of DOM include such water-column sources as phytoplankton exudates/diffusates, the cellular material of phytoplankton or bacteria released by viral lysis or zooplankton grazing, exoenzymes, zooplankton excretion and the dissolution of POM (including zooplankton fecal pellets).

For Chincoteague Bay, the watershed is small compared to river-dominated estuaries resulting in relatively low freshwater delivery (Boynton *et al.* 1996). Therefore, in this system, point sources represent only a small component of the total N inputs (< 0.01%) while diffuse-sources such as surface water and groundwater inputs and atmospheric deposition are dominant. Sources of N in wet deposition include agricultural facilities and fossil-fuel combustion. Livestock operations and fertilizer applications, including animal wastes, are common on the eastern shore of MD and VA. Feeding operations were considered to be the largest contributor of nitrogen loading to the Maryland coastal bays (Boynton *et al.* 1996). Both urea and amino acids have been identified in wet deposition (Russell *et al.* 1998).

In Chincoteague Bay there also appear to be important autochthonous DOM sources. For example, high DOC levels were associated with peaks in chlorophyll production that preceded the onset of brown tide blooms in 2002 (Simjouw *et al.* in press). Agriculturally enriched groundwater appears to be an important source of N in the coastal bays (Anderson *et al.* 2003) and benthic primary producers (macro- and microalgae) and macroalgal production are important sources of labile plant material present in these systems (McGlathery *et al.* 2001). Degradation of this plant material or their affect on DON fluxes into and out of the sediments may also play a part in regulating the quantity and quality of DOM available to phytoplankton.

We conclude that *A. anophagefferens* can exploit a variety of nutrients and that nutrient inputs, biotic interactions and the dominant recycling pathways determine which compounds are available. Because *A. anophagefferens* is able to use a variety of organic compounds, either directly or indirectly, they are able to bloom in physically similar but biotically diverse systems.

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A REVIEW AND ANALYSIS OF *EL NIÑO* RELATED INDICES

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ABSTRACT

In this paper, the correlation between *El Niño* and *La Niña* events and Roanoke annual precipitation is considered. *El Niño* is the cyclical warming of the sea surface in the eastern Pacific (coast of Peru). This above normal sea surface temperature causes increased precipitation in many parts of the world and at the same time pronounced droughts in selected parts of the world. *La Niña* is the unusually cold ocean temperatures in the eastern Pacific and induces climate effects that are opposite to that of *El Niño*. It is observed that in the Roanoke Region, a combined standardized rain and standardized sea surface temperature (SST) anomalies might show the correlation trend better than relating the SST anomaly to the precipitation values directly.

INTRODUCTION

During 2002 Roanoke suffered serious water shortage and its main source of water supply, the Carvins Cove reservoir, experienced extremely low water levels. In 2003, Roanoke received significant amounts of rainfall and the Carvins Cove has remained nearly full for more than a year. Coincidentally, the year 2003 has been declared an *El Niño* year and there is a belief that during *El Niño* years the Roanoke region receives wide spread heavy rains. However, systematic studies in support of this belief are lacking.

El Nino and La Nina Phenomena

The equatorial Pacific region receives some of the highest solar energy on Earth, with the majority of this energy being stored in the ocean as heat. This heat energy is distributed across the ocean surface by the pacific easterly trade winds. These winds are controlled by the atmospheric pressure gradient that exists over the region at a given time. The magnitude of the trade winds plays a key role in how sea-surface temperatures evolve over time. In a normal ocean-atmosphere state, these east-west trade winds cause the warmer surface waters to advect westward, and upwelling of colder water to occur along the eastern pacific coastline. The ocean surface is about 8°C warmer in the west than in the east, and the cool water of below 17°C lies within 50 m of the ocean surface at 110°W longitude of the east Pacific. In *El Niño* years, the trade winds weaken and the 17°C isotherm drops deeper to about 150 m in the east Pacific, indicating a warmer ocean surface. The warmer water results in heavy precipitation in Peru and droughts in Indonesia and Australia. During a *La Niña*, the east Pacific Ocean surface is cooler

than the normal. *La Niña* events sometimes follow the *El Niño* events, but not all. More details on theory and occurrence of *El Niño* and *La Niña* events can be found in (NOAA/PMEL 2004, NOAA/TAO 2004, Trenberth 1991)

***El Niño*/Southern Oscillation (ENSO) Events**

Three phases have been categorized within this ENSO cycle based on the orientation of the atmosphere/ocean interface above the Pacific Ocean:

- Neutral – the equatorial Pacific trade winds, sea-surface temperatures, and sea-level pressure distributions are in their normal state
- *El Niño* – easterly trade winds are weakened due to an increase in sea-level pressure in the western Pacific, causing sea-surface temperatures in the eastern Pacific to be significantly above normal
- *La Niña* – easterly trade winds are strengthened by a rise in sea-level pressure in the eastern Pacific, causing sea-surface temperatures in the eastern Pacific to be considerably colder than normal

ENSO Prediction

There are a variety of methods to assess the current status of the ENSO cycle and also to predict how it will evolve in the near future. This research focuses in on methods that are based on the sea surface temperature (SST) measured at select regions in the eastern equatorial Pacific. The National Oceanic and Atmospheric Administration (NOAA) and the Japan Meteorological Agency (JMA) have developed SST indices as described in the following.

- NOAA SST – Warm and cold episodes are based on the SST anomalies calculated from the measured SST values of the Niño 3.4 region (5°N-5°S, 120°-170°W) called the Oceanic Niño Index (ONI). The anomaly is the measured value minus the 30-year normal corresponding to the period of 1971-2000. If a 3-month moving average of ONI is greater than or equal to 0.5°C, the onset of an ENSO event is predicted. For a full fledged ENSO 5-month moving average exceedance of 0.5°C is adopted. For a *La Niña* event -0.5°C threshold is used (NOAA/CPC 2004).
- JMA SST – This index is a 5-month running mean of spatially averaged SST anomalies over the tropical Pacific: 4°S-4°N, 150°W-90°W. If index values are 0.5°C or greater for 6 consecutive months (including October, November, and December), the ENSO year of October through the following September is categorized as *El Niño*. A *La Niña* event is predicted for index values equal or less than -0.5°C. A Neutral event is adopted for all other values (Legler 1998).

METHODS AND RESULTS

Data

For the period of 1950-2003, monthly precipitation for Roanoke (NOAA/NCDC 2004), the JMA SST index (JMA 2004), and the NOAA monthly SST anomalies (NOAA/CPC Indices 2004) were analyzed. The precipitation values were calculated from the monthly totals of daily precipitation for Roanoke. Sea-surface temperature anomalies are based on observed data from a network of moored buoys that detect the differences in the temperatures of the Pacific, as well as the currents and winds.

Annual Rainfall and SST Anomalies

Figure 1 shows a plot of Roanoke monthly average of rainfall for each year and SST index values as a function of time. At the beginning of this research, the *El Niño* calendar years of 1951, 1953, 1957-58, 1963, 1965-66, 1969, 1972-73, 1976-77, 1982-83, 1986-87, 1991-95, 1997-98, and 2002-03 were used, and these are also shown.

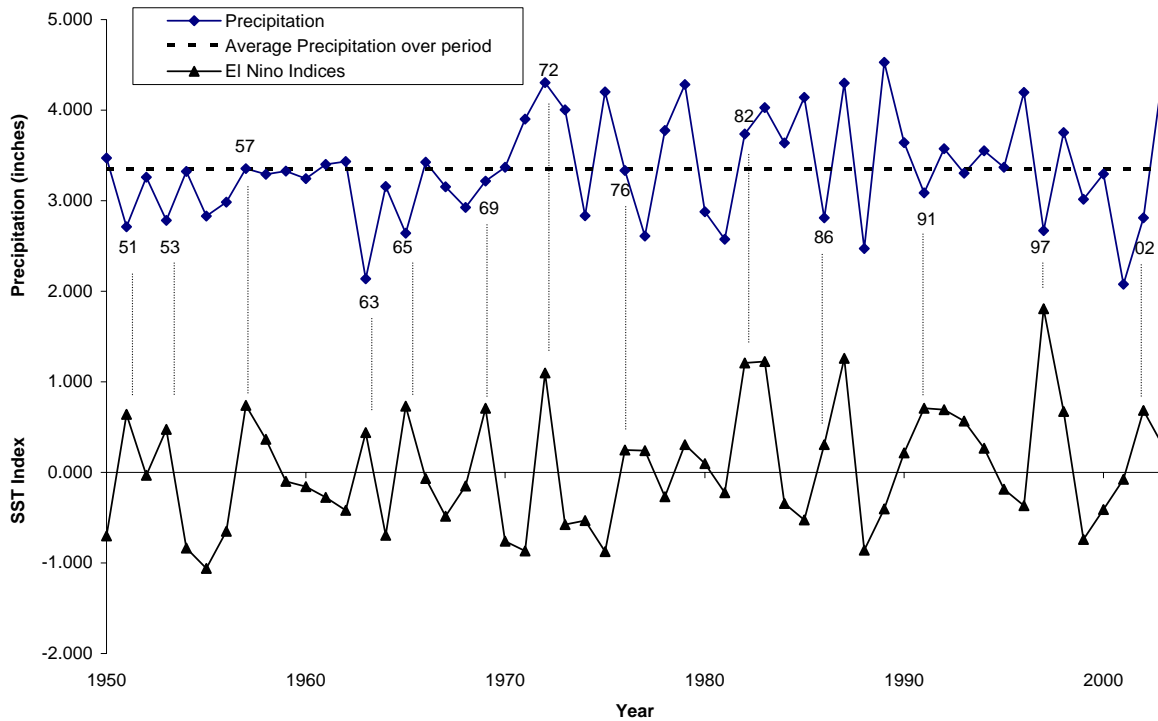


Figure 1. Roanoke monthly average rain vs. average SST index.

It is observed that the rainfall during the year after an *El Niño* event is above average. In Figure 2, the difference between the standardized rain and the standardized JMA SST index is plotted. The standardized variate is computed as: $(\text{variate value} - \text{average}) / \text{standard deviation}$. It can be seen that during *El Niño* years, this difference in standardized variates has a negative value and the year after it has a positive value.

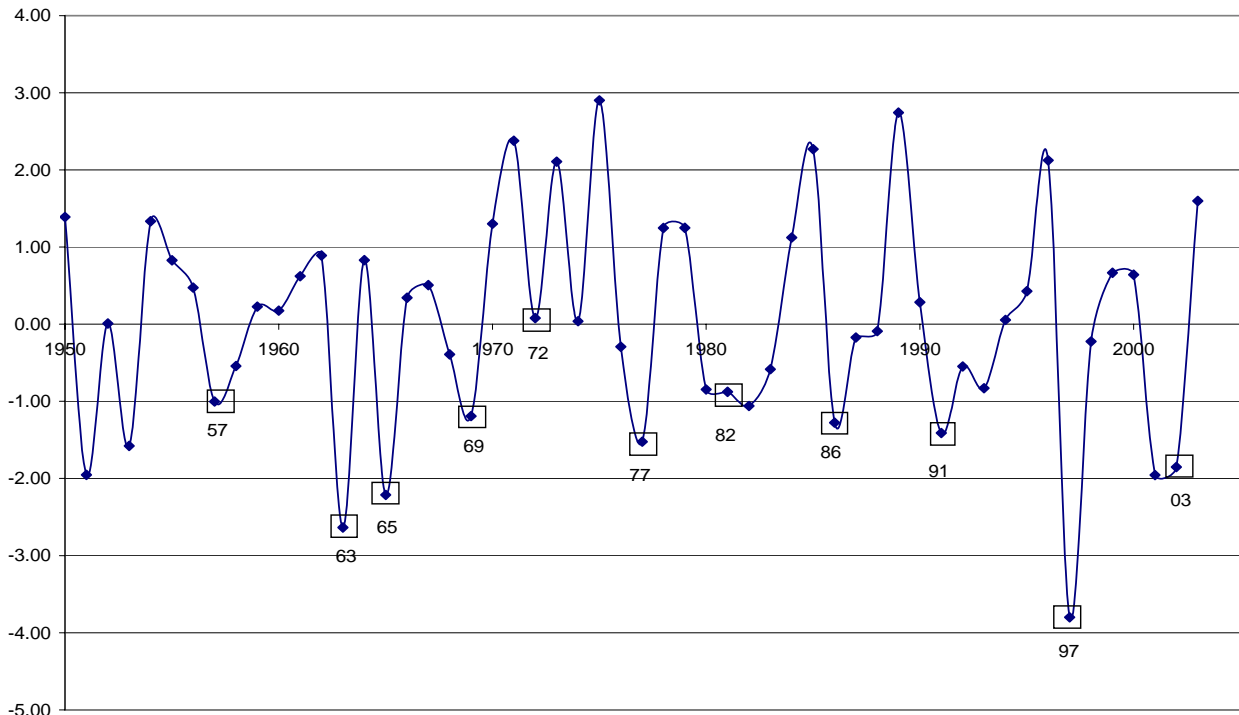


Figure 2. Difference between Roanoke standardized rainfall and standardized SST index.

Monthly Rainfall and SST Anomalies

This analysis used the monthly precipitation and SST data. Preliminary analysis revealed some subtle hints that SST anomalies and precipitation totals can move in the same direction above or below their mean at the same time. Examination of the two time series revealed that SST anomalies evolve in a sinusoidal nature, while Roanoke precipitation values exhibited a random pattern. These different responses may be due to the fact that precipitation totals for a month, and resulting year, can be significantly altered by heavy moisture events, such as hurricanes in the summer and major storms (*e.g.*, Nor'easters) in the winter, which can produce several inches of precipitation in just a few days. These weather events can impact southwest Virginia at any time and can account for a large percentage of the annual total precipitation, no matter what the ENSO phase is for that year.

Utilizing Prescribed Six Phases of *El Niño*

Based on (Wang 2000), there are six phases to *El Niño*: Antecedent [Aug(-1) – Oct(-1)], Onset [Onset Nov(-1) – Jan(0)], Development [Mar(0) – May(0)], Transition [Jul(0) – Sept(0)], Mature [Nov(0) – Jan(+1)], and Decay [Mar(+1) – May(+1)] (Wang, 2002). In the above designations, (-1) denotes the year before an *El Niño* year, (0) is the *El Niño* year, and (+1) is the year after an *El Niño* year. Averages for each phase were generated on the two data sets of SST anomalies and Roanoke precipitation values. Typical results are presented in Figure 3. The results indicate a potential tendency for increased precipitation values in the Roanoke region during the decay periods of SST or termination of the *El Niño* event.

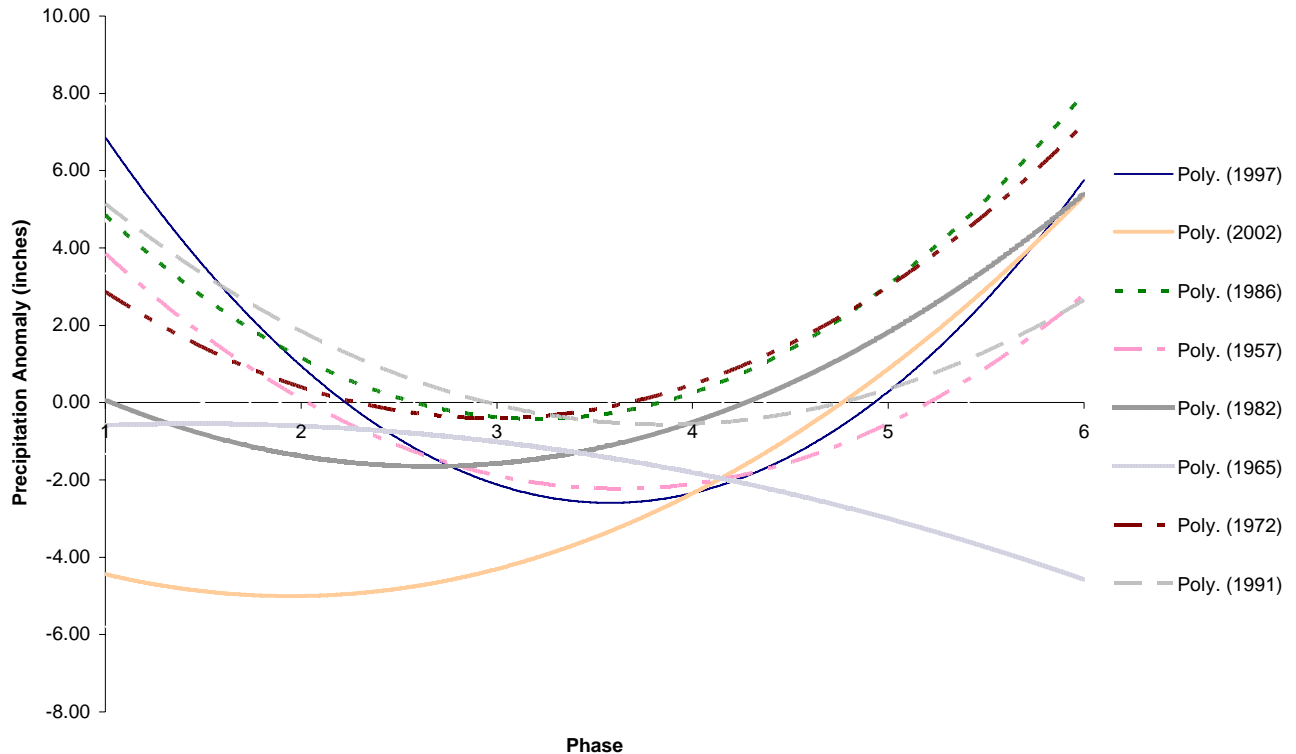


Figure 3. Precipitation trends using the 6 phase method on the 8 strongest *El Niño* events.

Data Analysis on Water Year Basis

Analysis presented thus far was based on calendar year data. It was determined that a review of ENSO prediction techniques should be completed to generate a finalized set of *El Niño*, *La Niña*, and Neutral years to use in further analyses. This review revealed that there are many interpretations as to what years are considered to be *El Niño*, *La Niña*, and Neutral, due to numerous classification methods that have been used (Green *et al.* 1997, Cayan *et al.* 1999, Lee *et al.* 2003, Barton and Ramirez 2004). Since the focus of this study is on water supply from precipitation, it was determined that the water year should be utilized instead of the calendar year. In late September, precipitation events often diminish in frequency, causing stream flows to return back to their base flow. Because of this, hydrologists formed the water year of October 1st – September 31st in order to keep major precipitation and stream flow events separated out from year to year. Calendar years can sometimes have major events that cross the boundary from one year into the next, which can complicate climatological assessments. The water year also fits well with this analysis because the JMA ENSO predictor described above uses the same time frame of October – September to delineate their ENSO years.

Analysis of Precipitation Using the Water Year and JMA SST

JMA SST data were evaluated to identify *El Niño*, *La Niña*, and Neutral years using the JMA standards. The resulting classification corresponds exactly with the years that were identified in (Green *et al.* 1997). Each phase of the ENSO cycle was evaluated separately to gauge precipitation trends that may exist. Monthly precipitation averages were generated and plotted to visualize the trends. Figure 4 shows the plot for each category with respect to the overall mean of monthly precipitation within the respective water years.

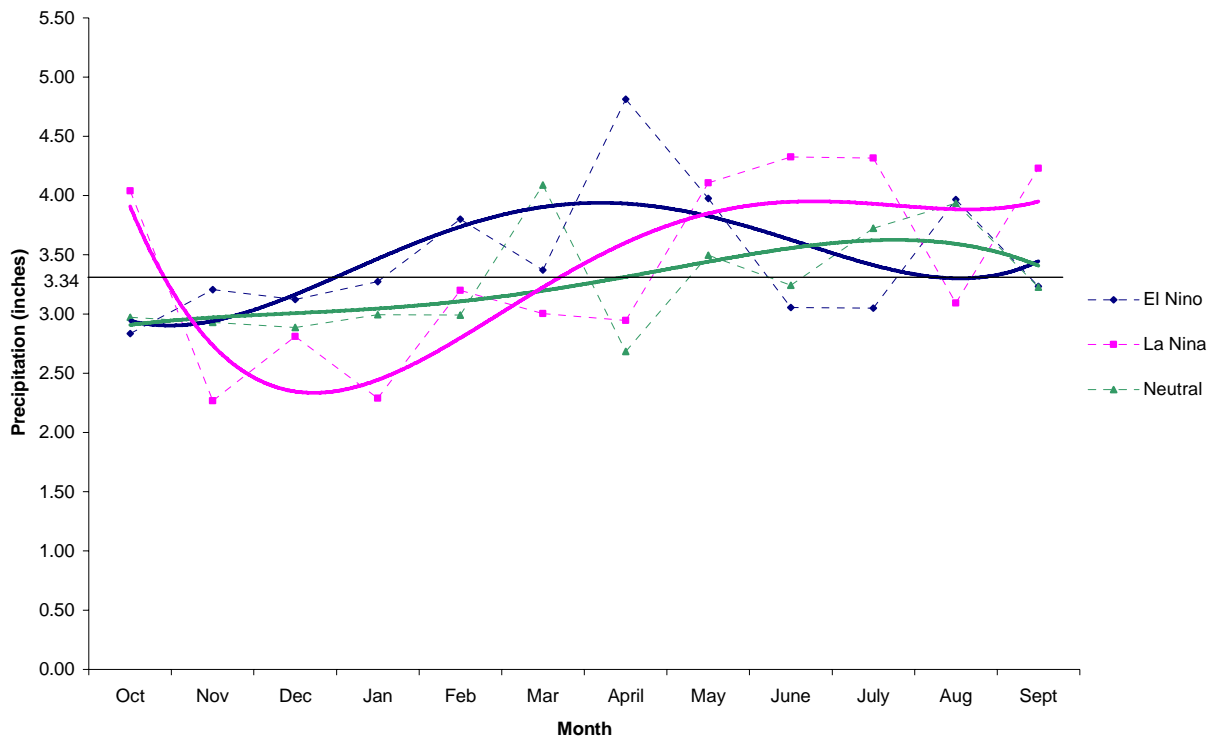


Figure 4. Average monthly Roanoke precipitation based off of the JMA criteria (climatological mean of monthly precipitation is 3.34).

Trend lines reveal that *El Niño* water years may have above average precipitation from February to May, *La Niña* years below average in beginning of water year and above average from May onward, and Neutral years slightly below average through April and then a little above normal until the end of the water year.

Analysis of Precipitation Using the Water Year and NOAA SST

NOAA SST data were assessed using NOAA ENSO prediction standards to see how this criterion classifies ENSO years in relation to JMA and if it reveals any different precipitation characteristics. There was a slight variation in this classification of each year because the NOAA criteria utilizes a 3 month-running mean of SST anomalies, while the JMA criteria uses a 5 month-running mean. This did alter monthly averages for each category, but not enough to significantly shift trends from what was seen in the previous JMA analysis, as seen in Figure 5.

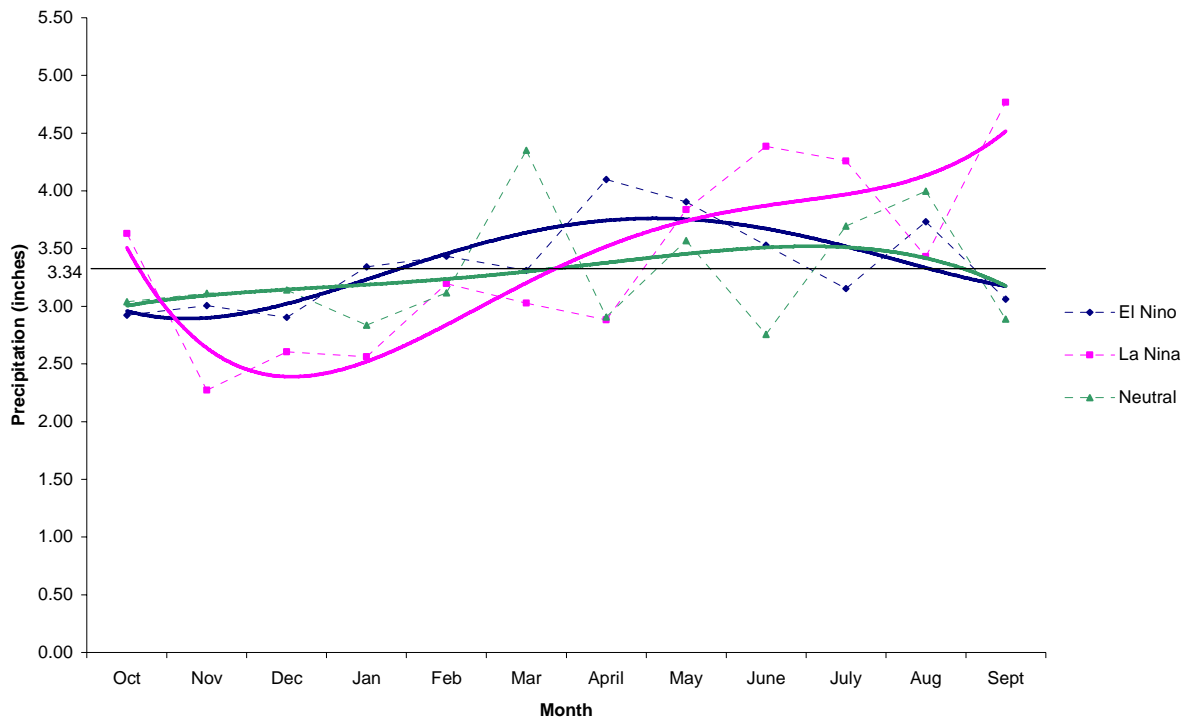


Figure 5. Average monthly Roanoke precipitation based off of the NOAA criteria (climatological mean of monthly precipitation is 3.34).

CONCLUSIONS

These analyses give subtle hints on how rainfall characteristics for Roanoke, Virginia might evolve as the result of an ENSO cycle, but the results are not definitive to reveal any dramatic departures from normal. Southwest Virginia seems to be in a region of the country where the ENSO cycle does not consistently impact precipitation trends because the region is situated in a geographic area that has moisture feed from many sources and contains very complex mountain topography. This combination can generate a wide variety of heavy moisture weather events that can occur during *El Niño*, *La Niña*, and Neutral years. Our preliminary results indicate that the decay phase of an *El Niño* event may lead to increased precipitation for Roanoke and there may be above normal rainfall during the latter months of *La Niña* water years.

ACKNOWLEDGMENT

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THE EFFECTIVENESS OF VOLUNTARY AND MANDATORY RESTRICTIONS ON WATER DEMAND IN VIRGINIA

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KEY WORDS: water, demand, restrictions

ABSTRACT

In response to the 2002 drought, Governor Warner issued Executive Order 33 that required municipal water suppliers to reduce water use through mandatory water use restrictions. Prior to the executive order, many water suppliers had already implemented a variety of voluntary and mandatory emergency policies to reduce water demand. As expanding water supplies becomes more challenging, the drought of 2002 highlighted the importance of demand-side management policies in future water supply planning efforts. However, little systematic information is known about the way Virginia residents respond to different water demand policies such as voluntary and mandatory restrictions.

This project will identify the factors that affect water demand during times of drought, especially at the residential level, with the focus on determining the effects that voluntary and mandatory restrictions have on residential water use. As the final statistical analysis has yet to be completed, this paper will highlight a graphical illustration of the issues involved with modeling water demand and capturing the effects of voluntary and mandatory restrictions. Conclusions of this graphical analysis related to modeling water demand are presented.

BACKGROUND

The 2002 drought was a serious natural event that has had important consequences and changes concerning water planning in Virginia. In most of the state, municipal water supplies were stressed to some degree. In some cases, supplies were severely depleted and emergency measures with either instituted or were being contemplated. In June 2002, 18 waterworks had instituted voluntary restrictions and 4 that had instituted mandatory restrictions. By August, as the drought intensified, 39 waterworks had instituted voluntary restrictions and 20 waterworks had instituted mandatory restrictions (Drought Management Report compiled by the Virginia Department of Environmental Quality, June and August reports). By the end of August, the drought had become so serious that Governor Warner issued Executive Order 33 that imposed significant restrictions on outdoor water use in most of Virginia. Fortunately, normal rainfall returned later in the fall and most of the provisions of the statewide restrictions were lifted by mid-November.

State officials saw the drought as a warning that better procedures for dealing with water supply and demand during times of drought were needed. This concern resulted in the issuance of a number of proposals by the Governor later that year and the State's subsequent Drought Response Plan (still in draft form at this date). Demand side management is expected to be an important part of these proposals and plans. Demand side management (DSM) uses conservation programs to reduce and control water demand in a locality, to match the water supply.

The water shortages created by the 2002 drought and the responses at both the local and state-level to deal with the situation may be a precursor to long-run changes in Virginia water management. Historically, the prevalent method for dealing with water supply in Virginia was supply-side management. Under this system, municipalities essentially took water demand as given and secured sufficient water supplies to meet this demand even under the most unfavorable circumstances.

However, Virginia municipalities face a number of challenges to building new reservoirs, expanding reservoirs, or in securing additional water withdrawal permits from rivers. Many of these challenges are legal in nature and reflect the increasing difficulty in securing new water supply sources in all regions due to environmental and legal constraints such as the Clean Water Act, Endangered Species Act (requirements for in-stream flows), Indian water rights claims (holding water rights for future use), NEPA and EIS requirements (Maddock and Hines 1995, Shabman and Cox 2004). This problem is being echoed throughout the country, particularly the southwest.

This difficulty in expanding water supplies in conjunction with continued population growth will mean that the risk of short-term (temporary) water shortages will likely increase in the future. As a consequence, future short-term demand side management (SDSM) will be an increasingly important part of local water supply plans. The two most common methods to reduce water use during the 2002 drought in Virginia, voluntary and mandatory restrictions, will be the SDSM programs evaluated in this study.

OBJECTIVES

In planning for potential water shortages, localities need to know how water use will respond to different drought management policies. With SDSM, the effectiveness of individual strategies becomes critical, as water demand levels must often be temporarily brought down to target levels in relatively short time periods. Water supply planners need insight into what level of water-use reduction can be expected with various short-term demand reduction strategies.

The objective of this analysis is to estimate the reduction in water use stemming from the voluntary and mandatory restrictions (forms of short-term demand side management programs) implemented during the 2002 drought. The paper will focus on residential water users as they typically consume the largest proportion of water in eastern waterworks. In fulfilling this objective, this analysis will also account for other factors that might influence residential water use such as price, seasonality, climate, and various demographic variables.

ESTIMATING CHANGES IN WATER USE

There appears to be limited empirical evidence of expected water use reductions that can be used by municipalities in Virginia during times of drought and periodic water shortages. Of the previous empirical work on water demand, most studies have typically focused their efforts on other aspects of water demand (price, income) rather than drought management restrictions. Few studies that have estimated the effectiveness of drought management policies and these studies tend to be in the southwest.

Changes in water use are typically estimated by multivariate statistical (regression) approaches. This approach uses large data sets with information on the main factors thought to influence water use. Multiple regression is appealing because of its potential to isolate the multiple factors than influences on residential water use. The main disadvantages are that the data requirements are comparatively high and that some level of technical expertise is required for the estimation.

In general, the approach in this study will use monthly municipal-level water use data broken down by user-type. Residential water demand (gallons per day per connection) will be estimated as a function of a number of variable classes, which will include price, income, other demographic variables, and season/climate variables. Additional information can be added to estimate the effect of SDSM programs on changes in water use.

Residential Water Use = function (SDSM programs, price, income, month, rainfall, temperature, demographic variables)

where SDSM programs = function (restriction type, publicity/information, and enforcement)

DATA

Historical monthly residential water use data was obtained from 22 counties and cities in Virginia. These localities each had varying degrees of water use restrictions in place in 2002. All localities included the 2002 drought year but the remainder of the data period varied by locality. The typical water use period was between 1999 and 2003. For each locality, historical data on climatic and demographic characteristics was also obtained.

A short survey was sent out the participating localities to determine what type of drought management policies have been implemented, how have such policies been publicized and how aggressively policies were enforced. The survey also requested information on current and past water pricing rates. The process of collecting this survey information is still underway.

PRELIMINARY RESULTS

Given that key information about drought management policies and pricing is still being collected, a final regression model has not been estimated at this time. However, preliminary evaluation of the data can be made using descriptive statistics. The remainder of this paper will provide a descriptive analysis of residential water use in Virginia and the experience during the 2002 drought.

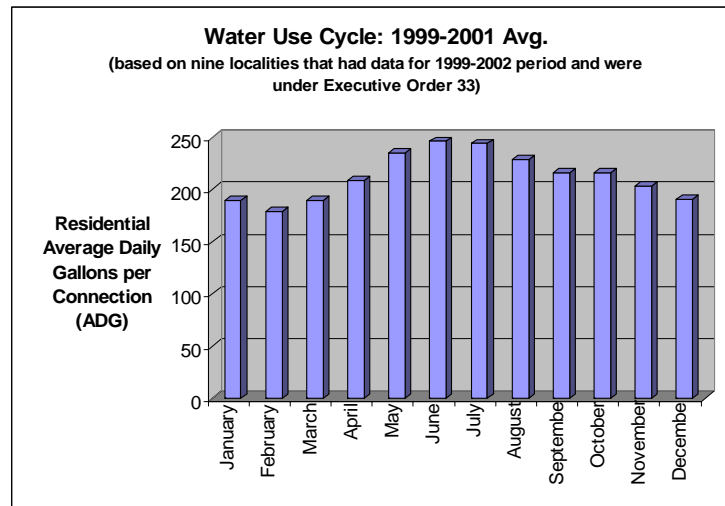


Figure 1. The general water use cycle.

Figure 1 represents the average annual residential water cycle for the localities in this analysis. Water-use is at its lowest during the winter months from December-March. Water-use then begins to increase in April, the start of the growing season, and continues to increase until it peaks in June and July. Water-use then slowly declines the rest of the year back to its winter baseline. The main point to take from this figure is that there are two main components of residential water-use: 1) Baseline uses that include mostly indoor use of water that can be approximated by the winter months, and 2) Seasonal uses during the growing season and warmer weather that include watering lawns and gardens, filling swimming pools, washing cars and driveways, *etc.* that can be approximated by the difference between summer and winter usage. As a rough rule, the season component is much more discretionary than the baseline component. Water use can be reduced in the baseline component by being more careful of household use (shorter showers, running only full loads of laundry, *etc.*), but only to a certain degree. In contrast, most of the seasonal component could potentially be reduced in times of water shortages.

Table 1. Comparison of 2002 water use to 1999-2001 average.

	Jan.	Feb.	March	April	May	June	July	Aug.	Sept.*	Oct.*	Nov.*	Dec.
1999-2001 Average	189	179	189	208	235	246	244	229	216	216	203	191
2002	188	172	178	199	233	272	273	245	199	181	175	179
<i>Difference (2002-avg)</i>	-1	-7	-12	-9	-1	26	29	16	-17	-35	-28	-11

* Indicates months when Executive Order 33 was in place.

Based on nine localities that had data from the 1999-2002 time period and where also under Executive Order 33.

Table 1 compares the 1999-2001 average (as seen in Figure 1) to the water use during the drought year of 2002. Water use rises above the expected average from June to August and then rapidly falls from this increased level during September, and continues at lower than expected levels for the remainder of the year. This reduction in water use matches quite closely with the September 1 effective date for Executive Order 33. The partial drop in August (from +29 to +16) could be attributable to a number of localities enacting self-imposed mandatory restrictions during that month. Both of these hypotheses assume that other factors influencing water demand are remaining constant. The regression analysis will be able to control for these other factors and determine if these drops in water use were in fact due to the hypothesized reasons.

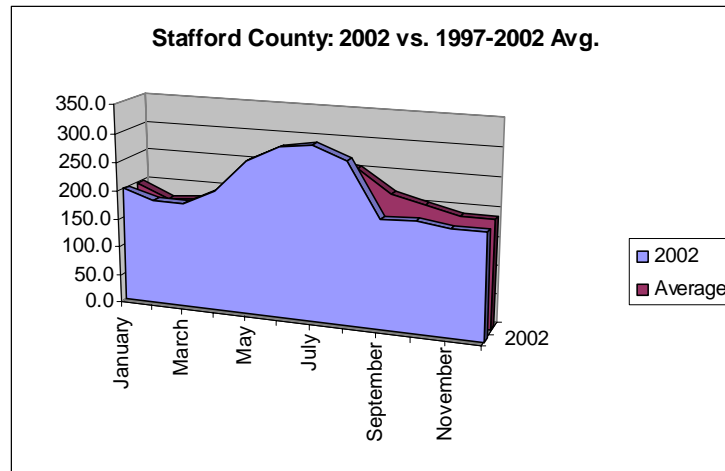


Figure 2. Effects of mandatory restrictions (1).

The impact that mandatory restrictions on water use can be illustrated in Figure 2. Mandatory restrictions were initiated in mid-August for Stafford County and were in effect until mid-November. By comparing water use in 2002 against a five year average (1997-2002 which was also a relatively dry period), it appears as if mandatory restrictions played an important role in reducing water use in 2002.

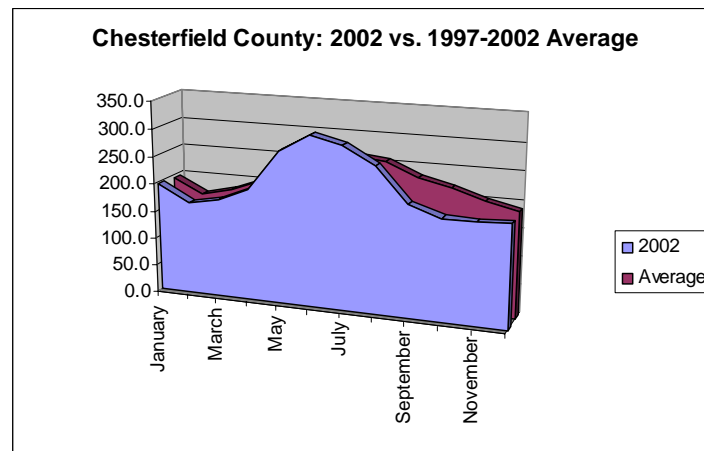


Figure 3. Effects of mandatory restrictions (2).

Yet, the experience with the drought impacted each locality differently. Figure 3 illustrates how the intensity of restrictions can impact the overall effectiveness of mandatory restrictions. Chesterfield County has very similar demographic characteristics as Stafford County (both suburban areas) with similar mandatory restrictions. The main difference between the two is that Chesterfield County had a much higher level of enforcement (345 violations issued with \$26,000 in fines). Both counties had similar short-term water use responses (August and September), but Chesterfield County seemed to have a much higher sustained level of water use reduction. One potential explanation for this persistence is the influence of observing continuing enforcement activities. Two main points here are that 1) enforcement level does appear to be having an impact, and 2) enforcement may have a lagged impact (enforcement this month may effect water use next month).

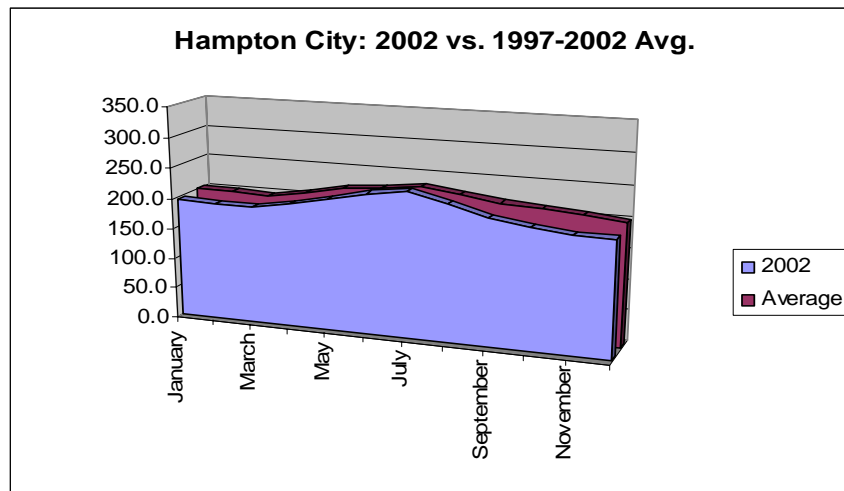


Figure 4. Urban (city) water use.

Furthermore, response to mandatory drought restrictions might be different between cities and more suburban areas. Figure 4 plots water use for Hampton City. Compared to Chesterfield and Stafford, water use varies less over the year. This could be due to the fact that urban residents generally having much smaller lawn sizes (or no lawns at all) and hence less discretionary water usage during the summer than suburban households. Because there is less discretionary use, the absolute effects of voluntary and mandatory restrictions will not necessarily be the same as with counties.

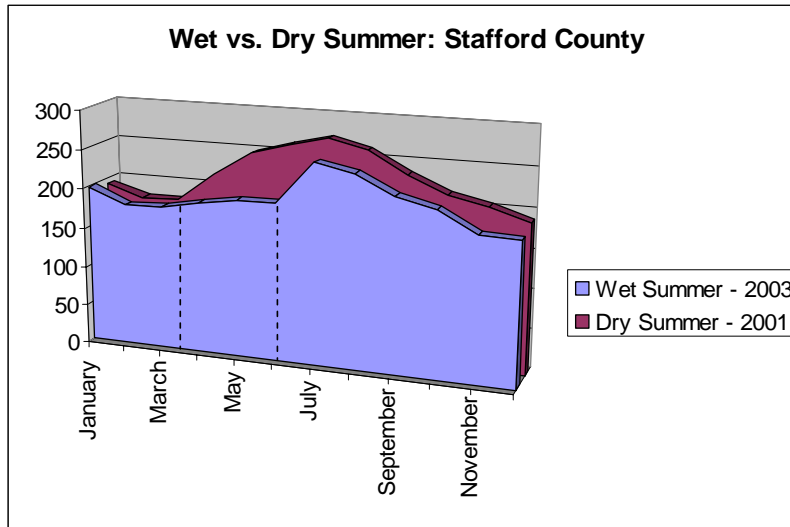


Figure 5. Effect of rain on water use.

Yet, other factors beside drought management policies could also explain changes in water use during the summer and fall of 2002. As previously noted, the increase in summer water usage is largely due to discretionary outdoor water uses that are related to the growing season and rainfall. The impact of rain on water use can be seen by comparing the water-use in Stafford County in a dry summer (2001 with no water-use restrictions) versus a wet summer (2003) in Figure 5. As can be seen, water-use during the wet year only increases slightly in April and continues this trajectory until the frequent rains let up in late June and July. At this point, water-use more closely resembles the dry year of 2001. Thus, localized rainfall in the late summer and fall of 2002 could partially explain some of the decline in residential water use. The multiple regression analysis will be used to isolate these separate impacts.

CONCLUSIONS

Some of the important conclusions that can be drawn from this graphical representation of water use are:

1. There appears to be evidence that voluntary and mandatory restrictions reduced water use during the drought of 2002. The extent to which this occurred cannot be completely recovered until all factors that influence water demand are controlled for.
 2. Higher enforcement levels seem to improve the effectiveness of mandatory restrictions.
- The main implications should be regarded as tentative until more complete statistical analysis is undertaken.

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ANALYSIS OF HOME PLUMBING PIPE FAILURES

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KEY WORDS: pinhole corrosion leak, random non-homogeneous poisson process, GIS (geographic information system)

ABSTRACT

Pinhole corrosion leak in home plumbing has emerged as a significant issue. In the major water distribution system managed by municipalities and water utilities the costs are distributed among all subscribers. The home plumbing repair/replacement cost and possible water damage cost must be addressed by the home owner. There are also issues of the value of home, insurance rates, health consequences, and taste and odor problems. These issues have become major concerns to home owners. The writers are developing a decision support system to help the home owner to decide whether to continue to repair or replace the plumbing system. In this paper a methodology that accounts for random arrivals of leak events in determining the optimal replacement time is presented.

INTRODUCTION

While corrosion in city water distribution systems has received constant attention, at home plumbing systems have been made generally considered immune to corrosion. However, recent spurt of at home copper pipe corrosion is causing anxiety among homeowners. For most homeowners, the home is their most valuable asset. The possibility of falling home value, losing insurance, water damage, frequent repairs, and health concerns are some key issues involved in the home plumbing maintenance. The home owners near the hot spot areas are considering additional treatment to water in terms of corrosion inhibition, different plumbing materials such as plastics and stainless steel, and coating the interior of the pipe. Homeowners are also seeking advice on whether to continue to repair or replace the plumbing system. In this paper a methodology for the optimal replacement of plumbing system is presented.

METHODS

Spatial Distribution of Plumbing Leaks

In the Washington, D.C. area there is a spread of home plumbing leaks. Based on the reported leaks an attempt has been made to identify the spatial distribution of these leaks. Figure 1 shows the areas with their zip codes. Table 1 identifies the areas designated by their zip codes having the most leak incidents. The ranking is based on number of leaks per person and absolute number of leaks. In Table 1, column (1) contains zip codes for the ranked number of leaks per person; column (2) has the population, column (3) has the number of leaks, column (4) has the

ranked number of leaks per person, column (5) has the ranks for the number of leaks per person, and column (6) has the ranks for the number of leaks. From Table 1 data, it is seen that the areas that have significant pinhole leak problems are located in close proximity with the water treatment plants. The purpose of this analysis is to calibrate the leak prediction model parameters in terms of hydraulic and water quality variables.

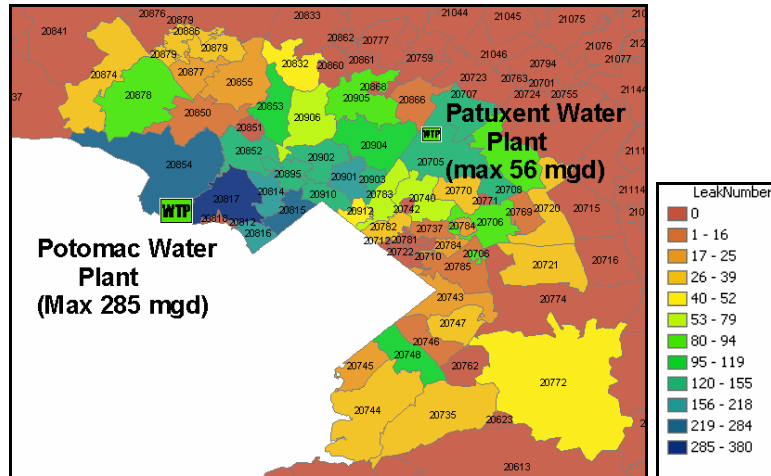


Figure 1. Distribution of observed leaks.

Table 1. Pinhole leak analysis for the WSSC distribution area.

Zip (1)	Population 2001 (2)	# of Leaks (3)	Pinhole/ Person (4)	Rank by Number of Leaks/ person (5)	Rank by number of Leaks (6)
20816	15143	173	0.011424421	1	6
20817	33954	380	0.011191612	2	1
20815	28270	284	0.010045985	3	2
20814	25916	218	0.008411792	4	4
20903	18039	127	0.007040302	5	12
20895	19713	134	0.006797545	6	10
20705	20551	129	0.006277067	7	11
20707	25815	149	0.005771838	8	8
20901	36935	205	0.005550291	9	5
20854	48226	261	0.005412018	10	3
20905	17295	93	0.005377277	11	15
20853	26432	119	0.004502119	12	13
20910	37144	155	0.004172949	13	7
20740	21503	79	0.003673906	14	16
20708	25633	94	0.003667148	15	14
20852	40252	140	0.003478088	16	9

Repair/Replacement Analysis

At the time of the n th leak, a decision has to be made whether to replace the plumbing system at a cost of F_n or to repair it at a cost of C_n . The scenario also implies that for the previous $(n - 1)$ leaks only repairs have been performed. If we assume that the plumbing will be replaced (C_n included in the sum should be observed to adjust F_n when necessary) at the time of n th leak, t_n , we can write the present worth of the total cost of the pipe as

$$T_n = \sum_{i=1}^n \frac{C_i}{(1+R)^{t_i}} + \frac{F_n}{(1+R)^{t_n}} \quad (1)$$

in which: R = discount rate, t_i = time of i th leak measured from the installation year (year), C_i = repair cost of i th break, F_n = replacement cost at time, t_n , T_n = total cost at time '0' (present worth).

When the system is new, it tends to experience very few leaks. An old system experiences more leaks under the same conditions. Therefore, the combination of varying time interval between breaks (accelerated breaks towards the end), relatively smaller repair cost, and a generally large replacement cost leads to the existence of a "U" shaped present worth of the total cost curve over time. The derivation of the threshold break rate, seeks the time of the minimum present worth total cost. Loganathan *et al.* (2002) have presented the following threshold rate equation

$$\text{Brk}_{th} > \frac{\ln(1+R)}{\ln\left(\frac{C_{n+1}}{F_n} + 1\right)} \quad (2)$$

in which: C_{n+1} = repair cost at $(n+1)$ th leak and F_n = replacement cost.

Now, from the observed data for any given system we can derive a current leak rate. Whenever the current leak rate, Brk_{cur} equals or exceeds Brk_{th} , the plumbing system should be replaced. Figure 2 shows the observed behavior for leak rate as a function of plumbing system installation period.

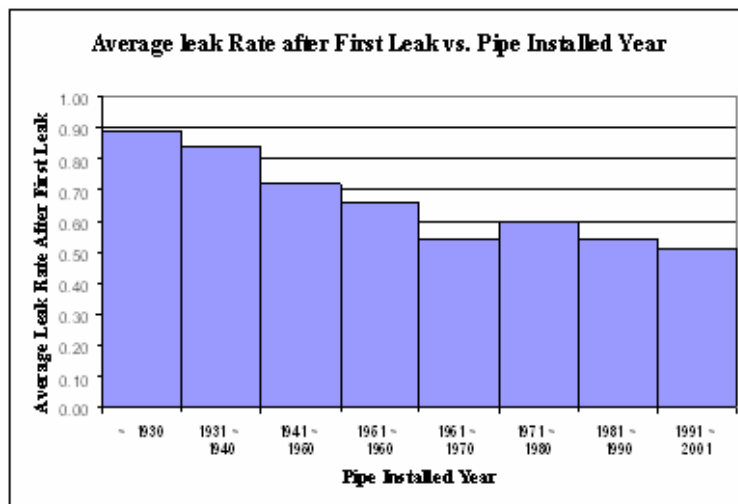


Figure 2. Leak rate as a function of installation period.

Current Leak Rate

To mimic the actual rate of occurrence of leaks, we adopt the following model due to Shamir and Howard (1979) given by $N(t) = N(t_0)e^{A(t-t_0)}$, where $N(t)$ = number of leaks per 1,000 ft length of pipe in year t ; t = time in years; t_0 = base year for the analysis (pipe installation year, or the first year for which data are available); and A = growth rate coefficient (1/year). Setting the leak rate $N(t)$ to be equal to the Brk_{th} given in the previous section, we obtain the optimal time of

replacement $t^* = t_0 + \frac{1}{A} \ln \left[\frac{\ln(1+R)F_n}{N(t_0)C_{n+1}} \right]$. For different values of t^* such as 35 and 40 years of

replacement time, initial values for $N(t_0)$, A , and t_0 can be obtained.

RESULTS AND DISCUSSION

Leak Arrival Process

The leaks are assumed arrive at the $N(t) = N(t_0)e^{A(t-t_0)}$ for a chosen set of $N(t_0)$, A , and t_0 .

The arrival process is modeled as a random non-homogeneous Poisson process (Law and Kelton 2000). From the simulated leak arrival times, the leak rate is calculated. If a close match is obtained between the calculated arrival rate and Figure 2, that set of parameters is retained as the optimal $N(t_0)$, A , and t_0 . Table 2 shows the adopted parameters. The early leak times less than a threshold value is attributed to installation flaws and the model is refined. Table 3 contains the generated leak arrival times.

The calibration process currently is based on the data as reported by consumers shown in Figure 2. There is an ongoing survey which should lead to additional data. The other attempt towards calibration is in terms of water quality and hydraulic variables. As shown in Figure 1, the distance from the water treatment plant plays a crucial role. The chlorine decay process, pressure zone and corrosion behavior of the major system will be studied to relate the calibration parameters to the corrosion process.

SUMMARY

A procedure for determining the optimal replacement time for plumbing systems is presented. Currently, the method of calibration is based on leak occurrence time data reported by the consumers. The strength of the method is in the manner it accounts for what is an economically sustainable replacement time. Therefore, some inherent errors in the data and lack of physically based process rules can be tolerated. The future developments will merge the both the economic viability and physical phenomenon.

Table 2. Parameters corresponding to optimal replacement time $t^* = 35$ years.

	$N(t_0)$	A
1931 ~ 1940	0.56	0.00239
1941 ~ 1950	0.45	0.00864
1951 ~ 1960	0.38	0.01347
1961 ~ 1970	0.30	0.02023
1971 ~ 1980	0.28	0.02220
1981 ~ 1990	0.22	0.02909
1991 ~ 2001	0.12	0.04641

Table 3. Possible leak arrival times for each decade.

1935	1945	1955	1965	1975
2.008706	2.430818	2.584726	4.395393	4.424354
2.593618	2.888229	4.022403	4.499624	6.240568
3.53083	3.429643	5.437175	6.625074	6.606802
4.287193	3.830268	5.53846	7.255024	8.638177
5.368382	5.44524	8.921351	9.56334	12.2834
7.357714	6.001327	9.766668	12.67435	12.73338
7.548096	8.010616	9.864177	13.14836	15.97968
8.213306	9.678512	10.22678	13.82237	16.31268
8.56669	11.86315	13.51015	16.93146	17.93062
9.732588	12.07392	14.66152	20.53313	17.98454
10.3015	14.55007	16.14691	21.36176	20.63563
11.73037	16.55121	17.27064	21.57251	24.71425
12.16936	17.19761	18.90004	23.38698	27.7037
14.32201	18.19462	19.18257	27.67447	31.63461
15.67608	19.50616	21.98365	32.2706	32.39679
18.58592	21.27403	23.70982	33.76195	34.1885
18.69301	21.30834	26.40986	35.50033	35.01319
18.70529	24.21335	27.22462	38.40354	38.15662
18.79593	24.57382	31.23096	40.08388	38.84498
19.41343	26.1938	31.36251	40.85306	40.20684
20.06656	29.48026	33.68048	45.83971	40.57304
22.41753	30.01544	34.99184	48.89913	45.89776
25.01933	32.55151	35.47126	51.85977	47.61501
25.22299	34.90772	37.14148	52.15133	51.61726

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TASTE THRESHOLDS OF COPPER IN WATER

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Copper pipes are estimated to be in 70-90% of current U.S. household drinking water distribution systems. Corrosion of pipes can cause taste and odor problems that may lead to metallic or bitter tasting water. The purpose of this study was to determine if humans, by using their sense of taste, can detect copper corrosion in their drinking water system. A modified triangle test was used to define taste thresholds of copper in varying types of water. Copper containing solutions were prepared using copper sulfate at concentrations ranging from 0.025-8 mg/L Cu. Water samples with varying chlorine, alkalinity and ion contents were tested to see if the taste threshold varied with different water quality. These taste thresholds were analyzed to see at what concentrations humans can detect copper and if varying water quality changed the taste threshold. Taste thresholds were also analyzed to see if humans can detect copper in their water at levels below Environmental Protection Agency (EPA) standards. The EPA primary and secondary drinking water standards for copper are 1.3 and 1.0 mg/L Cu respectively. Preliminary results using distilled water resulted in a taste threshold range of 0.3 – 3 mg/L Cu. A follow-up study will be conducted with a large panel and varying water quality.

WHAT DOES THE NEW BACTERIAL STANDARD MEAN IN VIRGINIA LAKES?

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KEY WORDS: fecal coliforms, *Escherichia coli*, bacterial methods, lakes

ABSTRACT

Escherichia coli colony counts in Virginia's waters are replacing the health department standard, fecal coliform colony counts. A comparison of methods and interpretation will be discussed for Smith Mountain Lake.

Recently the Department of Health changed its monitoring standard for recreational waters to the measurement of *Escherichia coli* (*E. coli*) populations instead of fecal coliform populations. The standard threshold is a geometric mean of 126 colonies per 100 mL of sample water and 235 colonies per 100 mL grab sample. In 2003 we evaluated the *Escherichia coli* populations at Smith Mountain Lake using a commonly used method, ColibluTM and in 2004 the ColiscanTM method was used as a means of comparison and to develop a reliable affordable method for *E. coli* identification and enumeration.

Water samples are collected from fourteen sites on Smith Mountain Lake six times each summer. The membrane filtration method for bacterial analyses was used with m-FC media prepared with rosolic acid, to measure fecal coliform populations, ColibluTM media and ColiscanTM media for *E. coli* identification and enumeration. Characteristic colonies were counted and recorded after prescribed number of hours of incubation and temperature.

Using the ColibluTM method, the mean total coliform population estimates for all sample sites was compared to the mean *Escherichia coli* population estimates. This represents averages from 1-13 % of *Escherichia coli* organisms per total coliform organisms. Total coliforms include many different species of bacteria, most of which are not indicators of fecal pollution. This method appeared to over estimate the *E. coli* populations.

In 2004 the ColiscanTM and the BiologTM methods was evaluated at the same sample sites in Smith Mountain Lake. Fecal coliforms were measured also at the same sites for the first two sample dates and *E. coli* counts compared favorably to fecal coliform populations

INTRODUCTION

Escherichia coli colony counts in Virginia's waters are replacing the health department standard of fecal coliform colony counts (9 VAC 25-260 of the *Virginia Administrative Code*. A comparison of methods and interpretation will be discussed for Smith Mountain Lake.

Fecal coliform bacteria refers to a group of many different species of bacteria most (but not all) of which live in the intestines of warm-blooded animals (mammals and birds). *Escherichia coli* (*E. coli*) a single species (but with many varieties, or strains) is within the fecal coliform group; found only within the intestines of warm-blooded animals. Most strains of *E. coli* do not cause illness, but strain 0157:H7 produces a toxin and can cause serious illness (USDHHS 2004).

The Colibblue™ and Coliscan™ methods are evaluated here and compared to the Biolog™ method of *Escherichia coli* identifications methods. The Coliscan™ method, which utilizes chromogenic substrates for the enzymes glucuronidase and galactosidase that result in virtually insoluble colored chromophores that have very minimal diffusion from the colonies of target bacteria. The Analytical Methods Staff and the Office of Ground Water of the U.S. EPA that the technology of the Coliscan™ MF medium found the method acceptable for use in National Primary Drinking Water Regulation compliance monitoring as a modification of the already approved MI agar medium (APHA 1999).

METHODS

Water samples were collected from fourteen sites on Smith Mountain Lake six times each summer. The membrane filtration method for bacterial analyses was used with m-FC media prepared with rosolic acid, to measure fecal coliform populations (APHA 1995), Colibblue™ media and Coliscan™ media for *E. coli* identification and enumeration. Characteristic colonies were counted and recorded after prescribed number of hours of incubation and temperature.

The methods described as Colibblue™ and Coliscan™ used media that has a special indicator chemical that *Escherichia coli* take up and grow as blue colonies. The water samples are filtered onto a sterile cellulose filter with a pore size of 0.45 µm, which is gridded. The filter is placed on a sterile absorbent pad with the Colibblue™ or Coliscan™ media added in a small sterile Petri plate (47 mm). The Petri dish with media and sample filter are incubated at 35 +/- 2.0 °C for 18-24 hours. After the allotted time the plates are observed and the blue colonies on the filter are counted. On the Colibblue™ and Coliscan™ media, non-fecal total coliforms show up as red colonies (total coliforms – cfus) (IDEXX 2004, Micrology 2004).

The Biolog™ method is a standardized micro-method that uses 95 biochemical tests to identify/characterize a wide range of bacteria. We first employ one of the *E. coli* methods as described earlier in this section to isolate colonies. The fecal coliform colonies are identified as blue tinted colonies on the mFC media plates and the *E. coli* colonies as blue tinted colonies on the Colibblue™ or Coliscan™ media plates. These colonies were aseptically transferred and grown on a special medium (Biolog™ Universal Growth Agar with 5% sheep blood). After incubation on media, cells are transferred to sterile inoculating fluid until an acceptable density is reached (as determined by a turbidity meter). This cell suspension is immediately inoculated

into the appropriate Micro-Plate, a multi-well plate containing the 95 different carbon sources such as sugars, alcohols, and amino acids. The Micro-Plates are incubated at 35-37° and read on the Biolog™ MicroStation Reader at 4-6 hours and again at 16-24 hours. The pattern of growth and color change (tetrazolium redox dye changes to purple when growth is present) is read into the Micro Log software database. These patterns are compared to the archived data from 524 Gram-negative aerobic bacterial species and are identified to species if possible (Biolog 2002). Specifically, we record colonies identified as *Escherichia coli* bacterial strains. We also record the number of different Gram-negative bacterial species and potentially find pathogenic bacterial species.

RESULTS

Using the Colibblue™ method in 2003, the mean total coliform population estimates for all sample sites was compared to the mean *Escherichia coli* population estimates. This represents averages from 1-13 % of *Escherichia coli* organisms per total coliform organisms. Total coliforms include many different species of bacteria, most of which are not indicators of fecal pollution.

Using the Colibblue™ method, the mean total coliform population estimates for all sample sites on July 7th was 571 cfus and the mean *Escherichia coli* population estimate was 34 cfus, while on August 6th, the mean total coliforms estimate was 1296 cfus and the mean *Escherichia coli* population estimate was 80 cfus. This represents averages from 1-13 % of *Escherichia coli* organisms per total coliform organisms. Total coliforms include many different species of bacteria, most of which are not indicators of fecal pollution. The Colibblue™ method appears to overestimate the *Escherichia coli* populations based on results showing higher number of *E. coli* colonies than fecal coliform colony counts. Figure 1 shows the increase in *E. coli* as the summer progresses based on the Colibblue™ method.

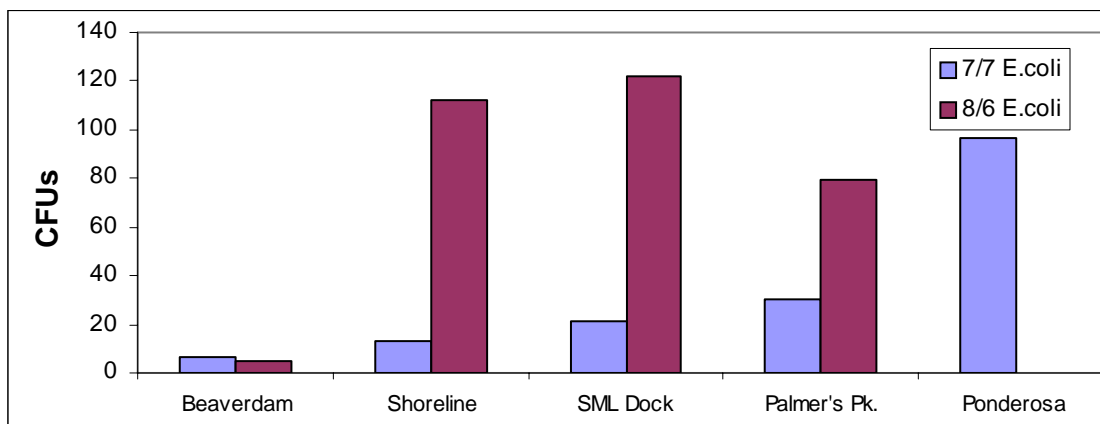


Figure 1. *Escherichia coli* average colonies per site sampled at Smith Mountain Lake in 2003 using the Colibblue™ method.

In 2004 the Coliscan™ method was evaluated at the same sample sites in Smith Mountain Lake. Fecal coliforms were measured also at the same sites for two of the sample dates and a conversion factor from 44% to over 100% for conversions from fecal coliform counts to *E. coli* counts was suggested by the data.

In a comparison of fecal coliform and *E. coli* populations using Coliscan™ in Smith Mountain Lake during the summer of 2003 and 2004 the interpretation of the results were very similar with the major difference being the significant difference in the scales of the graphs as seen in Figures 2 A & B, 3 A & B and 4 A & B.

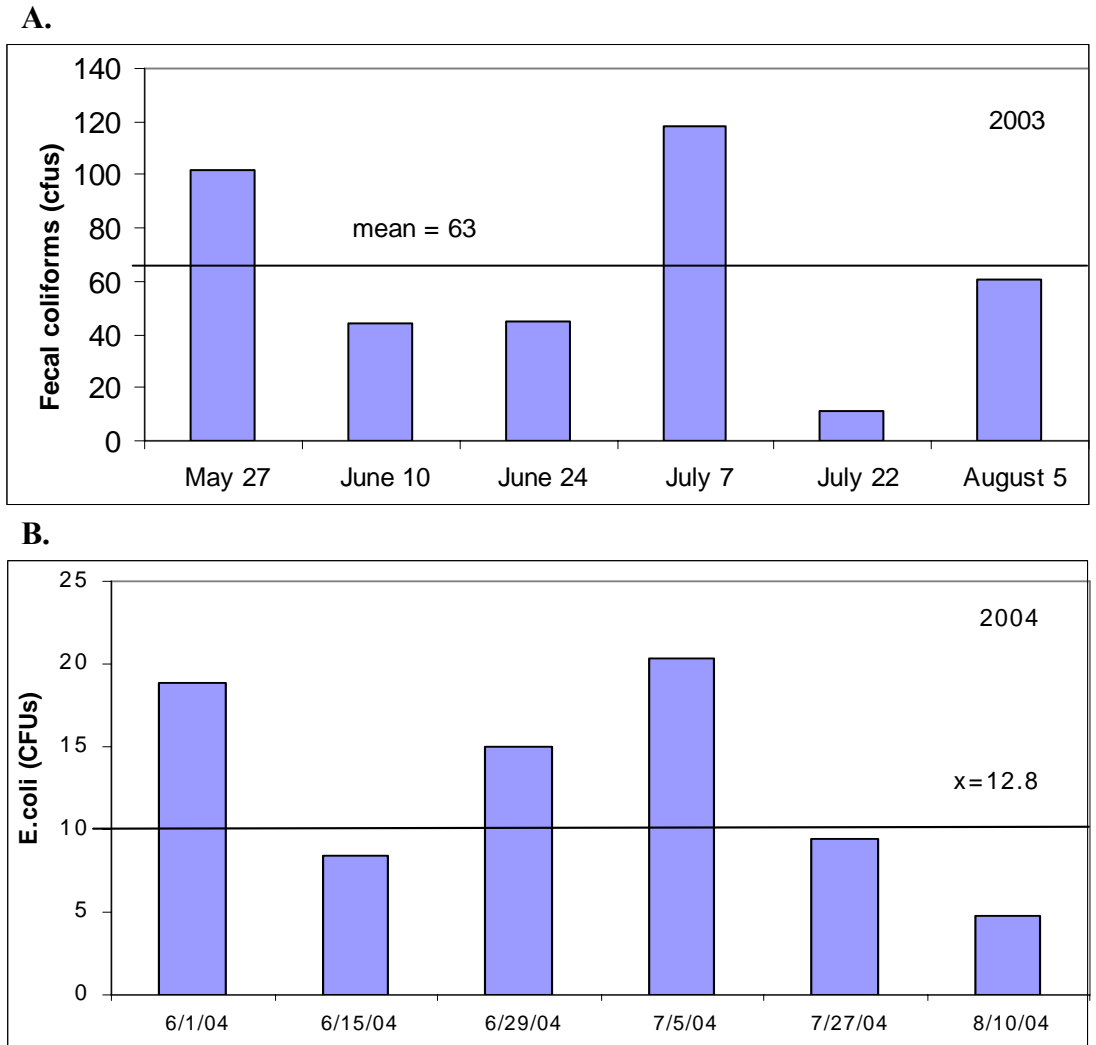


Figure 2A. Fecal coliforms vs. week sampled in 2003 and Figure 2B. *Escherichia coli* samples in 2004 on Smith Mountain Lake (Each sample date included 14 sites with 2 samples per site and three replicate filters per sample, n =84).

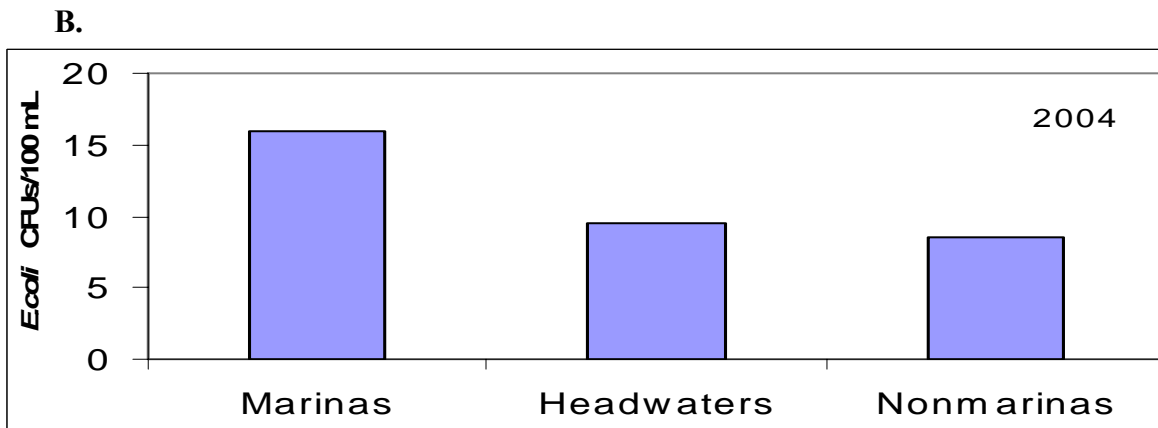
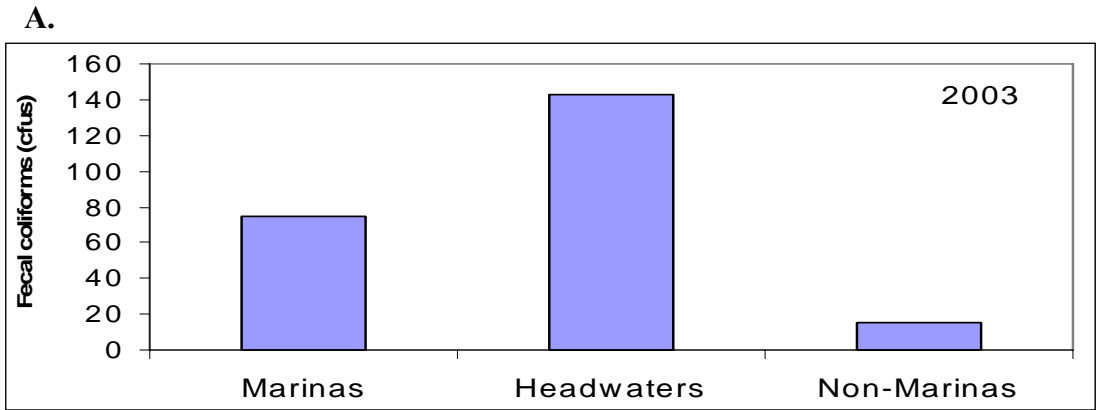


Figure 3A. Mean fecal coliform colonies vs. site type in 2003 and Figure 3B. *Escherichia coli* colonies vs. site type in 2004 on Smith Mountain Lake (7 marina sites, 5 non-marina sites, and 2 headwater sites).

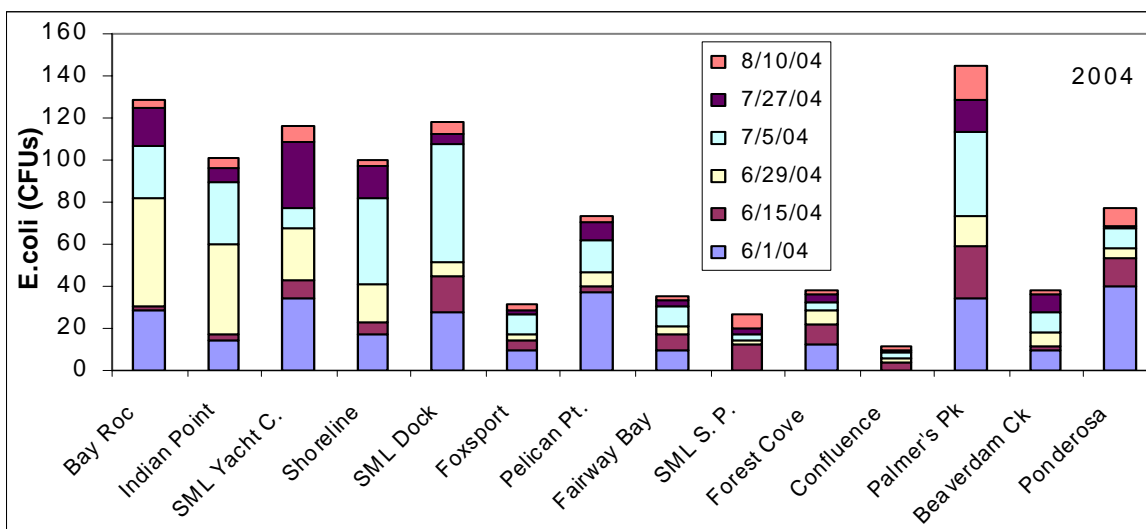
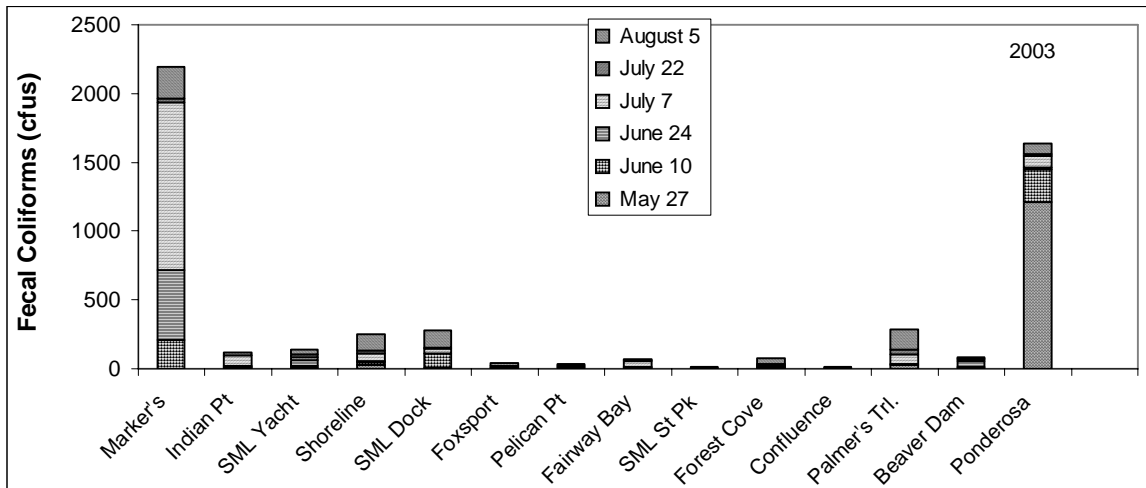
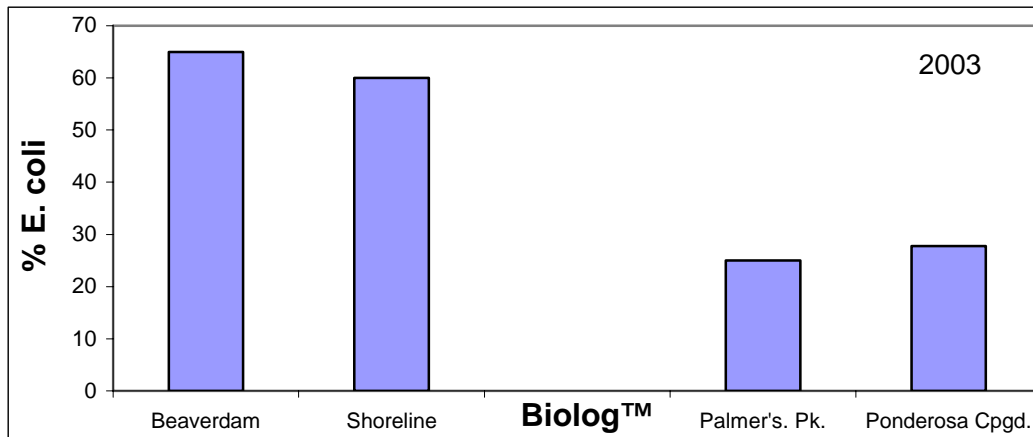


Figure 4A. Sum of fecal coliform counts in 2003 and Figure 4B. Sum of *E. coli* counts at each site for all sample dates for Smith Mountain Lake. There are two sampling stations at each site.

The Biolog™ method gives a different type of result and because of its time consuming and expensive nature only a few samples were tested with this method. In 2003 we evaluated four samples, two in the Roanoke channel (Beaverdam Creek and Shoreline Marina) and two in the Blackwater channel (Palmer's Trailer Park and Ponderosa Campground). The average percent *Escherichia coli* per fecal coliform was 44.4% and the Roanoke channel sites had a higher percent of *E. coli* (65 and 60 %) then the Blackwater channel sites (25 & 28 Three of the five colonies evaluated) in 2003 were identified as *E. coli* and they were all blue colonies from mFC media (fecal coliforms (Figure 5A.). In 2004 thirty colonies from Coliscan™ plates were inoculated into the Biolog™ media, seventeen of the blue colonies were identified as *E. coli* and five other species of fecal coliforms were identified (Figure 5B).

A.



B.

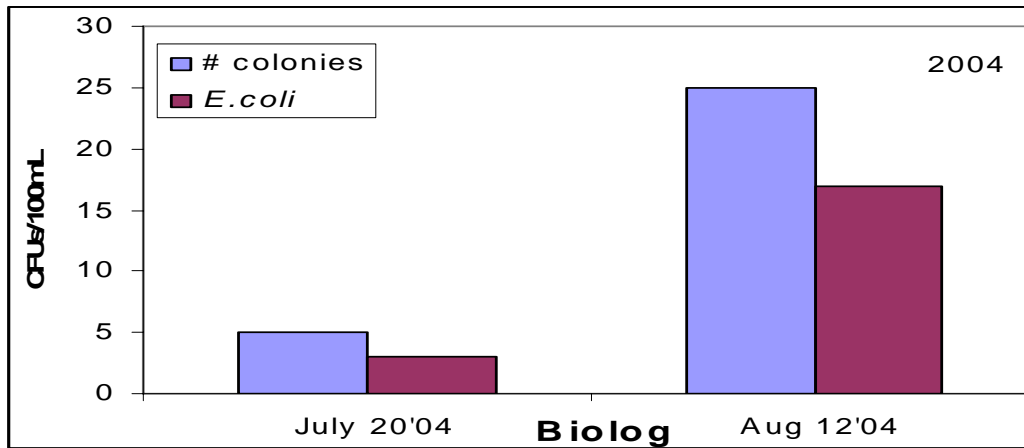


Figure 5A. Percent of *Escherichia coli* colonies per fecal coliform colony per site sampled in 2003 and Figure 5B. Number of blue colonies tested and the number of *E. coli* identifications in 2004 at Smith Mountain Lake using the Biolog™ method.

Comparisons were made with the 2003 fecal coliform data and the 2004 fecal coliform and *E. coli* data as shown in Figures 6 and 7. The highest values for 2003 fecal coliforms values were left off the graphs because the high numbers would have made the scale too broad and smaller values could not be compared. Ponderosa campground had fecal coliform populations of 1210 cfus and 238 cfus for sample week 1 and 2 in 2003 and Marker's/Bay Roc Marina had values for fecal coliforms of 208 and 505 in sample week 2 and 3. The results indicate in 2004 neither fecal coliforms nor *E. coli* violated the Virginia Department of Health standards for fecal coliforms or *E. coli*. The ratios between fecal coliforms and *E. coli* do not appear to be consistent, therefore a conversion factor to convert previous years fecal coliform data to the newer standard of *E. coli* data is not an effective method of comparing different years data.

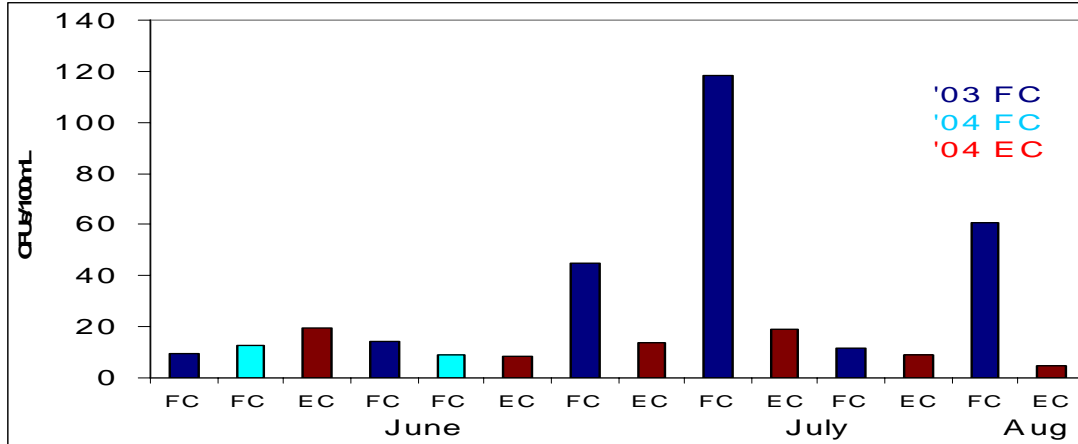


Figure 6. Comparison of fecal coliform and *Escherichia coli* colony numbers at different sample dates in 2003 and 2004 in Smith Mountain Lake.

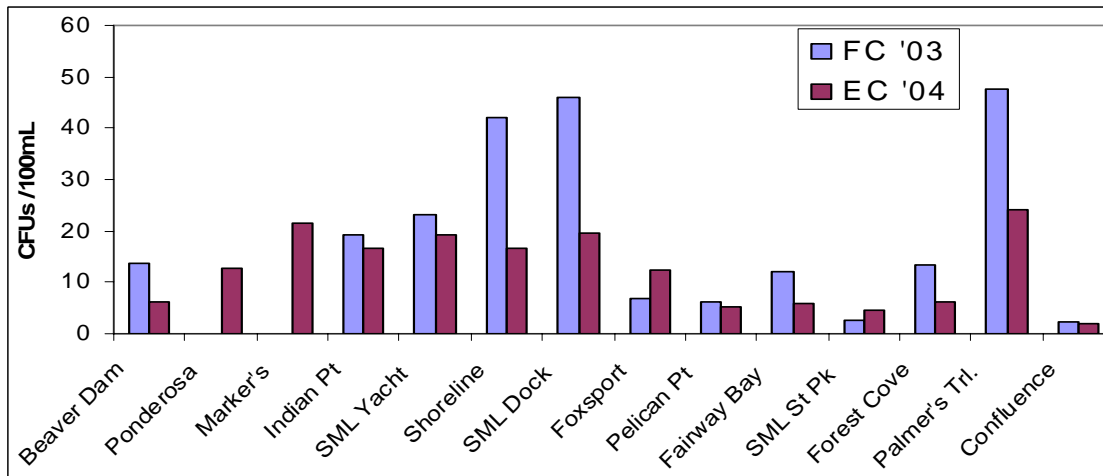


Figure 7. Comparison of fecal coliform and *Escherichia coli* colony numbers at different sample sites in 2003 and 2004 in Smith Mountain Lake.

DISCUSSION

Methods for *E. coli* identification and enumeration must still be evaluated with more sample sites and more bodies of water. The Coliblue™ method does not appear to be useful but the Coliscan™ method needs further evaluation. The cost of *E. coli* data gathering to evaluate water quality will be greater than the fecal coliform methods are but may require less expertise in performing the test.

Another *E. coli* identification and enumeration method not studied in this research is the Colilert™ method as described below. This method will be evaluated during the 2005 sampling season.

Colilert™ is another method for identification and quantification of *Escherichia coli* and is approved and recommended by the Virginia Department of Environmental Quality (Du 2004).

Colilert™ uses the patented Defined Substrate Technology® (DST®) to simultaneously detect total coliforms and *E. coli*. Two nutrient-indicators, ONPG and MUG, are the major sources of carbon in Colilert™ and can be metabolized by the coliform enzyme β -galactosidase and the *E. coli* enzyme β -glucuronidase, respectively. *E. coli* use β -glucuronidase to metabolize MUG and create fluorescence. Colilert™ is a U.S. EPA-approved, 24-hour test for drinking and source waters (APHA 1999).

Although comparable in cost to the other two methods tested, less expertise is needed in laboratory analyses of samples for *E. coli* and it may be less subjective than determining the color of the colonies which is necessary in the other two methods evaluated.

Replacing the bacterial water quality standard in Virginia is an understandable and useful change, however further study must be done to find the method best suited for the resources available to scientists and citizens.

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IDENTIFYING SOURCES OF FECAL POLLUTION IN A MIXED USE WATERSHED IN VIRGINIA

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KEYWORDS: bacterial source tracking, *Enterococcus*, *Escherichia coli*, antibiotic resistance analysis, fluorometry

ABSTRACT

Prince William County, a rapidly developing county in Northern Virginia, has watersheds of mixed rural and urban uses. As part of Virginia Department of Environmental Quality regulations, recreational waters must be tested and remain under a certain standard for levels of fecal indicator bacteria. During the 2003-04 monitoring, 18 sites were sampled monthly, along with two storm events, for *Escherichia coli* and *Enterococcus* to provide 5,184 isolates at 24 isolates per sample. These water-sample isolates were compared to a known source library (KSL) of fecal contamination collected from the Northern Virginia region.

Bacterial source tracking (BST) was used to identify the sources of fecal pollution, allowing optimal effectiveness in remediation efforts. Antibiotic Resistance Analysis (ARA), a biochemical test, classifies isolates based on differential resistance patterns to a combination of antibiotics. In addition, fluorometry is being used on water samples to detect optical brighteners present in the water. Optical brighteners are found in household detergents and indicate the presence of human wastewater. This project involves the first thorough testing of fluorometry for the detection of human signatures in freshwaters and will be compared with the results obtained from BST.

To date, fluorometry has indicated human sources of pollution at those locations where BST indicated the same thing. In addition, BST has shown that birds and wildlife are major contributors, especially in urban areas, while livestock are a major source in agricultural areas.

This project expands the sampling to new watersheds and continues the sampling of previous watersheds at reduced levels. Applicability of results will be compared across neighboring watersheds with substantially different uses. Sampling over the summer of 2004 will further characterize sources of fecal pollution during high levels of contamination as a result of livestock and wildlife activities. This will allow direct remediation of identified sources in water improvement strategies.

METHODS

Three library profiles were created consisting of 1,991, 1,250, 2,127 *Enterococcus* known source isolates each. The first two libraries were designed for rural specific and urban specific profiling respectively. Both libraries contain wildlife, bird, and human profiles; however the rural library also contains ARA fingerprints for livestock, while the urban contains ARA fingerprints for pet species. The third library is a combination of the urban and rural libraries, in which the pet species fingerprints were added to the rural library. Each known source isolate was fingerprinted by evaluation for growth (1) or no growth (0) on a total of thirty concentrations of nine antibiotics. The isolate profiles were then combined into libraries and analyzed using discriminate analysis (Wiggins 1996). The KSL isolates came from the Northern Virginia region and 70 isolates were collected locally in Prince William County.

Samples were taken for the months of June, July and August 2004 from ten locations in Prince William County, Virginia. Duplicate, samples were taken for the months of July and August at two of the sites. Sediment samples were collected for two other sites for the months of July and August. Sampling for June and August occurred shortly after storm events.

Enterococci were isolated from each of the samples through membrane filtration according to standard methods (Eaton *et al.* 1995). ARA was performed on 24 isolates monthly from each location. The resistance profiles of the sample isolates were compared to the KSL's to determine the most likely source of each isolate (Hagedorn *et al.* 1999).

A separate aliquot of sample water was analyzed for fluorescence using the Turner Designs 10-AU-005-CE fluorometer. For the month of June samples were analyzed for Raw Fluorescence using a .005% Food Lion Clean (FLC) detergent as the standard. For the month of July the basic sensitivity was set to 32% and the Span to 59% using FLC. The month of August was processed at a basic sensitivity of 38% and a Span of 48% using a 125 µg/L standard of fluorescent brightener FB28, which was equivalent to the FLC solution.

A third aliquot of sample water was tested for turbidity using the Hanna Instruments HI93703 Turbidity Meter. A calibration curve was created between a known concentration of FLC and increasing levels of turbidity. This curve was used to create an adjusted level of fluorescence for each water sample. The adjusted levels of fluorescence were then correlated to the ratio of human classification for each month at each location.

Each location was analyzed by map to profile stream-adjacent dominant land for one mile upstream of the sample site. This was used to suggest the known source library for ARA comparison. Each library was then used in a side-by-side comparison of probability values (P-values) in isolate classification for each location.

RESULTS AND DISCUSSION

Within each individually tailored library different Average Rates of Correct Classification (ARCC) were found. The Rural KSL, which was used to analyze two locations, had an ARCC of 85.0%. The Urban KSL, which was used to analyze six locations, had an ARCC of 95.8%.

The combined library had a lower ARCC of 81.3% and was used to analyze two locations. The addition of the Pets isolates from the urban to the rural library blurred the borders between the classes of fingerprints and thereby reduced the ARCC by 3.7%. Ideally, the addition of this data would increase both the ARCC and the average probability of classification (Average P) (Table 1). As both the ARCC and most average probabilities were reduced, this suggests a need for a more complex ARA fingerprint. A third measure of the successful addition to the library would be to examine the rate of dissimilar or unknown classification. This would illustrate the need for new known source samples of novel patterns. The percentage classification below a P-value of 0.7 includes overlapping, therefore indefinite, and completely unknown ARA patterns. Discriminate analysis of the combined library developed a fifth template pattern such that fewer low P-values occurred.

Table 1. P-value statistics according to known source library and site; selected library in bold.

Library		Rural	Urban	Combined
Bull Run 1	Average P	0.868	0.901	0.857
	% P < .7	13.889	12.500	12.500
	% P < .3	1.389	4.167	0.000
Bull Run 2	Average P	0.837	0.837	0.821
	% P < .7	16.667	20.833	16.667
	% P < .3	0.000	6.250	0.000
Catharpin Run	Average P	0.815	0.778	0.786
	% P < .7	30.556	25.000	36.111
	% P < .3	0.000	12.500	0.000
Youngs Branch	Average P	0.754	0.736	0.819
	% P < .7	36.111	41.667	25.000
	% P < .3	0.000	9.722	0.000
Broad Run	Average P	0.792	0.672	0.810
	% P < .7	30.556	45.833	25.000
	% P < .3	0.000	18.056	0.000
South Run	Average P	0.858	0.923	0.875
	% P < .7	16.667	8.333	13.889
	% P < .3	1.389	1.389	0.000
Flat Branch	Average P	0.772	0.890	0.821
	% P < .7	32.143	10.714	25.000
	% P < .3	5.357	7.143	0.000
Buckhall Branch	Average P	0.855	0.819	0.841
	% P < .7	14.286	26.786	14.286
	% P < .3	0.000	8.929	0.000
Kettle Run 1	Average P	0.813	0.854	0.799
	% P < .7	31.944	25.000	34.722
	% P < .3	0.000	0.000	0.000
Kettle Run 2	Average P	0.842	0.881	0.835
	% P < .7	16.418	13.433	16.418
	% P < .3	0.000	2.985	0.000

$\text{Adj FB ppb} = \text{FB ppb} + 1.28147 \times e^{(2.68083 \times 10^{-3} \text{FTU})} \times \text{FB ppb} \times \text{FLC}_{.01} / 200$
Where $\text{FLC}_{.01} = .01\% \text{ FLC in FB ppb}$, FTU = turbidity units

Figure 1. Adjustment of fluorescence due to turbidity.

$\% \text{ Human} = 100 \times (0.04508 + 0.000782 \text{ FB ppb} + 0.00174472 \text{ FTU})$
--

Figure 2. Percentage human isolates as a function of fluorescence and turbidity.

Table 2. Percentage type of known source isolates for libraries.

	Rural	Urban	Combined
Bird	10.48	12.77	9.87
Human	30.89	40.06	29.11
Livestock	33.09	0.00	31.19
Pet	0.00	9.49	5.76
Wildlife	25.54	37.68	24.07

In comparing the fluorometric readings with turbidity data, it was found that the fluorometer readings require some adjustment for turbidity (Figure 1). However, more measurements of the turbidity/fluorescence relationship are required in order to increase calculation confidence. Due to the lack of confidence, turbidity measurements are included in the regression of human signature on fluorometry (Figure 2). In addition to more data points on the curve, the turbidity adjustment curve should be verified by sample centrifugation.

In every watershed and sample site the dominant source of contamination was from wildlife (Tables 3). This was likely influenced by the storm events just prior to two of the three samplings (Porter 2003). Further sampling over periods of lower flow will give a more complete site profile and possibly verify a high wildlife signature.

It is also of note that no *Enterococcus* exhibiting a bird fingerprint was detected. Bird ARA classification was present in all the libraries (Table 2). No signal may occur due to a low presence of birds near the sample sites, or there may have been a significant change in local bird *Enterococcus* antibiotic resistance. Recent results of bird classification within Prince William County suggest the former conclusion (Chapman *et al.* 2004). Yet, precisely because the rates of bird correct classification remain high within the libraries, new bird isolates should be acquired to verify the local integrity and applicability of the libraries. Further inspection of sample sites for bird populations is also necessary.

Discriminate analysis showed human signature in all sample sites except Youngs Branch and Broad Run. It was particularly pronounced, >15% of the isolates (Table 3), at both Kettle Run sites. Kettle Run 1 is located near a park and a golf course. Kettle Run 2 consists of houses on low-lying wood plots. Both of these sites have local septic systems that are aging. The high human presence is likely due to leaking septic tanks leaching into Kettle Run. The rural library sites require further monitoring to determine if the human *Enterococci* are present in significant

amounts. Typically humans share some of their intestinal flora with pets (Herbein and Hagedorn 1995) and some pet excrement may be included in the isolate percentages.

Table 3. Percentage contribution to fecal loading from each source over a three-month average based on identification at P > 0.70.

Urban KSL						
Source	Bull Run 1	Bull Run 2	South Run	Flat Branch	Kettle Run 1	Kettle Run 2
Bird	0	0	0	0	0	0
Human	2.78	4.17	5.55	1.34	17.9	16.4
Pets	4.17	6.25	0	7.14	0	0
Wildlife	81.9	68.73	86.1	82.1	63.9	77.6
Rural KSL			Combined KSL			
Source	Catharpin Run	Buckhall Branch		Source	Youngs Branch	Broad Run
Bird	0	0		Bird	0	0
Human	9.72	12.5		Human	0	0
Livestock	4.17	3.57		Livestock	0	5.55
Wildlife	55.6	69.6		Pets	18.1	13.9
				Wildlife	61.1	61.1

The remaining ten months of sampling will provide further detail into the site profiles. At this point it appears that remediating the waters of Prince William County to below Virginia Department of Environmental Quality limits for fecal indicators in recreational waters will be difficult. The wildlife signature is a difficult target for removal from suburban waters (Simmons and Herbein 1995). An exemption would be in order for these waters provided that the human and pet signatures are reduced to undetectable levels.

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BACTERIA TMDL DEVELOPMENT FOR PINEY RUN WATERSHED, VIRGINIA

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KEY WORDS: load-duration, stage implementation, BST, TMDL

ABSTRACT

This paper presents the development of a bacteria TMDL for the Piney Run watershed using load-duration methodology. Piney Run was listed as impaired due to violations of the water quality standard for fecal coliform bacteria. The load-duration approach was used to develop the TMDL for this watershed. Using this approach, the allocation of reductions to individual sources is accomplished by determining the relative contribution from various non-point sources using BST data. The results indicated that on an average basis, relative contributions of bacteria are 6% human, 22% pet, 41% livestock, and 31% wildlife. The load-duration curve was developed using flow data collected at the USGS gaging station, and the applicable water quality standard. Each water quality observation is then assigned to a flow interval by comparing the date of each observation to the flow record of the reference stream. The stream flow from the date of the water quality observation is then used to calculate a flow-duration interval and observed load in the stream. The loads on the load-duration curve were used to determine the annual loads. The results indicate that the highest exceedance of the water quality standard occurred at a high flow that is exceeded only 9% of the time. This represents the flow condition under which the largest bacteria reduction is required to meet water quality standards. To meet the instantaneous *E. coli* standard (235 cfu/100mL), this load would have to be reduced by 94% to an allowable load of 5.41×10^{13} cfu/yr. The required reductions in loads were then computed using the average annual *E. coli* load and the TMDL under average flow conditions. The required reduction of 94% is to be applied to each of the four non-point sources to attain a 0% violation of water quality standards. In order to evaluate interim reduction goals (*i.e.*, phased implementation), few reduction levels (70%, 50%, and 25%) and their associated violation rates were also assessed. Based on these reduction levels, an Implementation Plan would be developed to attain water quality standards in the watershed.

INTRODUCTION

Section 303(d) of the Clean Water Act and U.S. Environmental Protection Agency's (EPA's) Water Quality Planning and Management Regulations require states to develop Total Maximum Daily Loads (TMDLs) for waterbodies which are exceeding water quality standards. TMDLs represent the total pollutant loading that a waterbody can receive without violating water quality standards. The TMDL process establishes the allowable loadings of pollutants for a waterbody based on the relationship between pollution sources and in-stream water quality conditions. By following the TMDL process, states can establish water quality based controls to reduce

pollution from both point and non-point sources to restore and maintain the quality of their water resources (EPA 1991). The Commonwealth of Virginia's 1997 Water Quality Monitoring, Information, and Restoration Act (WQMIRA) codifies the requirement for the development of TMDLs for impaired waters.

The paper describes the development of a bacteria TMDL for Piney Run which was listed as impaired on Virginia's 1998 303(d) Total Maximum Daily Load Priority List and Report and Virginia's 2002 303(d) Report on Impaired Waters.

WATERSHED DESCRIPTION AND IMPAIRED SEGMENT

Piney Run is located in Loudoun County in the Middle Potomac-Catoctin Basin (USGS Hydrologic Unit Code 02070008). The upper limit is the headwaters of Piney Run and it extends downstream to the confluence of Piney Run with the Potomac River. The watershed (Figure 1) is approximately 3 miles long and 6 miles wide having an area of approximately 15.2 square miles. The impaired segment, a 3.52 miles in length, is listed in 303 (d) due to a violation of Virginia's water quality standard for fecal coliform bacteria. The watershed is monitored at station 1APIA001.80.

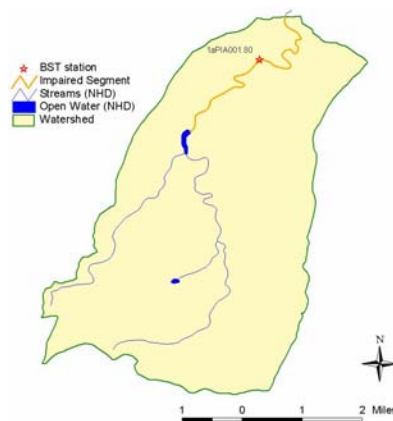


Figure 1. Map of the Piney Run watershed indicating monitoring station 1APIA001.80.

The watershed study area is predominately forest land (71.0 percent), with the majority of the remaining area in pasture land (26.9 percent). The remaining two percent consists of residential areas, crop land, wetlands, and open water. A land use map indicates that the pasture land tends to be located closer to the stream, while the forest land is farther from the stream.

WATER QUALITY PROBLEM IMPAIREMENT

Out of 19 samples collected during the 1998 assessment period, 5 samples exceeded the water quality standard for fecal coliform. During the most recent 2002 assessment period, 5 of 22 samples exceeded the fecal coliform standard. The seasonal distribution of fecal coliform data is presented in Figure 2. It also displays monthly exceedances (instantaneous fecal water quality standard - 1000 cfu/100mL).

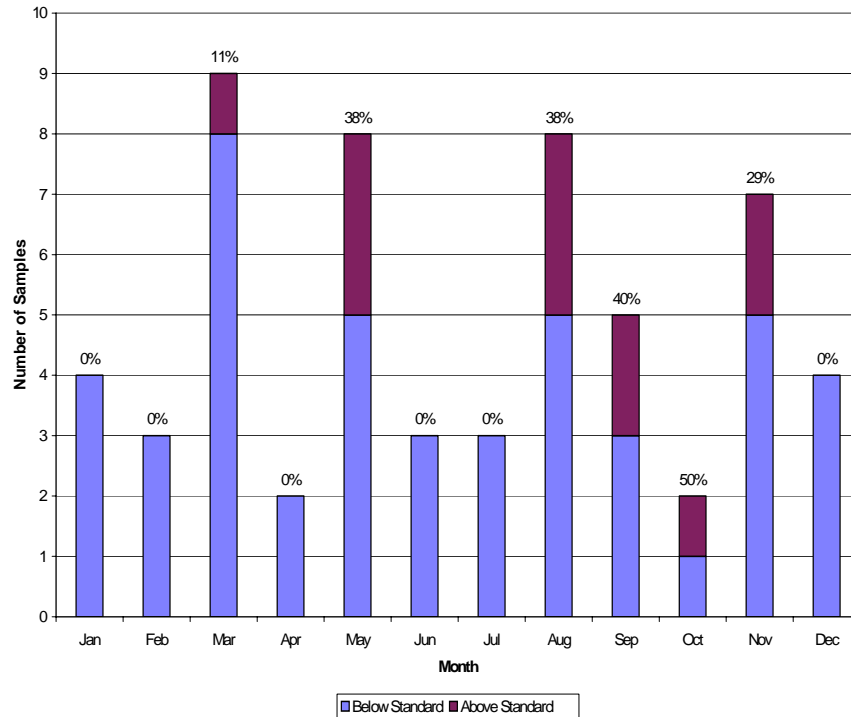


Figure 2. Seasonal distribution of fecal coliform samples and violations.

In a load-duration bacteria TMDL, source assessment is accomplished by determining the relative contribution of various sources towards the fecal bacteria present in the stream water. This method of source identification was achieved through Bacteria Source Tracking. To support BST analyses, bacteria loading in a watershed was estimated. These load estimates are broken into point and non-point sources. The non-point source load estimates presented here are the loading to the soil surface of the watershed; not the estimates of in-stream loads.

BACTERIA SOURCE TRACKING (BST)

A total of 12 ambient water quality samples were collected at the monitoring station and were analyzed by MapTech, Inc. for BST analysis. The analyses determined the relative contribution of overall bacteria by human, pet, livestock, and wildlife sources. Fecal and *E. coli* bacteria were also enumerated as part of the analyses. The BST data indicated that approximately 69% of the bacteria found in the Piney Run comes from anthropogenic (6% human, 22% pet, and 41% livestock) sources, and the remaining 31% from the wildlife sources.

TMDL DEVELOPMENT

In an effort to complete bacteria TMDLs in a timely and cost-effective manner, the use of load-duration analyses has been investigated. It has been determined that the load-duration method of calculating a TMDL produces results only slightly more conservative than if the TMDL had been determined through computer modeling.

The load duration method employs an entire stream flow record to provide insight into the flow conditions under which exceedances of the water quality standard occur. Exceedances that occur under low flow conditions are generally attributed to loads delivered directly to the stream such as straight pipes and livestock with access to the stream. Exceedances that occur under high flow conditions are typically attributed to loads that are delivered to the stream in stormwater runoff. Exceedances occurring under during normal flows can be attributed to a combination of runoff and direct deposits.

FLOW AND LOAD-DURATION CURVES

In order to develop a flow-duration curve, a long-term record of flow data for the target stream is needed. Where very little flow data exists for a target stream, a reference stream with having long flow measurements must be used. In present study, the USGS and Loudoun County began operating a stream gage that is co-located with the DEQ water quality monitoring station in 2001. The USGS stream gage 01636690 has one year of published data, from October 1, 2001 to September 30, 2002. Daily provisional flow data are available from October 1, 2002 until December 2, 2003. Daily average flow measurements were available. In order to extend the period of flow record to span the 1998 and 2002 assessment periods, the Piney Run flows were correlated with flows on Catoctin, Goose and Passage Creeks. Piney Run correlated best with Catoctin Creek, and the regression was developed and used to extend the flow record from 1988 to the present.

In order to use the load-duration method to develop a TMDL, a flow-duration curve must be developed for the impaired stream. This is accomplished by first developing a flow-duration curve for the reference stream. To plot the flow values for the period of record of the reference stream, the PERCENTILE statistic function was used. The resulting percentile of a given flow was then subtracted from 1 to yield the percent of time that a given flow is exceeded by the flows of record. The flow duration interval values thus obtained were plotted with the corresponding flows to yield a log/normal flow duration curve.

A load-duration curve was developed by multiplying each flow level along the flow-duration curve by the applicable water quality standard (235 cfu/100 mL). The resulting curve represents the maximum allowable load at each flow level, the TMDL. Since the TMDL and required reductions must be in terms of an average annual stream flow, the loads on the load-duration curve were converted to annual loads. In order to plot existing fecal coliform (FC) data against the *E. coli* (EC) standard/TMDL line, it was necessary to translate the FC data to EC data. It was done by using a translator equation, developed from a regression analysis of 493 paired FC/EC data sets from the DEQ's statewide monitoring network.

Figure 2 shows water quality data and the exceedances of the water quality standard occur under high and normal flow conditions. The highest exceedance of the water quality standard (circled) occurs at a high flow that has been exceeded approximately 9% of the time (~26 cfs). This represents the flow condition under which the largest bacteria reduction is required to attain water quality standards. The translated load at this flow condition is 8.79×10^{14} cfu/yr. Under the instantaneous *E. coli* standard of 235 cfu/100mL, this load would have to be reduced by 94% to an allowable load of 5.41×10^{13} cfu/yr. In order to determine the necessary load reduction at

the average annual flow condition, a second curve must be drawn through the highest exceedance described above. The second curve represents the magnitude of the highest observed exceedance if it were to occur over any flow condition (Figure 3).

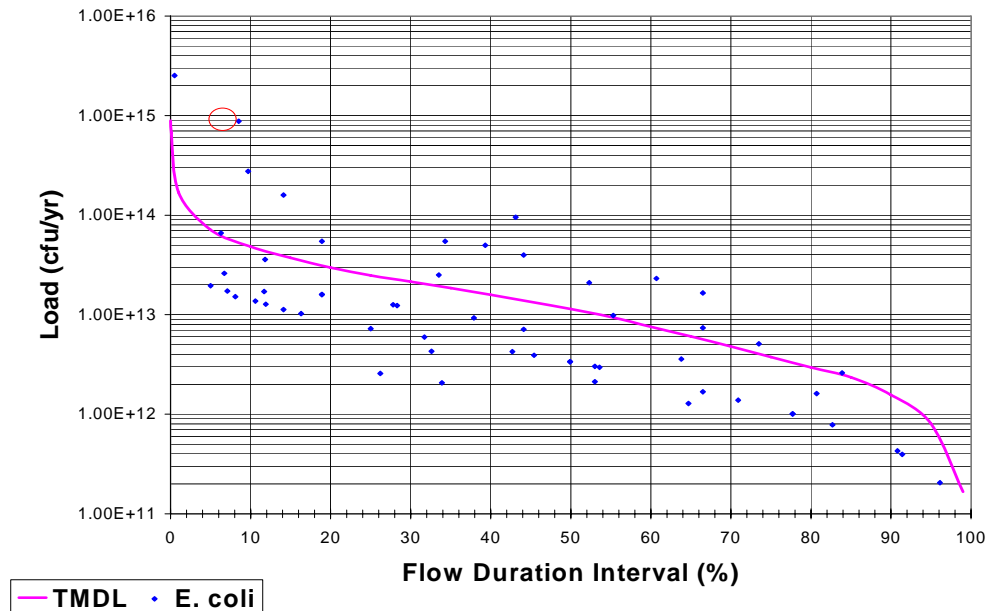


Figure 3. Load duration curve with observed data and maximum exceedance curve.

TOTAL MAXIMUM DAILY LOAD

A Total Maximum Daily Load (TMDL) consists of (1) point source/waste load allocations (WLAs), (2) non-point sources/load allocations (LAs) where the non-point sources include natural/background levels, and (3) a margin of safety (MOS) - implicit or explicit. In present analysis, an implicit MOS was incorporated through the use of conservative analytical assumptions. This includes the use of the single-most extreme observed water quality violation event used to develop the maximum exceedance curve over the entire range of flow conditions.

In the case of load-duration analysis, the TMDL is expressed as the total number of colony forming units (cfu) per year as opposed to cfu/day. This is because the load-duration TMDL must be based on the average annual flow condition. The average annual flow for the Piney Run is calculated from the average annual flow from the USGS stream gage. The estimated average annual flow for Piney Run is 10.49 cfs, associated with a flow duration of 28.8%. From this, an average annual *E. coli* load and TMDL were calculated from the max-exceedance and TMDL curves and are shown in Figure 4.

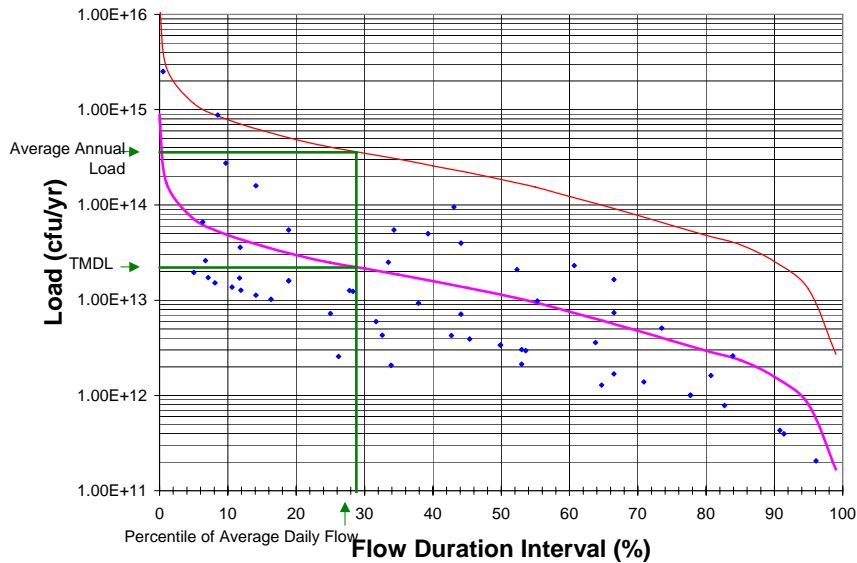


Figure 4. Load duration curve having TMDL and estimated average annual *E. coli* load for Piney Run at station 1APIA001.80.

POINT SOURCES

Two bacteria point source discharges were identified in the watershed. Both point sources are covered under Virginia Pollution Discharge Elimination System (VPDES) general permits program for sewage discharge for having less than 1,000 gallons per day. The permitted point sources are presented in Table 1. Permitted loads were calculated by multiplying the permitted discharge concentration (126 cfu/100 ml) times the design flow, resulting 3.48×10^9 cfu/yr for both discharges.

LOAD ALLOCATIONS

The average annual *E. coli* load is 3.58×10^{14} cfu/yr, and the TMDL under average annual flow conditions is 2.20×10^{13} cfu/yr. These values are used to calculate required reductions. By subtracting the waste load allocation (known value) from the TMDL (as determined above), the load allocation can be determined. These values are given below in first column of Table 1.

The annual average TMDL and *E. coli* load values together with the waste load allocation (WLA) from the permitted bacteria sources were used to determine the required reduction (Table 1). Since the required reduction will only apply to the non-point sources, the LA value was used to calculate the required percent reduction.

Table 1. TMDL and required reduction for Piney Run.

Load Category (annual average)	Allowable Loads (cfu/yr)	Average Annual EC Load (cfu/yr)	Required Reduction
WLA	3.48×10^9	3.48×10^9	0%
LA	2.20×10^{13}	3.58×10^{14}	94%
MOS	0 (implicit)		
TMDL	2.20×10^{13}	3.58×10^{14}	94%

As shown in Table 1, the WLA represents less than 0.02% of the TMDL load, and virtually has no effect on the LA reductions. To apply above reductions, the average annual *E. coli* load had to be allocated for all BST identified all non-point sources. Table 2 shows the allocated loads, reduction applied, and the allowable loadings for each source.

Table 2. Average annual load distribution, reduction, and allowable load by source.

	Total (cfu/yr)	Human: 6% (cfu/yr)	Pet: 22% (cfu/yr)	Livestock: 41% (cfu/yr)	Wildlife: 31% (cfu/yr)
Average Annual Load	3.58×10^{14}	2.27×10^{13}	7.75×10^{13}	1.45×10^{14}	1.12×10^{14}
Reduction	94%	94%	94%	94%	94%
Allowable Annual Load	2.20×10^{13}	1.39×10^{12}	4.77×10^{12}	8.94×10^{12}	6.92×10^{12}

STAGE I IMPLEMENTATION PLAN

As stated in above section, the TMDL requires a 94% reduction in non-point source loading to attain a 0% violation of water quality standards. This can be achieved through a phased process. In order to evaluate interim reduction goals for a phased plan, several reduction levels and their associated violation rates were assessed. Reduction curves similar to the max exceedance/reduction curve of Figure 4 were plotted on the load-duration curve (Figure 5). The violation rates for 94%, 70%, 50%, and 25% load reductions levels were found to be 0%, 9%, 19% and 22%, respectively. Based on this analysis, a suitable Phase I reduction level would be 70%. Table 3 presents the load allocations for this reduction level. These load reductions will be used in developing the implementation plans and executing them on phased basis. VADEQ will continue to monitor Piney Run in accordance with its ambient monitoring program. VADEQ and VADCR will use the data to evaluate reductions in bacteria counts and the effectiveness of the TMDL in attainment of water quality standards.

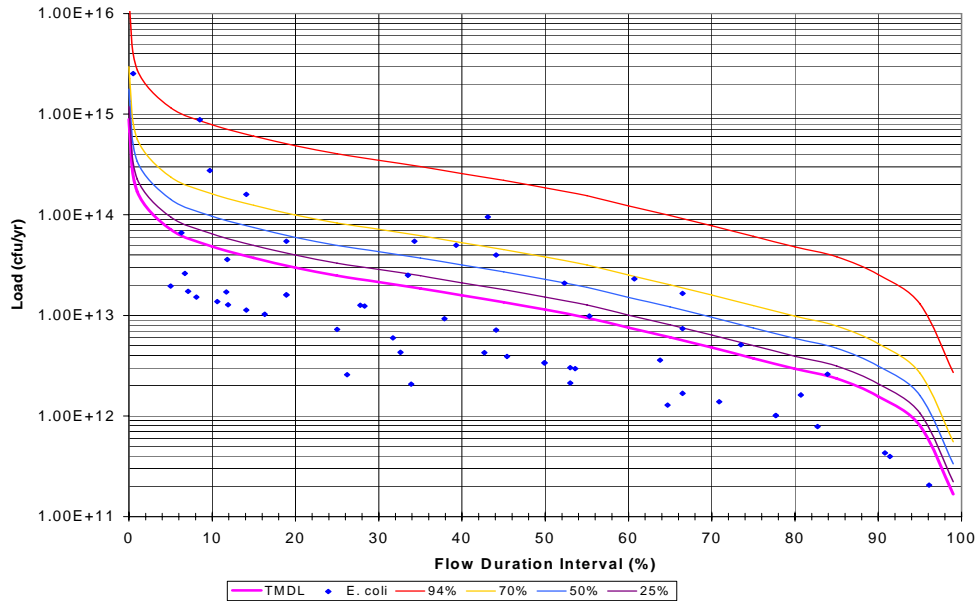


Figure 5. Load duration curve with TMDL and reduction curves for Piney Run.

Table 3. Phase I load allocations (based on a 70% reduction).

	Total (cfu/yr)	Human (cfu/yr)	Pet (cfu/yr)	Livestock (cfu/yr)	Wildlife (cfu/yr)
Average Annual Load	3.58×10^{14}	2.27×10^{13}	7.75×10^{13}	1.45×10^{14}	1.12×10^{14}
Reduction	70%	94%	94%	94%	18%
Target Annual Load	1.07×10^{14}	1.39×10^{12}	4.77×10^{12}	8.94×10^{12}	9.22×10^{13}

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BACTERIAL SOURCE TRACKING IN VIRGINIA'S PUBLIC BEACHES

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KEYWORDS: bacterial source tracking, antibiotic resistance analysis, fluorometry, *Enterococcus*, public beaches

ABSTRACT

There are 17 public beaches in Virginia, which provide over 38 km of waterfront access. All beaches are currently required to perform water quality monitoring during the swimming season. Weekly water samples are collected by local health departments beginning in mid-May and ending in September. Collected samples are filtered and/or analyzed for the presence of a fecal indicator organism, *Enterococcus*. Monitoring done during the summer of 2003 showed higher than normal counts of enterococci prompting numerous beach advisories. In the interest of public health, attempts to lower these enterococci counts are desired.

Bacterial source tracking (BST) provides a method to identify the sources of fecal pollution, allowing optimal effectiveness in remediation efforts. Antibiotic Resistance Analysis (ARA), a biochemical test, was performed to determine the source of fecal contaminants on all Virginia beaches, and particular focus was given to beaches that experienced one or more exceedances during the summer of 2004. In addition, flurometry was used on water samples to detect optical brighteners present in water samples. Optical brighteners are found in laundry and dishwashing detergents, and indicate the presence of human wastewater. Although fluorometry in open bodies of water has proven ineffective, testing of waters collected from storm drains may help detect human signatures from broken or leaking sewer lines.

Water samples were collected along coastal, Chesapeake Bay, and river beaches in conjunction with the Virginia Department of Health. Sampling sites focused on designated swimming areas along beachfronts and locations surrounding storm drain outfalls. Beach postings due to high fecal bacterial counts are almost always associated with precipitation and runoff events and support the contention that contamination is transported to the beach via storm drains. The sources of fecal bacteria during these events are varied and were divided into categories for birds, wildlife, pets, and humans. Sampling over the summer of 2004 has identified problematic beaches and the major sources of pollution so that remediation efforts can be directed at identified sources of contamination.

INTRODUCTION

In 1986, The United States Environmental Protection Agency (EPA) recommended the use of enteric bacteria *Escherichia coli* (*E. coli*)(freshwater) and enterococci (marine) in recreational

waters as pathogenic indicator organisms. Pathogenic indicators may or may not cause disease, but are consistently linked to the detection of pathogens. The *Ambient Water Quality Criteria for Bacteria – 1986* used the results of several studies (USEPA 1983, USEPA 1984) to determine that *E. coli* and enterococci were more reliable predictors of the presence pathogens in the water than the previously used fecal coliforms.

In October 2000, passage of the federal BEACH (Beaches Environmental Assessment and Coastal Health) Act mandated that coastal and Great Lakes states adopt the standards of the EPA's *Ambient Water Quality Criteria for Bacteria - 1986*, or standards that offered equal protection to human health, by April 2004. The BEACH Act requires standards be set for pathogenic, or pathogenic indicator organisms and mandates the creation and implementation of regular beach monitoring, and subsequent notification of the public when standards are exceeded. The Virginia Department of Health (VDH) began a state-wide, weekly monitoring of public beaches in 2001 conducted by local health departments. Currently all but one of Virginia's public beaches undergo weekly testing of bacterial levels from mid-May through September.

Virginia uses the indicator organism *Enterococcus* for all public beach waters. *Enterococcus* is the preferred indicator organism because it is found in the fecal matter of all warm-blooded animals, and exhibits a longer survival time than *E. coli* (a freshwater indicator organism) in marine waters (EPA 1984). Standards for enterococci are: a geometric mean of 35 for two or more samples taken in a calendar month, or a single sample maximum of 104 colony forming units per milliliter (CFU/mL). A sample found that exceed this standard requires local health departments to post notice of a swimming-advisory and alert the media concerning the beach from which the sample was obtained.

Of particular concern to health departments is water contaminated with human fecal materials. Ingestion of water carrying a human signature can cause gastroenteritis or a variety of ear, nose and throat diseases caused by fecal pathogens and viruses. Viruses are considered responsible the majority of illness to swimmers, although bacteria and protozoa also contribute to swimming-related illnesses (NRDC 2004). Knowledge of the source of fecal material in the water can aid in the assessment of risk to swimmer's health.

In an attempt to locate the source of fecal contaminants (enterococci) ARA was applied to all beaches beginning in June of 2004. ARA compares the antibiotic resistance patterns of unknown isolates to the resistance patterns of isolates whose origins are known. Particular focus was given to those beaches that exceeded state water quality standards numerous times during the course of the summer. A multiwatershed known-source library was constructed to account for geographic variations across Virginia's coastal region.

In addition to ARA, all water samples were analyzed for the presence of optical brighteners. Optical brighteners are organic compounds, used in household detergents that absorb ultraviolet light and reemit it at a longer wavelength. Using a fluorometer, these compounds can be detected in water at measurable concentrations. Detection of these compounds at high concentrations may indicate the presence of human wastewater in a given sample or sample area.

MATERIALS AND METHODS

Study Area

This study was conducted at the Virginia public beaches currently being monitored by the Virginia Department of Health. These beaches include those bordering the Atlantic Ocean (Virginia Beach City and Eastern Shore), beaches bordering the Chesapeake Bay (Virginia Beach City, Norfolk, Hampton, and Eastern Shore), and beaches bordering the James River (Newport News), the York River (Newport News and Gloucester) and the Rappahannock River (Fairview). Table 1 lists all beaches and number of sampling locations monitored during the summer of 2004.

Table 1. Monitored beaches and number of sampling points.

District	Beach/Location	Sampling Points
Rappahannock	Fairview Beach	4
Newport News	Hilton Beach	1
Newport News	Huntington Beach	3
Newport News	Anderson's	1
Newport News	King/Lincoln Park	1
Newport News	Yorktown Beach	1
Norfolk	Norfolk	9
Hampton	Grandview Pier	1
Hampton	Buckroe Pier	2
Hampton	Salt Ponds	1
Hampton	Mill Point	1
Virginia Beach	Virginia Beach	24
Eastern Shore	Assateague Island	4
Eastern Shore	Kiptopeke State Park	2
Eastern Shore	Cape Charles Town	4
Eastern Shore	Guard Shore	2
Three Rivers	Gloucester	1

Sample Collection

Samples used for enumerations were collected in sterilized plastic bottles by local health departments in accordance with the DRAFT - VDH Beach Monitoring Protocol July 2004. All samples with the exception of Virginia Beach were collected in the surf or swimming area no more than 20 meters from the shore. Health Department workers, aboard a police boat, collected Virginia Beach samples 100-200 meters offshore. Numerous samples were collected outside the normal sampling points, focusing primarily around storm drains, which emptied near, or into potential swimming areas. Samples collected specifically for this study were kept at 4°C for no more than 28 hours before beginning analyses. This holding time is longer than the maximum allowed time of 6 hours (VDH 2004) to use enumerations, however acceptable by TMDL (USEPA 2000) and comparable counts have been recorded (Pope 2003).

Isolation of Enterococci

Samples analyzed by all local Health Departments, except the Eastern Shore, used the Enterolert (IDEXX Laboratories Inc., Westbrook, Maine) to report enterococci counts in colony-forming units (CFU) per 100 milliliters. The Enterolert test shortens the time required to obtain results, from 48 to 24 hours, as compared to membrane-filtration and growth on a selective agar, with no statistically significant differences (Budnick 1996). The Eastern Shore Health Department uses membrane filtration and incubation on brain heart infusion agar (BHI) (Difco) to obtain enterococci counts. Water samples found to exceed the state standards were mailed overnight-express, in refrigerated packages, to Virginia Tech. All samples arriving at Virginia Tech were immediately filtered through 0.45- μ m pore size membrane filters and grown on m-Enterococcus agar (BBL, Baltimore, MD) for 48 hours at 35°C to obtain Enterococcus isolates. After incubation, individual red to burgundy colonies were picked using sterilize toothpicks and grown in 96-well microwell plates as described in Hagedorn *et al.* 1999. During the summer of 2004, approximately 150 water samples were collected and filtered, and 841 isolates were analyzed by ARA.

Antibiotic Resistance Analysis

ARA was performed essentially as described in Hagedorn *et al.* 1999. Thirty concentrations of nine different antibiotics were used, and isolates were evaluated as growth or no growth. Results for each isolate were analyzed against a known-source library using discriminate analysis (Wiggins 1996).

Watershed and Library Composition

This project required a diverse library of known sources to account for potential variations in wildlife, bird, and human populations along Virginia's river, bay, and oceanfront beaches. Therefore, libraries from five different areas were combined to produce one large 1107 isolate multi-watershed library. Isolates for this library were collected in the Virginia coastal regions of the Potomac River, the Lynhaven River, the York River, Virginia Beach, and North Carolina's Outer Banks. Multiwatershed libraries have proven effective for extended geographic areas (Wiggins 2003). The library was divided into 4 categories (Birds, Humans, Pets, and Wildlife) and produced an Average Rate of Correction Classification of 93.0%.

Fluorometry

A Turner Designs 10AU Fluorometer was used to detect optical brighteners in water samples. The fluorometer was calibrated with known standards using the optical brightener FB-28 at concentrations ranging from 15 to 500 milligrams per liter (mg/L). A linear relationship was established with the known concentrations prior to taking measurements on all water samples. The basic sensitivity of the instrument was set at 45% using a 250 ppb standard of FB-28. Blanking of the fluorometer was done using de-ionized water and the standard solution concentration was set using a 125 ppb standard of FB-28. All other levels were left at the default settings. Water samples were analyzed within 24 hours of arrival at room temperature.

RESULTS

Numerous beaches proved to be problematic over the summer of 2004. A strong correlation was discovered between the presence of visible storm drains, which emptied into or around

swimming areas, and the number of days a beach spend under an advisory. All beaches that spent at least one day under a swimming-advisory are displayed in Figure 1 and those beaches with visible storm drains are denoted with an asterisk. Only Fairview Beach missed a significant number of days with no visible storm drains on the beach. However, a broken storm drainpipe was observed leaking in a sinkhole just off the main beach. It is not unreasonable to assume that water from this storm drain could have infiltrated into the soil in flown into the Rappahannock River at Fairview Beach, increasing enterococci counts.

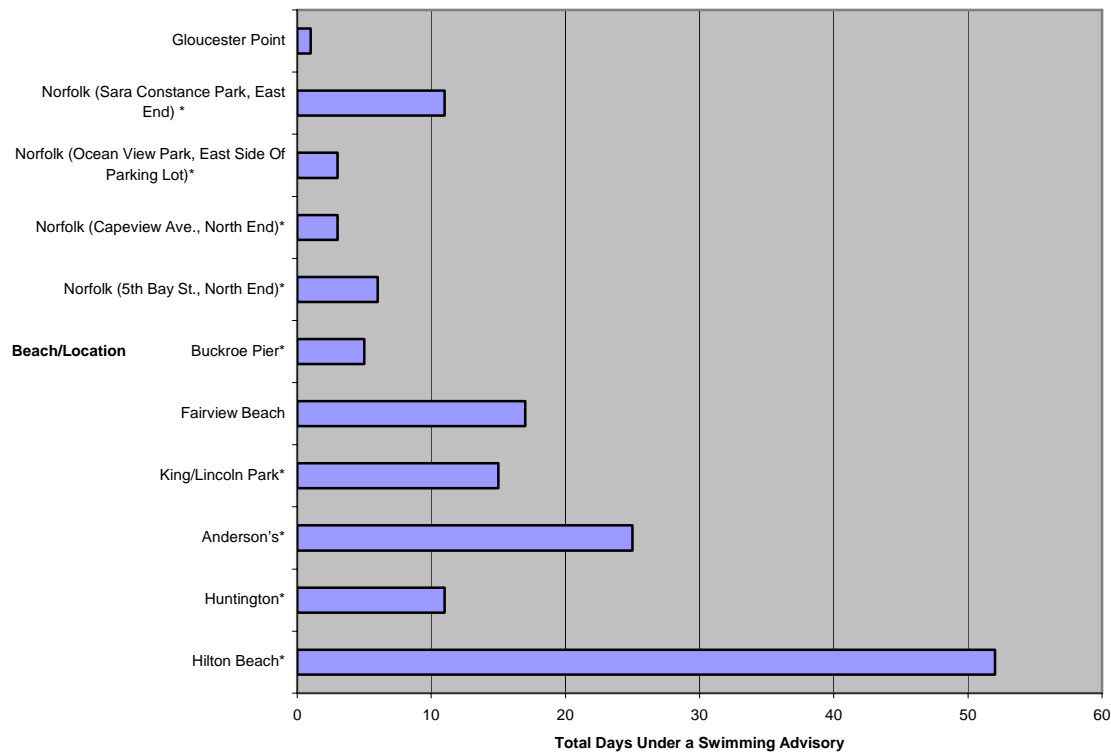


Figure 1. Total number of days problematic beaches were under an advisory from 5/24/04-8/31/04 (Beaches with visible storm drains are denoted with an asterisk.).

Storm drains have been linked to increased rates of illness in Santa Monica Bay, CA (NRDC 2003). A 1995 study conducted by the Santa Monica Bay Restoration Project found that people swimming within 400 yards of storm drain outfalls had a 57 percent greater chance of becoming ill than those swimming farther away. The increased illness can be attributed to sewage-polluted water as the result of urban runoff.

Increased human waste increases the risk of swimmers becoming ill as a result of pathogenic bacteria or viruses. Figure 2 shows the number of isolates from problematic beaches where significant isolates were analyzed and placed into four different source categories. Also included in Figure 2 are an analysis of the potential modes of transport of fecal contamination into Hilton and Fairview Beach. Hilton SW contains isolates from a storm drain that empties onto the

eastern side of Hilton Beach. Fairview Sinkhole is a located at 8th Street where an exposed, corroded storm drain has broken and allowed storm runoff to leak into a sinkhole just off the beach. As human waste is of primary concern Figure 3 illustrates the percentage of human isolates found in the beaches of Figure 2.

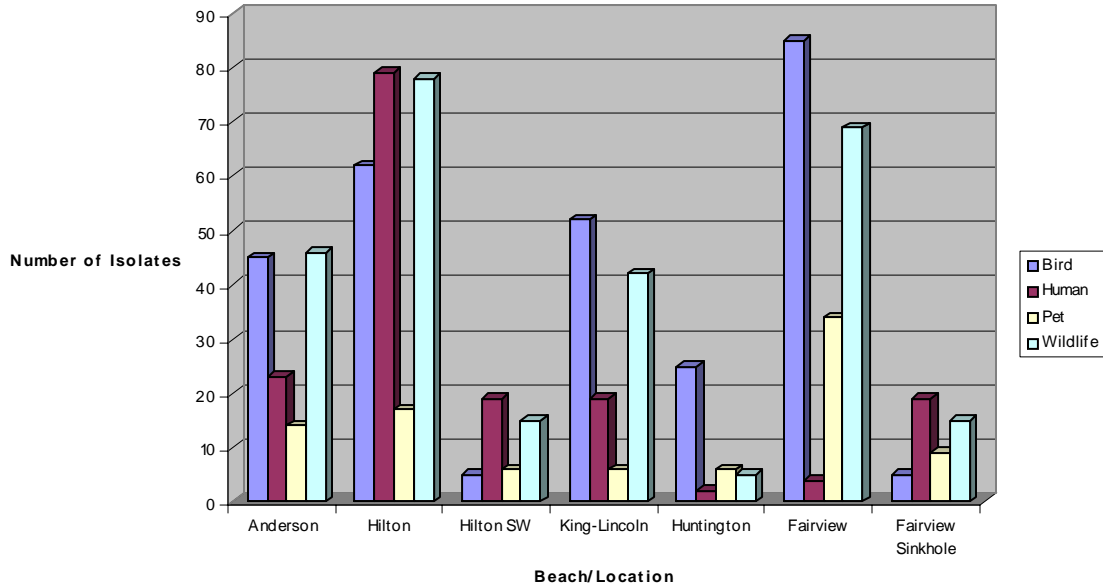


Figure 2. Number of isolates attributed to bird, human, pet, and wildlife sources from problematic areas.

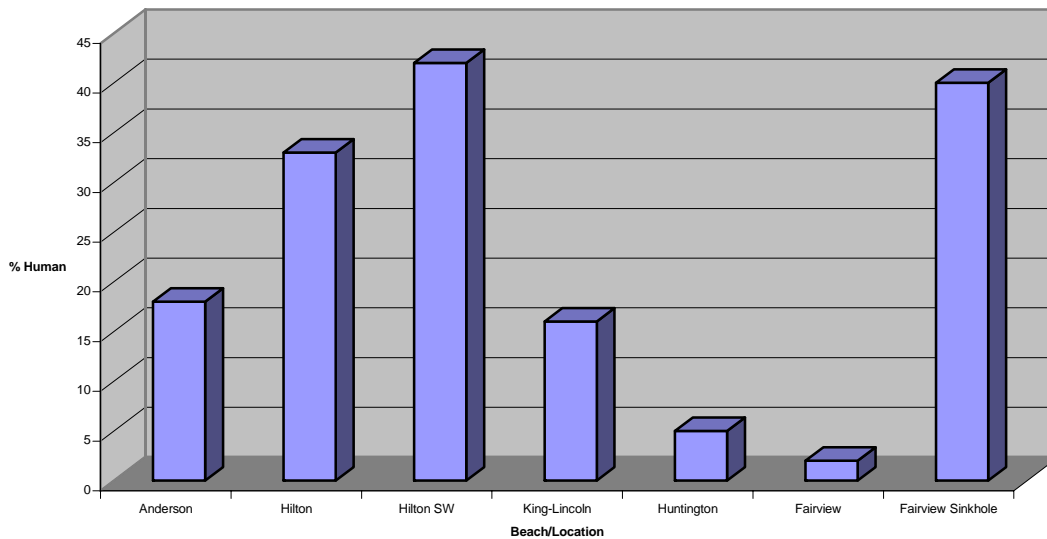


Figure 3. Percentage of human isolates detected in beaches from Figure 2.

The detection of optical brighteners in beach water proved to be ineffective, as concentrations became so dilute in open-ocean, bay, or river water, that potentially elevated levels were

indistinguishable from background levels. The variation of background levels typically ranged from 12 mg/L to 50 mg/L, although readings of 80-90 mg/L were not uncommon. However, elevated readings were found in runoff water of storm drains containing high levels of human *Enterococcus*. Table 2 shows concentrations of optical brighteners in mg/L compared to the percentage of isolates that were identified as human in flowing storm drains and at the regular sample sites swimming areas. Levels above 100 mg/L are suspected to be above of the normal background range in beach waters, and thus may indicate a mixing with human wastewater. Both Hilton Storm Drain samples and on the Fairview Sinkhole samples had reading in the high range and a high percentage of human isolates. Elevated levels discovered in Huntington Storm Water may have been the result of aromatic compounds such as diesel fuel, which can influence fluorometer readings (Turner Designs 1999).

Table 2. Comparison of optical brighteners (mg/L) to the percentage of human isolates found at Hilton, Fairview, and Huntington Beaches.

Beach/Location	Date	Optical Brighteners (ppb)	% Human
Hilton Storm Drain	6/24/2004	170	33
Hilton Beach	6/24/2004	36.9	21
Hilton Storm Drain	8/2/2004	109	33
Hilton Beach	8/2/2004	45.1	10
Fairview	6/15/2004	42.6	0
Fairview Sinkhole	6/15/2004	78.7	37.5
Fairview	6/23/2004	87.1	0
Fairview Sinkhole	6/24/2004	171	42
Huntington Storm Drain	6/24/2004	196	8

CONCLUSIONS

Many of Virginia's Public Beaches have experienced no problems during the summer of 2004 and show no signs of developing into problematic beaches. These beaches include Virginia Beach, the beaches of the Eastern Shore (Table 1), and Hampton Beaches, with the exception of Buckroe Pier. Several beaches have shown only minor problems but should continue to be monitored closely in the future. These beaches include Gloucester Beach, Fairview Beach, and Norfolk Beaches. The beaches of Newport News (excluding Yorktown) will likely continue to have severe problems with fecal contamination in the future, and will require remediation and possible elimination of storm drain runoff to comply with state standards and swimmer safety. Use of fluorometry in open beach water may not be useful for detecting a definitive human signature. However, detection of optical brighteners in storm runoff may still provide an effective means of assessing sewer leakage into swimming areas. Continued testing of the effectiveness of fluorometry in storm runoff will be necessary in future projects.

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TOWARD ESTABLISHING WATER RESOURCES POLICY FOR SHENANDOAH VALLEY WATERSHEDS

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KEY WORDS: water resources, goals, process, planning, regional, community

ABSTRACT

In October 2002, elected officials from the Shenandoah Valley met as the Regional Water Resources Policy Committee (RWRPC) to begin a broad dialogue among local governments about common water issues facing the region. The RWRPC began its work recognizing the need for a comprehensive approach to water resources management and built its mission statement around development of a Shenandoah Valley Water Resource Comprehensive Plan. They sought a planning process that was: (1) goal driven and involving a broad range of community stakeholders; (2) based on a sound understanding of available resources; and, (3) integrated with state and federal planning requirements in order to optimize limited financial resources.

As the first step, the RWRPC conducted a Watersheds Policy Integration Assessment (Assessment) to establish a blue-print for the development of the Comprehensive Plan and to begin the process of aligning water resources management efforts with goals and priorities. The process designed for the Assessment consisted of: (1) an extensive stakeholder input process to identify regional water resources goals as well as strengths, weaknesses, opportunities, and threats; and, (2) a data collection process, including information on agencies and organizations involved in water resources management, the existing programmatic and regulatory framework, and best practices/model programs.

In this paper, we discuss how the strategy was put into practice to arrive at a consensus on the goals and how a database structure was used to match goals with existing information and

organizational resources and to identify resource gaps. At this time, the database is still being populated, so the work is very much an effort in progress. The Assessment does, however, represent an excellent example of what regional cooperation and coordination can achieve.

INTRODUCTION

A 1999 forum, “Regional Water Relationships and the Future of the Northern Shenandoah Valley” resulted in the counties of Clarke, Frederick, Page, Shenandoah, and Warren, and the City of Winchester, agreeing on the need to engage in water resources planning over a 50 year planning horizon. At the time, severe drought conditions highlighted water supply as a common regional concern. It was also proposed that a regional authority might be a potential solution. As a result, an ad hoc committee was formed, and a strategic-level water supply planning process was initiated. The water supply study showed population demands outstripping water supply in the North Fork by 2025. Work ended in 2002 with a proposal for a hybrid authority that would conduct planning, much like a water basin commission, as well as implement solutions. However, while there was broad agreement on the need for planning, there was hesitation about the creation of a regional authority. Therefore, in response to the planning need, the Northern Shenandoah Valley Regional Commission created a Regional Water Resources Policy Committee (RWRPC), with the goal of creating a Water Resources Comprehensive Plan. The counties of Clarke, Frederick, Page, Shenandoah, and Warren, the City of Winchester, and the Town of Front Royal each appointed an elected official to serve on the RWRPC. When the group first met on October 31, 2002, the members recognized that the effort could not be truly comprehensive without the participation of upstream and downstream jurisdictions. Invitations were issued, and Augusta County and Jefferson County, West Virginia were represented at the next meeting. Also joining the RWRPC were Rockingham County, the City of Staunton, and Berkeley County, West Virginia.

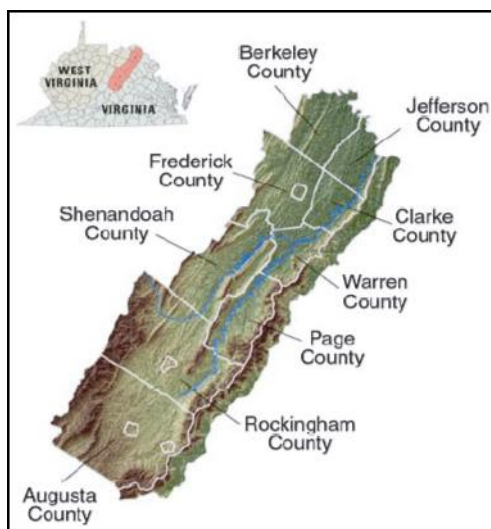


Figure 1. Shenandoah Valley watersheds planning area.

In June 2003, the RWRPC received a staff report: “The Shenandoah Valley Watersheds – Water Resources Comprehensive Plan White Paper” which stated, “To be action oriented and

comprehensive over a wide range of issues, each having many stakeholders and plans, a strategic approach is recommended. The plan format has three main components:

1. Goals: Goals will be established based on local and regional concerns.
2. Current Situation: Current situations may be established from existing studies or, when lacking, studies may be initiated by the RWRPC or such studies endorsed by the Committee.
3. Policies and Actions: Strategies in response to dealing with the gap between the goals and current situation are likely to be a mix of policies which lead to actions.”

Through a grant of \$25,000 from the Agua Fund arranged by stakeholders active in the process, the RWRPC developed an RFP for a Watersheds Policy Integration Assessment. Based on a competitive process, the RWRPC engaged the services of AMEC Earth & Environmental, Inc. The draft Assessment was presented to the RWRPC on August 5, 2004 and will be received in final form on October 7, 2004. The RWRPC, its stakeholder involvement process, and the Assessment represent a new “regional community” model for expanding cooperation and increasing coordination for regional resources like water – in a multi-community, multi-jurisdictional, multi-regional, and multi-state environment region like the Shenandoah Valley.

WATERSHEDS POLICY INTEGRATION ASSESSMENT

The Assessment was initiated by the RWRPC to help define a path for meeting its goal of developing the Water Resources Comprehensive Plan. Key to defining this path was recognition of the importance of developing a regional consensus on the goals that would serve as the foundation for the Comprehensive Plan. Once these goals were developed, it would then be possible to gain an understanding of potential hurdles and opportunities, as well as to create the tools needed for regional leaders to begin aligning management efforts with common goals. Based on these needs, the Assessment was designed to result in three key outcomes: (1) a regional water resources goals framework; (2) an assessment of how existing information sources and activities support the goals; and, (3) a proposed Comprehensive Plan outline.

Three major work elements were conducted to support the Assessment outcomes. These included two focus groups, a stakeholder survey, and the development and population of a water resources database. The two focus groups resulted in an initial set of water resources goals that served as the basis for a larger discussion and ultimately resulted in the consensus goal statements approved by the RWRPC. The survey served several functions. First, it served as a means of checking the validity of the work of the focus groups in developing the goal statements. Second, it resulted in valuable qualitative information about why stakeholders become involved in water resources issues, what is working (and not working) with regard to water resources management, and what opportunities are available to further water resources protection.

The water resources database was designed to organize the large amount of information collected as a result of the project and to allow the information to be sorted and analyzed by long-term goal. However, the database was also designed to take into consideration its potential to serve as a regional information sharing and network building tool.

REGIONAL WATER RESOURCES GOALS

Although goal setting is nothing new in the Shenandoah Valley, the development of water resources goals that apply both locally and regionally represents a significant accomplishment. The purpose of devoting time and energy to creating these goals is to help decision-makers better focus energy and limited resources as well as to ensure that those responsible for the region's water resources (including government and non-government entities) are not working towards conflicting goals. Identifying long-term goals makes it possible to quickly adjust regional efforts in response to a rapidly changing environment because the ultimate direction that the region wishes to go has already been defined. In designing a process to develop regional water resources goals, the primary challenge was how to create a sense of ownership by local elected officials as well as among the region's many active stakeholder groups. The process was further complicated by the multi-state nature of the effort, which meant dealing with different regulatory drivers and political climates. For instance, legislation adopted by the Virginia General Assembly in 2003 will require local and/or regional water supply planning in Virginia, while there is not yet such a correlation in West Virginia.

A multi-step process was selected for the development of the water resources goals, which began with the use of facilitated focus groups in March 2004. Two stakeholder focus groups were conducted, one at James Madison University in Harrisonburg and one at the Opequon Water Reclamation Facility in Winchester. The advantage of the focus group approach is that it allows for in-depth discussion of specific questions and allows the facilitator to create a sense of group ownership of the final product. However, this advantage is tempered by the inherent limit on the number of participants – which can create a sense of exclusivity if the process is not well designed. The solution chosen to overcome this issue was to allow potential focus group participants to self-nominate, and to then allow the study team to choose the final focus groups based on geographic representation and representation from a range of interests and expertise. Several means of communication were used to advertise the focus groups, including a press release and the Internet. A total of 23 participants were chosen for the two sessions.

Each focus group began with an initial brainstorming session, where participants were asked the question “As water resource users, what should be our goals for water resources in the Shenandoah Valley?” Each participant was allowed to provide one answer in round-robin fashion until everyone had a chance to speak. The process was repeated until no more goals could be identified. The next step in the process was a SWOT (strengths, weaknesses, opportunities, and threats) exercise. The purpose of the SWOT exercise was to enable focus group participants to clearly see where the region stands now – in other words, the region's *situation*. Then, the responses to the SWOT exercise were ranked by the focus group participants. The information and rankings were used to validate or modify the goals identified during the initial brainstorming process.

The focus group process resulted in over 40 issues and ideas – far too many to serve as an effective directional tool for the Comprehensive Plan. In addition, many of the identified “goals” were actually recommended “strategies” or “actions.” As a result, the issues and ideas were first sorted into 12 categories, and then criteria were adopted to tease goal statements out of the issues and ideas. These criteria included that the statement needed to be value based, take a long view,

be directional in nature, allow the concentration of resources, and support translation to strategies and actions. Based on this process, draft goal statements were presented to the North Fork Minimum Instream Flow Technical Committee (MIF) on March 17, 2004 and at a meeting of local elected officials on March 30, 2004. The MIF acts as a technical advisory group to the RWRPC. A revised set of goals was presented to the RWRPC on April 1, 2004 and released to local governments for review and comment.

The goals adopted by the RWRPC on May 6, 2004 are presented in matrix format (Table 1) and are divided into “primary goals” and “supporting goals.” While all of the goals are inter-related and support the larger concept of “Take Care of the Water” (adopted as the overarching goal statement), the primary goals tend to be physical and measurable, while the supporting goals tend to be more thematic. For instance, while the supporting goal of “Data and Information” is an important goal in itself, it is thematic in that it is essential to support each of the primary goals.

Table 1. Shenandoah Valley water resources goals matrix.

PRIMARY GOALS					
Water Supply Sustainability	Water Quality	Natural Systems	Planning & Reg. Cooperation	Education & Stewardship	Recreational Access
<i>Ensure water supply and demand are kept in balance so that Valley residents, businesses, farms, and aquatic life all have the needed level of sustainable water (ground and surface).</i>	<i>Aggressively achieve the level of water quality (ground and surface) required to support the human, business, and agricultural needs in the Valley, without sacrificing the needs of the watershed’s fish and other aquatic life.</i>	<i>Protect and enhance the natural systems that are integral to water resources protection, including: karst geography, vegetative buffers, forests, and wetlands.</i>	<i>Achieve a broad regional consensus on the direction of water resources policy, planning, and management so that common goals can be achieved and solutions implemented more effectively and cost-efficiently.</i>	<i>Have well informed, conservation-minded citizens, business people, and elected officials who are actively involved in promoting water resources stewardship.</i>	<i>Ensure reasonable public access to the Valley’s water resources while respecting private property rights and the need to protect water quality.</i>
SUPPORTING GOALS					
Data and Information					
<i>Provide Valley leaders and citizens alike with accessible, reliable, and objective information and scientific data needed to support informed water resources decisions.</i>					
Financial Resources					
<i>Provide or obtain the financial resources needed to meet the Valley’s water resources goals, continuously prioritizing efforts to maximize the value of each available dollar.</i>					
Build on Existing Abilities and Relationships					
<i>Strengthen the Valley’s ability to address water resources issues by effectively using and adding to the skills of local, regional, state, and national resources.</i>					
Agricultural and Open Space Heritage					
<i>Enhance the Valley’s agricultural and open space heritage linkage to water resources stewardship.</i>					
Economic Advantage					
<i>Enhance the Valley’s economic advantage by protecting and wisely using water resources.</i>					
Standards and Regulations					
<i>Optimize standards and regulatory tools necessary to meet the Valley’s water resource protection and planning needs and consistently and equitably enforce these standards and regulations.</i>					

The development of common water resources goals that apply both locally and regionally represents a significant accomplishment by the RWRPC and establishes the key organizational driver behind the Comprehensive Plan. The goals were established by the stakeholders in a “bottom up” process that gives a solid basis to the both the Comprehensive Plan and the development of supporting tools for decision makers. One of these is the database discussed in the next section. Throughout the goal-setting process, the fundamental importance of water supply and water quality to the economic prosperity of the region was stressed.

REGIONAL WATER RESOURCES DATABASE

A primary deliverable of the Assessment was a regional water resources database. The database was created to organize resource information in a way that allows the RWRPC to link regional water resources goals with existing information and data. This allows the RWRPC to prioritize future planning and data gathering efforts. The database is organized around the 12 water resources goals and contains information on: (1) the agencies and groups working to achieve water goals; (2) data sources helpful to understanding the current situation; (3) existing activities and programs; and, (4) state and federal regulatory requirements. The work effort consisted of the development of a Microsoft Access[®] database table structure and data entry forms, as well as the creation of standard database reporting functions. The database is divided into four main functions: data entry; data viewing; report generation; and, database re-configuration. Several meetings were conducted with RWRPC members to discuss database structure and design.

During this process, two key design elements were defined and built into the data base structure: (1) web-based simplicity/accessibility; and, (2) expandability, or sometimes called extensibility. The first, web-based simplicity/accessibility, recognizes that the database will reside in a distributed form. Key elements will be brought together in one place, but most of the information will be available by accessing material from web-based sources. That presents a number of difficulties, primarily in the lack of homogeneity among formats. Early on it was agreed that the problem could not be addressed within the scope of this project. We could and did design data entry forms to standardize information describing water projects in the Valley (goals, objectives, funding, contacts, and so on), and as much other metadata as possible. Also recognized was the fact that many decision makers wanted high level summary reports initially to examine the overall situation. So, information for reports had to be available easily and in readily digestible formats for a wide variety of people who would not necessarily be experts in scientific, engineering, or related fields.

The concept of expandability (extensibility) has been mentioned briefly in that the database was expected to be a regional sharing and network building tool. The RWRPC recognized that the utility of the database went far beyond the immediate scope of the Assessment, and that the database could serve as a focus for strengthening social and institutional networking in the Shenandoah Valley region. As a result, a major outcome of the Assessment was a recommendation by the RWRPC for the Shenandoah Valley Pure Water Forum to seek funding to maintain and continue to develop the database as a regional information sharing and network building tool. Based on this action, the Pure Water Forum has developed a concept for “ShenWater: A Knowledge Based-Network.” The concept relies on the following two goals:

- (1) Develop the database into an accessible, user-friendly information source that has expanded functions and serves the Shenandoah Valley as an up-to-date knowledge base.
- (2) Determine how to encourage and enable decision makers, professionals, experts, and citizens to use and contribute to the knowledge base when considering water issues.

NEXT STEPS

Foremost among the next steps is to develop the Water Resources Comprehensive Plan. The Comprehensive Plan needs to incorporate the right mix of incentives, education, regulation, knowledge about the resources, as well as estimates of growth and demands for water for all kinds of uses including economic development, agriculture, and conservation and recreation. Model programs need to be studied to determine what has worked and what has not. Mandated programs need to be examined to determine which ones can best be solved locally and which need regional approaches. To provide the knowledge base for moving forward, the database structure started during the Assessment needs to be completed and made accessible in a web-friendly format.

SUMMARY AND CONCLUSIONS

The formation of the Regional Water Resources Policy Committee was the next logical step in a process that had been developing in the Valley for a substantial period – perhaps for 20 years or more – to come to grips with both water quality and quantity issues. The prolonged drought provided the impetus to take that next step. One of the strengths of the RWRPC is that it is made up of elected officials who are responsive to the public, rather than being a group of appointed individuals. The participation in the RWRPC cuts across two states and covers both the Shenandoah and Opequon watersheds in the Shenandoah Valley. Finally, the RWRPC is supported by a broad cross section of stakeholders representing government, various state and federal agencies, public institutions, universities, corporations, and citizens groups. The Assessment was successful in establishing a clear set of goals with wide community “buy-in”. This involvement across the entire Valley community bodes well for establishing comprehensive water policy throughout the Shenandoah Valley.

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VIRGINIA'S TMDL IMPLEMENTATION PLAN DEVELOPMENT

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KEYWORDS: stakeholders, focus groups, targeting BMPs, GIS, milestones, watershed modeling

ABSTRACT

Stakeholder participation and buy-in are integral parts of the Virginia Department of Conservation and Recreation's (DCR) Total Maximum Daily Load (TMDL) implementation planning efforts. Considerable effort is undertaken to create focus groups and a steering committee whose membership consist of watershed residents, local organizations, and federal, state and local government agencies. DCR staff, in conjunction with contractors, facilitate these meetings to help assure that state and federal requirements are understood. Due to the variability in the required nonpoint source load reductions for TMDLs developed in Virginia and non-linear implementation costs associated with these reductions, it is extremely important that watershed planning efforts are both specific and comprehensive. TMDL implementation requires specificity of individual contributing pollutant sources and their relative contributions. Targeting the precise implementation measures is essential to achieving water quality goals. DCR has developed watershed and pollutant specific TMDL implementation plans (IPs) to date. These plans provide a case study on the use of GIS, monitoring data, and watershed modeling to effectively inventory, target BMP placement, establish costs, and project water quality improvements. Without the ability to effectively target BMP placement, pollutant reduction goals can be adversely affected. Without quantifying BMP types and overall numbers a valid cost benefit analysis cannot be developed. Without the ability to quantify control measures and target placement of these measures, developing schedules for water quality milestones or projecting dates of water quality achievement will be extremely difficult. This type of watershed planning is gaining the support of affected stakeholders and conservation groups across Virginia.

INTRODUCTION

Virginia's 1997 Water Quality Monitoring, Information and Restoration Act states in section 62.1-44.19:7 that the "Board (State Water Control Board) shall develop and implement a plan to achieve fully supporting status for impaired waters". DCR initiated a TMDL implementation planning process in the fall of 2000 to determine the effort and resources needed to fulfill EPA implementation plan requirements and to fulfill Virginia's legislative requirement for the development of TMDL IPs. In fulfilling the state's requirement for the development of a TMDL IP, a framework was established for reducing non-point source pollution (NPS) loads and achieving the TMDL load allocations to obtain the TMDL water quality goals. Without the development of an effective implementation plan and actual implementation of pollution control measures the TMDL is nothing more than a study with findings that will likely be disregarded.

Six IPs (Table 1) have been developed or initiated to address fecal coliform, nitrate-nitrogen, and benthic impairments on 19 separate stream segments in five geographic areas of the state (Figure 1). An additional seven IPs are to be initiated by the end of 2004. The IP's consists of monitoring, public outreach, targeting analysis, milestone development/responsible parties, cost estimation, and funding components. IPs are developed in accordance to the *Guidance Manual for Total Maximum Daily Load Implementation Plans* (DCR and DEQ 2003).

Table 1. Implementation plans and impairments.

Geographical Area	Impairments
North River (Rockingham County)	benthic, fecal coliform, nitrate-nitrogen,
Holman's Creek (Rockingham/Shenandoah Counties)	benthic, fecal coliform
Catoctin Creek (Loudoun County)	fecal coliform
Willis River (Buckingham and Cumberland Counties)	fecal coliform
Blackwater River (Franklin County)	benthic, fecal coliform
Middle Fork Holston (Washington County)	benthic, fecal coliform

Considerable effort was undertaken to create focus groups and steering committees whose majority membership were watershed residents with as little governmental involvement as possible. State governmental agencies and the contractor roles were to facilitate these meetings and to assure state and federal requirements were achieved and to document the process undertaken.

Review of TMDL development

All TMDLs were based on models utilizing Hydrologic Simulation Program – Fortran (HSPF) except for the benthic impairments. The benthic TMDLs were based on a reference watershed approach using the Generalized Watershed Loading Function (GWLF model). Each had detailed land use land cover GIS data and modeled the delivery of the pollutant loadings to the stream. Summary of the TMDL allocations development included:

- Most/all livestock must be excluded from streams within all impairments;
- Substantial land-based NPS load reductions are needed in Muddy Creek and Pleasant Run (North River watershed) and a 10% reduction of FC in runoff from improved pasture/hayfields in the Hutton Creek (Middle Fork Holston) watershed;
- Poultry processing plant must reduce NO₃-N levels in outflow by 35%;
- Most/all failing septic systems and straight pipes must be identified and corrected;
- Implicit in the requirement for correction of straight pipes is the need to maintain all functional septic systems;
- Anthropogenic FC sources will be addressed in stage I of the implementation plan, setting aside any reduction of wildlife. The Virginia Department of Environmental Quality (DEQ) will re-assess streams after stage I to determine progress and to see if water quality standards have been attained.
- Primary stressors for benthic impairments include phosphorus and sediment with sediment as the predominant stressor.

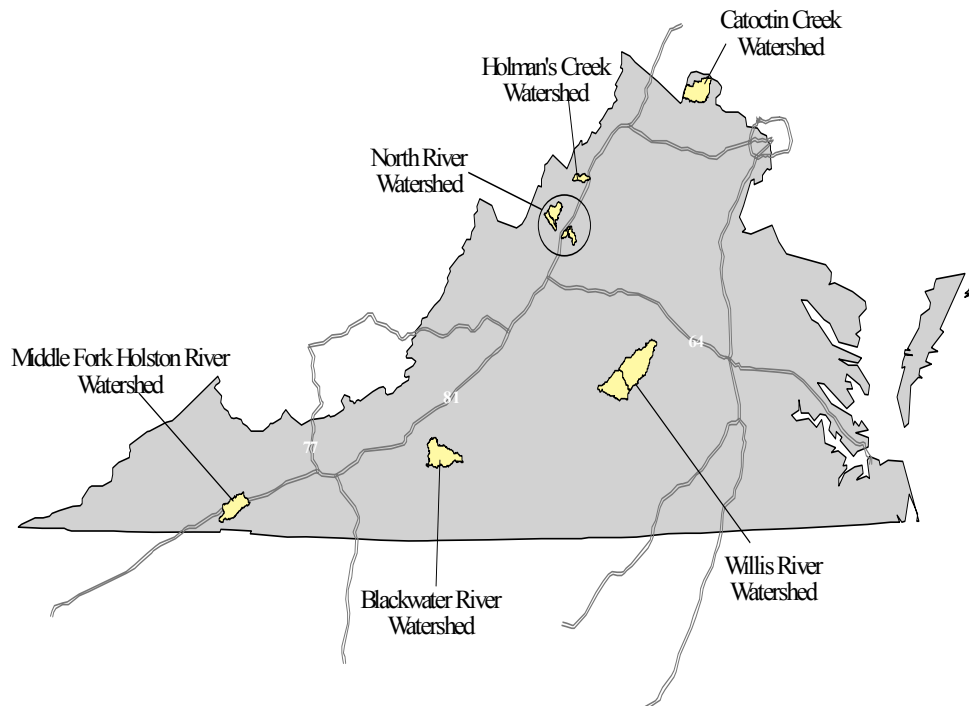


Figure 1. DCR's TMDL IP project areas.

METHODOLOGY

Process for Public Participation

Citizens and interested parties in the watersheds were encouraged to become involved in the IP process. Public participation took place on three levels. First, public meetings were held to provide an opportunity for informing the public as to the end goals and status of the project, as well as, a forum for soliciting participation in the smaller, more-targeted meetings (*i.e.*, focus groups and steering committee). Second, focus groups were assembled from communities of people with common concerns regarding the TMDL process and were the primary arena for seeking public input. The following focus groups were formed: Agricultural, Residential, Environmental, and Governmental. A representative from DCR, local agencies (*i.e.*, County government, Soil and Water Conservation District) and the contractor (MapTech, Inc.) attended focus group meetings in order to facilitate the process and integrate information collected from the various communities. Third, a steering committee was formed with representation from all of the focus groups; local, state and federal agencies, Farm Bureau, community clubs and organizations, citizen groups, and MapTech.

Throughout the public participation process, major emphasis was placed on discussing BMP specifications, location of control measures, education, technical assistance, and funding. The Agricultural and Governmental Focus Groups agreed that potential control measures identified through the implementation plan process should be practical, cost-effective, equitable, and based on the best science and research available. It was determined through the Residential Focus Group input that stream-walks must be performed during implementation to accurately identify

straight pipes and failing septic systems.

All focus groups agreed that education is key to getting people involved in implementation. There must be a proactive approach by agencies to contact farmers and residents to articulate exactly what the TMDL requires and what structural and management BMPs are needed to obtain water quality goals. For the agricultural community, small workshops and farm visits would be needed to accomplish this. For residential issues, small community meetings similar to small workshops proposed for the agricultural community were recommended for educating homeowners about septic system maintenance. It was generally recognized that homeowners are unaware of the need for regular septic system maintenance.

Assessment of Control Measures

The quantity of control measures required during implementation was determined through spatial analyses of land use, stream-network, elevation, building-footprint, and soils maps along with regionally appropriate data archived in the VDCR Agricultural BMP Database and TMDL development documents. The map layers and archived data were combined to establish high and low estimates of control measures required overall, in each watershed, and in each subwatershed. Additionally, input from local agency representatives and contractors were used to verify the analyses. Estimates of control practices needed for full implementation in the North River watershed (VDCR 2001), as an example, are listed in Table 1. Associated cost estimations for systems needed for full livestock exclusion and land-applied reductions were calculated by multiplying the unit cost per the number of units in each subwatershed.

Table 1. Estimation of average control measures with unit cost needed during implementation for agricultural and residential programs in North River watersheds.

Control Measure	Unit	Estimated Units Needed	Average Cost / Unit (\$)
<i>Agricultural Program:</i>			
Full Exclusion System	System	478	15,023
Cropland Fencing	Feet	105,500	1.10
Hardened Crossing	System	190	2,000
Loafing Lot Management System	System	35	31,396
Dairy Waste Control Facility	System	18	61,499
Poultry Waste Control Facility	System	3	22,132
Cover Crop	Ac	5,154	138
Technical Assistance	man-year	18	50,000
Administrative Assistance	man-year	6.5	35,000
<i>Residential Program:</i>			
Septic System Replacement	System	17	2,500
Alternative Waste Treatment System	System	27	7,500
Septic System Repair	System	10	1500
Technical Assistance	man-year	1	50,000
Administrative Assistance	man-year	0.5	35,000

Cost estimations to fix failed septic systems and replace identified straight pipes were based on the combination of drain-field maintenance, new septic systems, or alternative waste treatment system. It was assumed that sites were evenly split between needing standard systems (*i.e.*, septic systems or drainfield maintenance) and alternative systems (*e.g.*, peat moss filter systems). To determine man-years necessary for agricultural technical assistance during implementation, the total practices needed to be installed annually was divided by the number of BMPs that a technician from the soil and water conservation district (SWCD) had historically processed in a year. In quantifying technical assistance needs, man-hours for educational outreach activities were also considered. Members of the Residential and Governmental Focus Groups estimated technical man-years and administrative man-years required to provide residential technical assistance and educational outreach identified during plan development.

RESULTS

Calculations for the amount of stream fencing required account for the perennial and intermittent streams miles in per watershed. Associated with the streamside fencing through pasture (and woodland adjacent to pasture) are the number of full livestock exclusion systems consisting of streamside fencing, cross fencing and watering source. Streamside fencing of cropland estimates did not require a full livestock exclusion system; instead, it is assumed that temporary poly-wire will be used to restrict livestock from entering stream. The existing fencing was taken into account when estimating costs.

In order to address the land reductions needed in several watersheds, the benefit of including a 25 ft. buffer with streamside fencing was first calculated. Given that reductions were not sufficient to meet TMDL reduction goals, additional control measures will need to be implemented to obtain FC land reductions. Land reductions can be addressed through installation of loafing lot management systems, manure incorporation in soil, installation of animal waste control facilities, pasture management, conversion of pasture to hayland, and export of waste. The number and location of failing septic systems and straight pipes were based on numbers reported in the TMDL.

Table 2. Estimated total implementation cost for agricultural BMPs, residential BMPs, and technical assistance in North River watersheds.

Control Measure	Average Total Cost (in million \$)
Livestock Exclusion BMPs	7.67
Land-applied BMPs	2.98
Residential BMPs	0.26
Technical Assistance	
<i>Agricultural Programs</i>	1.13
<i>Residential Programs</i>	0.07
Total	12.11

As depicted in Tables 2, the total average cost to install control measures that will ensure full livestock exclusion from streams in the North River watersheds is \$7.67 million excluding technical assistance (VDCR 2001). The total cost to install control measures to obtain the land-

applied reductions in the four watersheds is estimated at \$2.98 million excluding technical assistance. The total cost estimated for repair/replacement/installation of private sewage systems was \$260,000. Technical assistance cost accounts for an additional \$1.2 million.

Targeting Analysis

Implicit in the process of a staged implementation is targeting of control measures. Targeting of critical areas for BMP installation was accomplished through analysis of land use, farm boundaries, stream network GIS layers, monitoring results, survey responses, and HSPF modeling. Monitored data collected during the development process was used together with spatial analysis (GIS and HSPF modeling) results to identify subwatersheds where initial implementation resources would result in the greatest return in water quality improvement. A comparison was done assuming a 50 percent implementation of livestock exclusion practices using a targeted approach by subwatershed verses completely random placement (in the overall watershed). Examples of the effectiveness of targeting specific subwatersheds from Keeling, 2003 are illustrated in Table 3. It was assumed that failed septic systems in close proximity to a stream would have a larger impact on water quality than a system located in upland areas. Therefore, spatial analysis using GIS was performed to identify residents within 300 feet of a stream. Using the results, efforts can be made to contact identified residents first during implementation to address septic system failures and straight pipes. Monitoring showed larger impact from livestock sources in some watersheds as compared to others where human sources were more prevalent.

Table 3. Effect of targeting BMP placement on modeled FC geometric mean water quality standard violations in selected TMDL IP watersheds.

Milestone	Dry River	N.F. Blackwater	Middle Blackwater
	(%)	(%)	(%)
Existing	64	83	82
Non-targeted	54	74	61
Targeted	31	29	40

Milestone Development

Milestone development was not as specific as targeting reductions. This is justified since it is unlikely that a voluntary program will have 100 percent participation from the residents in a watershed regardless of the targeting efforts of local SWCD. Therefore, broad more random reduction scenarios (the overall watershed) were modeled to layout a timeline that would coincide with a 5 year funding commitment from VDCR. This produced 4 milestones that span 10 years with the first 3 stages heavily dependent on implementation occurring at an optimistic rate in the first 5 years. These milestones from VDCR, 2001 are detailed in Table 4 for the North River project area and are similar for the other project areas.

Table 4. Estimation of FC geometric mean water quality standard violations at each milestone in Dry River, Muddy Creek, Pleasant Run, and Mill Creek.

Milestone	Dry River (%)	Muddy Creek (%)	Pleasant Run (%)	Mill Creek (%)
Existing	64	100	99	100
1	54	97	95	98
2	0	63	1	65
3	0	0	0	0

DISCUSSION

The primary benefit of implementation is cleaner waters in Virginia. It is hard to gage the impact that reducing fecal contamination will have on public health, as most cases of waterborne infection are not reported or are falsely attributed to other sources. However, because of the reductions required, the incidence of infection from fecal sources, through contact with surface waters, should be reduced considerably. Additionally, because of stream-bank protection that will be provided through exclusion of livestock from streams, and restoration of the riparian area through implementation in some areas, the aquatic habitat will be improved and progress will be made toward reaching the General Water Quality standard (benthic) in these waters. The vegetated buffers that are established will also serve to reduce sediment and nutrient transport to the stream from up-slope locations. In areas where pasture management is improved through implementation of grazing-land-protection BMPs, soil and nutrient losses should be reduced, and infiltration of precipitation should be increased, decreasing peak flows downstream.

The residential programs will play an important role in improving water quality, since human waste can carry with it human viruses in addition to the bacterial and protozoan pathogens that all fecal matter can potentially carry with it. Benefits to homeowners include an improved understanding of private sewage systems, including knowledge of what steps can be taken to keep them functioning properly and the need for regular maintenance. This knowledge will give homeowners the tools needed for extending the life of their systems and reducing the overall cost of ownership. The average septic system will last 20-25 years if properly maintained.

CONCLUSIONS

Implementation

Implementation for the North River, Blackwater and Middle Fork Holston projects began in the fall of 2001, implementation in Catoctin and Holman's will begin in the fall of 2004. The final milestone will be de-listing of the impaired segments from the Commonwealth of Virginia's 303(d) List of Impaired Waters, which is anticipated to occur by 2011 or 2014. Based on meeting the above milestones, a five-year implementation plan outline was formulated. Progress toward end goals will be assessed during implementation through tracking of control measure installations and continued water quality monitoring. It is recommended that continued water quality monitoring be made based on the existing monitoring network and spatial distribution of the staged implementation plan.

Stakeholder's Roles and Responsibilities

Achieving the goals of this effort (*i.e.*, improving water quality and removing these waters from the impaired waters list) is without a doubt dependent on stakeholder participation. Not only the local stakeholders charged with implementation of control measures, but also the stakeholders charged with overseeing our nation's human health and environmental programs must first acknowledge there is a water quality problem and then make changes in our operations, programs, and legislation to address these pollutants.

Successful implementation depends on stakeholders taking responsibility for their role in the process. The primary role, of course, falls on the landowner. However, if voluntary approaches prove to be ineffective and the public "will" is to force compliance with existing laws through court actions, then landowners may be faced with implementing corrective actions without economic assistance.

Lessons Learned

The following lessons have been learned based on IPs developed today and the implementation projects initiated:

- It is important to make sure that participants on steering committee and working groups are provided information about water quality standards and what the TMDLs require.
- In adopting water quality standards, based on EPA criteria, states need to carefully consider attainability of such standards.
- Available funding motivates actions, this applies to government agencies supporting implementation as well as landowners making improvements on their land.
- Changing behavior in a voluntary approach requires building relationships that take time.
- Stakeholders are slow to engage in the implementation plan process due to apprehension about what the plans will require and a committed source of funding to carry the plan through.

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**EVALUATING CONSERVATION RESERVE PROGRAM
IMPACTS ON SMITH CREEK EROSION RATES
IN THE SHENANDOAH VALLEY**

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ABSTRACT

This project created an evaluation study on four different properties over a four-mile stretch of Smith Creek. This project documented the effects of stream bank restoration efforts on erosion and water quality. These sites were chosen due to their various connections with the Conservation Reserve and Enhancement Program (CREP). The four sites include one that implemented CREP interventions in 2002, a reference reach that was already forested, a site implementing CREP at the time of the study, and a final site that has not yet decided to enter the program. Through the use of bank pins, cross-sectional analysis, and biological and chemical monitoring, and regular visual surveys at each site we were able to create a baseline data set for evaluating the long-term benefits of tree plantings and bank restorations on overall stream health. Evidence has already begun to show the success of stream restoration through one year of data collection. Accomplishments with stream bank restorations were evident after Hurricane Isabel struck in September 2003. This process continues under the auspices of the Virginia Department of Forestry and the sponsorship of James Madison University.

INTRODUCTION

The Virginia Department of Forestry, in cooperation with local soil and water conservation districts, the Farm Service Administration, and the USDA's Natural Resource Conservation Service (NRCS) are involved in setting aside land along streambanks for the purpose of reducing

erosion and preventing surface runoff of nutrients into streams. This project selected four sites along Smith Creek as part of a yearlong data collection and analysis on erosion and water quality. The four locations were chosen due to their connection to the Conservation Reserve Enhancement Program (CREP) and the various stages of riparian buffers and stream banks represented at each site.

The Conservation Reserve Enhancement Program is an effort focused on preserving the environment by rehabilitating riparian corridors. This cost-share approach employs the landowner to take action with financial incentives. The program removes agriculture and livestock from direct access to streams or other natural waters. By removing agriculture from the immediate stream corridor, the farmer will no longer cultivate or spread various forms of fertilizer near the edge of the stream, reducing potential nutrient enrichment.

There are two main objectives to removing any livestock from the stream. First is to lower nutrient levels in the water. High nutrient levels cause rapid growth of algae blooms, which consume dissolved oxygen during decomposition. Dissolved oxygen is essential for respiration and survival of fish, plants, and macroinvertebrates. Livestock excrement plays a significant role in the increase of nitrogen in the waterways.

The second objective is reducing the sediment load in the water. When livestock are permitted continuous access to a stream, they accelerate bank erosion by knocking soil into the water and trampling vegetation. The soil lost due to erosion becomes sediment, which then disperses throughout the stream. An increase in sediment load often deprives freshwater macroinvertebrates of their natural habitat.

Landowners involved in CREP must abide by several regulations. The landowner must remove livestock and agriculture for a minimum width of 35 feet. If livestock are present, they must be fenced out of the area and an alternative water source provided. Fast growing grasses and shrubs are then planted to prevent short-term erosion. A variety of hardwood trees are planted in the area along the stream to establish a root system that will prevent erosion for years to come. The landowner can choose from 30 species of trees that they would like to have planted on the property; however, 80 percent of the selected trees must be natural hardwoods, such as oaks.

The participating landowner signs a 15-year contract that prevents the trees from being harvested and excludes livestock or agriculture from the buffer area. The landowner receives a “rental” payment as partial compensation for loss of productivity from this portion of his/her land. At the conclusion of the contract, the landowner assumes the entire responsibility for the condition of the CREP installment.

The ultimate goal of CREP is to improve water quality in the Chesapeake Bay. The fishing industry in the Bay has declined dramatically in the past 20 years, due to water quality conditions and sediment load. Century old grass beds, which provide a habitat for fish and aquatic life, are being lost at an alarming rate. The sediment clogs the grass beds and prevents essential light from reaching the aquatic vegetation.

Riparian buffers can have a major impact on erosion and water quality. Some of the benefits include reduced sediment and nutrient loads, decreased water temperatures leading to increased dissolved oxygen, improvement of aquatic habitat, provision of wildlife corridors and habitat and increased interest in aesthetics and recreational use (Palone 1998).

STREAM BANK EROSION

Stream bank erosion is the direct removal of banks and beds by flowing water, usually during bankfull events. A bankfull event refers to the level of stream flow that has enough energy to alter the stream channel. Several factors that influence the amount of change a stream channel experiences during bankfull events include the vegetative cover and the slope of the bank.

The presence of vegetation, specifically riparian forest buffers, can greatly impact the extent of bank erosion during bankfull events. Stream banks without vegetation are more susceptible to erosion because the soil is left exposed. Vegetation impedes the flow of runoff, and provides an anchor for the exposed soil. Research has been done that demonstrates the relationship between riparian buffers and erosion rates, though more is needed to fully understand the impact of grazing and agriculture on the riparian and aquatic ecosystem. A study in Bear Creek, Iowa examined erosion rates for one year along an 11 kilometer reach, which included cow pastures, horse pastures, crop land, and forested buffers within the riparian zone (Zaines 2004). This study found that “if riparian forest buffers had been established along all the non-buffered segments of the 11 km stream reach, total stream bank soil loss would have been reduced by approximately 72 percent” (Zaines 2004). The study also concluded that as riparian buffers mature, they become more effective at preventing bank erosion (Zaines 2004).

The slope, or gradient, of a stream bank affects the impact of a bankfull event. The slope of a bank is measured in comparison to the water level. For example, 30-degree banks gradually slope away from the water level and 90-degree banks are perpendicular to the water level. Generally, the higher the degree of the angle, the more prone the bank is to erosion from bankfull events. A 30-degree bank angle, as the ideal, dissipates the flow of water throughout the bank and onto the flood plain. In contrast, when the flow of water encounters a 90-degree, or vertical, stream bank, the energy does not dissipate. Instead, the energy of the stream flow cuts into the vertical bank and causes erosion. Banks with an angle greater than 90-degrees are considered undercut, and are prone to mass wasting. Mass wasting refers to erosion in which the flow of water causes large portions of a stream bank to slough off into the stream channel.

Stream bank erosion is detrimental in that it increases sediment load and deposition, changes the stream channel, and causes the loss of valuable land. First, as discussed above, stream bank erosion accounts for about 50 percent of a stream’s sediment load. High sediment loads cloud the water, preventing sunlight from reaching aquatic vegetation. Deposition can also affect aquatic life by clogging their habitat on the channel bottom.

The process of bank erosion and sediment deposition can cause changes in a stream channel. Bank erosion widens the channel, while the sediment lost deposits on the stream bottom and channel becomes shallow. A wide, shallow channel does not support the same variety of aquatic

life as a narrow, deep channel. It is also difficult for a stream bank with heavy erosion to correct itself, thus continuing this process and causing the loss of valuable land over time.

SMITH CREEK WATERSHED AND STUDY SITES

According to a TMDL fact sheet, Smith Creek flows along a 25.82 mile path through Rockingham and Shenandoah counties with a drainage basin of 106 square miles (*Total Maximum Daily Load Development for Smith Creek* 2004). Smith Creek is part of the Shenandoah River watershed, which is part of the Potomac River basin, which contributes to the Chesapeake Bay. According to the Shenandoah Water Window, the land cover for the Smith Creek watershed is approximately 51 percent agriculture, 45 percent forested, and 4 percent urban (*Shenandoah Water Window* 2004). Most of the forested land lies along the western side of Massanutten Ridge, a 40 mile long spine that divides the North Fork and South Fork of the Shenandoah River. The main channel of Smith Creek flows parallel to Massanutten from South to North. Our study focused on the agricultural areas.

Four study sites were selected for this project, reflecting the varying degrees of participation in CREP. They are located along a 5 miles stretch of the river beginning in northern Rockingham County, extending into South Shenandoah County near New Market.

The most southerly is the Witmer property, which has been in CREP for a year and half now, and has had significant bank work. The immediate upstream neighbor of the Witmer property is a farmer who has been in CREP for five years and has taken significant steps to preserve and enhance his waterway. The bank on the Witmer land was pulled back to a 3 to 1 grade on both sides of the stream for roughly 100 feet on each bank. The right bank, as shown in Figure 1, was pulled back in 2002 and has biologs at its base to prevent toe erosion and give the young trees and grasses an opportunity to take root. The left bank was pulled back in the fall of 2003 and a cedar revetment was used to provide bank protection. Hurricane Isabel caused severe damage at this location as most of the phototubes were bent to the ground, potentially killing many of the trees and the cedar revetments were a total loss. Before the Witmer farm joined CREP, their fields were used as dairy pastures. Today the land is used to cut hay to feed livestock at other farms, and winter wheat is planted in late fall to prevent erosion in what is typically not the growing season.

The reference reach land is a stretch of forested stream banks used for comparison to the other properties. A reference reach is a site that represents the “ideal” bank conditions. As shown in Figure 2, the banks of the land are heavily vegetated for at least 30 feet on both sides of the stream. The extensive root network of the trees is the ideal bank support system and the long-term goal of CREP. Cattle grazed close to the left bank of the stream until 2001, but they are presently fenced out and there are no obvious signs of erosion from livestock.

The King farm is 32 acres of pasture and crop land with only minor vegetation along the stream. Three years ago, the land was used as a cattle pasture where livestock were allowed direct access to the stream. Smith Creek flows approximately a half-mile loop through the farm. The upstream neighbor of Mr. King is a cattle farmer who continues to allow cattle direct access to Smith Creek; as a result there is significant erosion and mass wasting of stream banks.

In 2003, the King farm joined CREP. Significant stream bank restoration efforts took place including a 3 to 1 pull back and biologs. A wide variety of trees and fast growing grasses were planted along the length of the stream in an effort to prevent erosion. Hurricane Isabel caused significant damage to the CREP trees as the severe flood washed many of the trees away, destroyed others, and caused further losses that will not be fully realized for some time.

The Turner property is the downstream neighbor of King's and has potential to take part in CREP in the future. The Turner land is currently used for agriculture and the majority of the stream banks are well vegetated. The exception is near a large logjam where erosion is severe and both banks are vertical. Figure 3 shows the erosion on the right bank, directly downstream from the logjam. The logjam is still present and continues to cause significant erosion as the flow is directed around the obstruction and toward the banks.

MONITORING CHANGES IN STREAM CHANNEL DIMENSIONS

A stream cross-section is used to determine the width and depth of a channel. At each site a set of 18-inch rebar pieces were placed to mark measuring points outside bankfull, and additional stakes were driven into the stream banks at set intervals. These rebar pieces were placed far enough from the stream bank to encompass any changes in the channel. The exact location of the rebar was then triangulated using permanent objects in the area such as trees and large boulders. We established benchmarks at these permanent objects and recorded their heights. This allows future surveyors to set up their equipment at the exact location each visit.

We determined the channel depth of each site by first setting up a surveyor scope on top of a tripod. We then stretched a tape measure across the channel between the two highest rebar pieces using stakes to secure the tape to opposite ends of the bank. We collected data by looking through the surveyor scope to take an elevation measurement from the surveyor rod. Measurements were initiated on the left bank of the stream (looking downstream) and continued through the channel to the right bank. We recorded measurements for every foot along the channel cross-section. Notes were made for specific measurements such as bank edge, water edge and bankfull.

STREAM PROFILES

The Witmer farm had major reconstruction work on the stream in 2002 and the spring of 2003, including a three to one pull back of stream banks on both sides of the streams, and the installation of toe protection. The first streambed profile was done in August 2003, showing the smooth banks and undulating channel. Only the lower quarter of blue profile above had water at the time of the measurement. (The figure above and all subsequent figures show significant vertical exaggeration.) In September of 2003 Hurricane Isabel struck, dropping around 5.5 inches of precipitation in a short time. The water level reached above the profiles in Figure 1.

The second measurement, in pink, shows some cutting of the streambank, especially on the right side looking downstream. This is the location where the cedar revetments were ripped out by

Isabel's flood. The stream channel also deepened, and some cutting occurred in other places on the banks. This measurement was done just as the early, cool season grasses began growing.

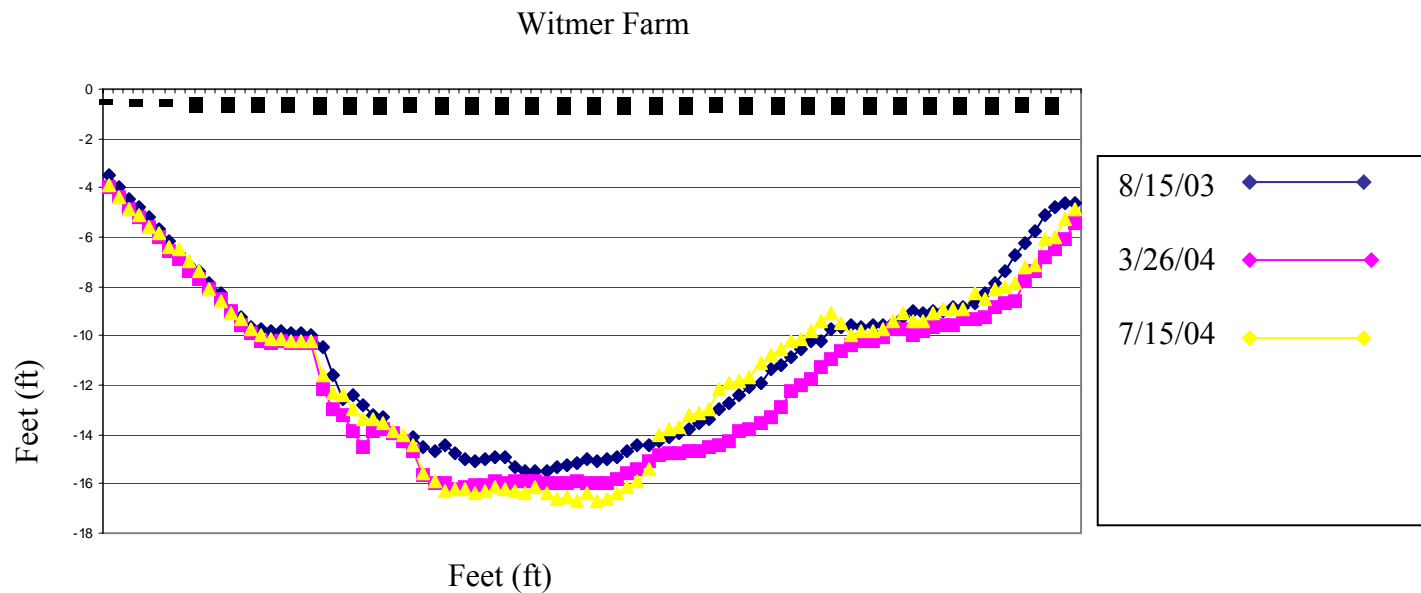


Figure 1. Stream bed profile of the Witmer Farm on three different dates all following streambank reconstruction in the spring of 2003.

The final measurement, in yellow, occurred in July 2004 and again significant things happened. Spring rains were heavy at times, bringing water levels to near bank full on occasion, and showing how well the restoration effort worked despite the hurricanes damage. The right side of the stream saw significant deposition, the channel continued to deepen, and vegetation growth has begun to secure these changes in place. Our observation showed little exposed soil, except at some place on the right bank where the revetments were lost, and the development of healthier stream conditions.

The reference reach was selected for comparison to the other sites (Figure 2). As can be seen there is very little change to the profile between August 14, 2003 and July 30, 2004. The minor difference between the August and March profiles versus the July profile can be easily explained by a misplacement of the starting value of the measuring tape. Adjusting one foot to the left eliminates the discrepancy and creates overlap on all three dates, including the hurricane event.

The Turner farm is just downstream from the King property and has not been entered into CREP as yet, though Mr. Turner is very willing to allow access and has expressed interest in the program. The farm has stretches of riparian land protected by trees, and other sections with major problems including the cross section analyzed above. This reach of Smith Creek holds a very large, fallen sycamore that captured considerable debris, included other trees, forming a significant logjam when Isabel struck in late September. The damage to the streambanks during the hurricane continued to grow in the following months (Figure3). Winter flows showed significant right bank cutting. The bank receives the bulk of the energy from water diverted by the logjam, and most of the vegetation was lost due to severe undercutting. This continued

during heavy spring rains, forming an almost vertical bank eating it way into cropland featuring, at present, a monoculture of corn.

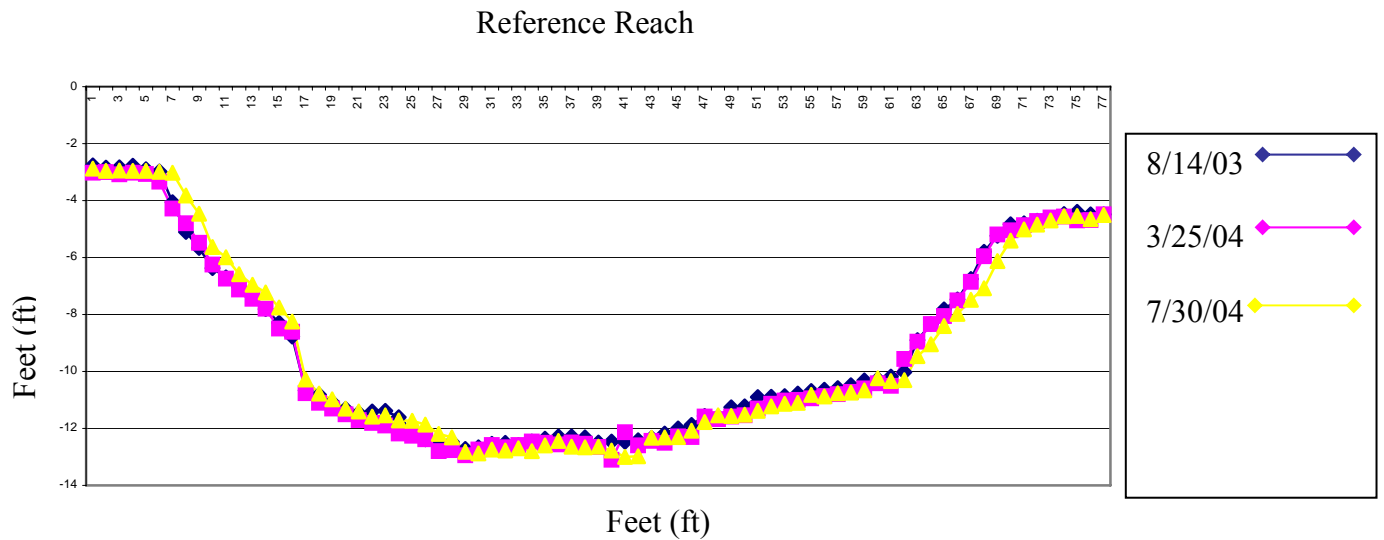


Figure 2. Reference Reach on three different dates covering the project to date.

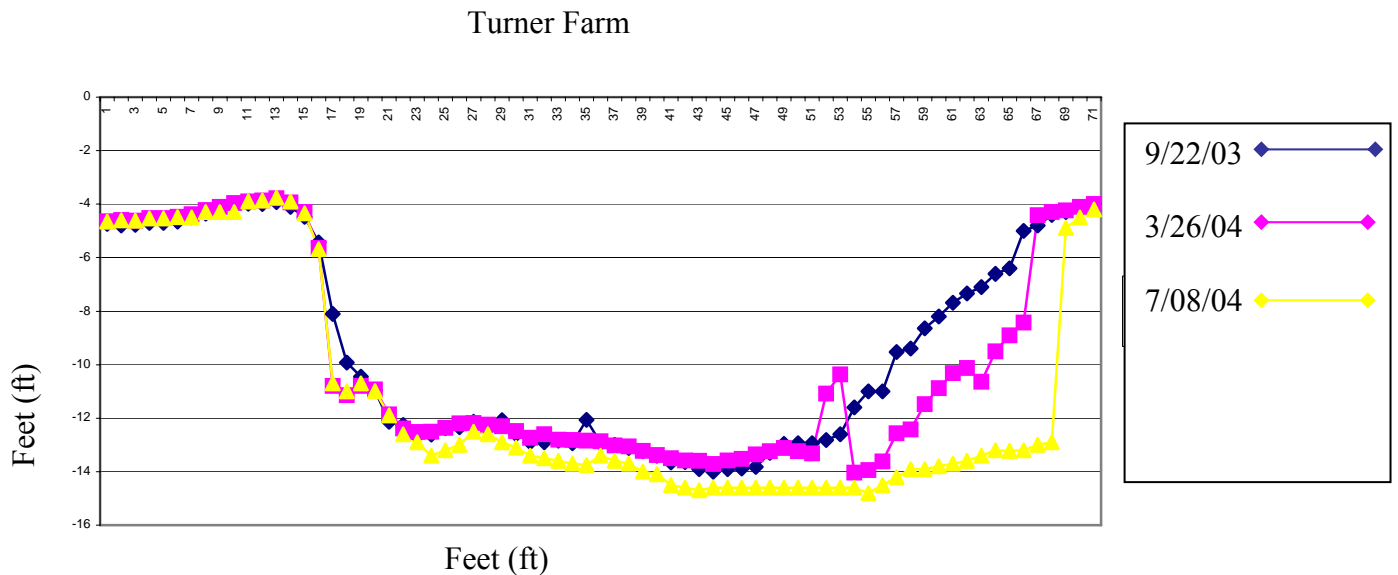


Figure 3. Profile of the Turner Farm shows the reach as it looks just before Hurricane Isabel in September 2003, then in March and in July of 2004. Erosion damage to this site has proven highly significant.

This right bank shows a need for restoration work, including removal of at least portions of the logjam. The exposed bank area is approximately 120 feet long, and the pullback and protected area required would total about one eighth of an acre of what is prime bottomland soil. Without protection considerably more land could be lost. Weighing the balance of losing your best soil to stream protection or erosion makes it tough on the landowner.

The King Farm is no longer an active farming operation. The entire riparian zone is included in CREP, and Ducks Unlimited is converting some areas to wetland. Thus, it has seen major changes in vegetation over the last year, in addition to the planting of trees and streambank pullback and restoration efforts. Much of Smith Creek's bank area was affected by erosion. Large segments of vertical banks were undercut, and mass wasting was a problem. This can be seen in Figure 4. The blue line represents the profile just before Hurricane Isabel. Nearly eight feet of stream bank was lost on the left side of the creek, and the channel profile shifted as well.

Actual work on the stream pullback began in October 2004 once the ground dried enough for equipment to operate. Pullback commenced at water level and continued to the top of the bank, about a vertical distance of 7 to 8 feet. Deposition on the right bank began during the flood event and continued throughout high water events of the winter, continuing into spring. The stream channel itself saw no significant changes from winter through spring, but narrowing and deepening are indicated. It is too early to predict long term impacts at this time.

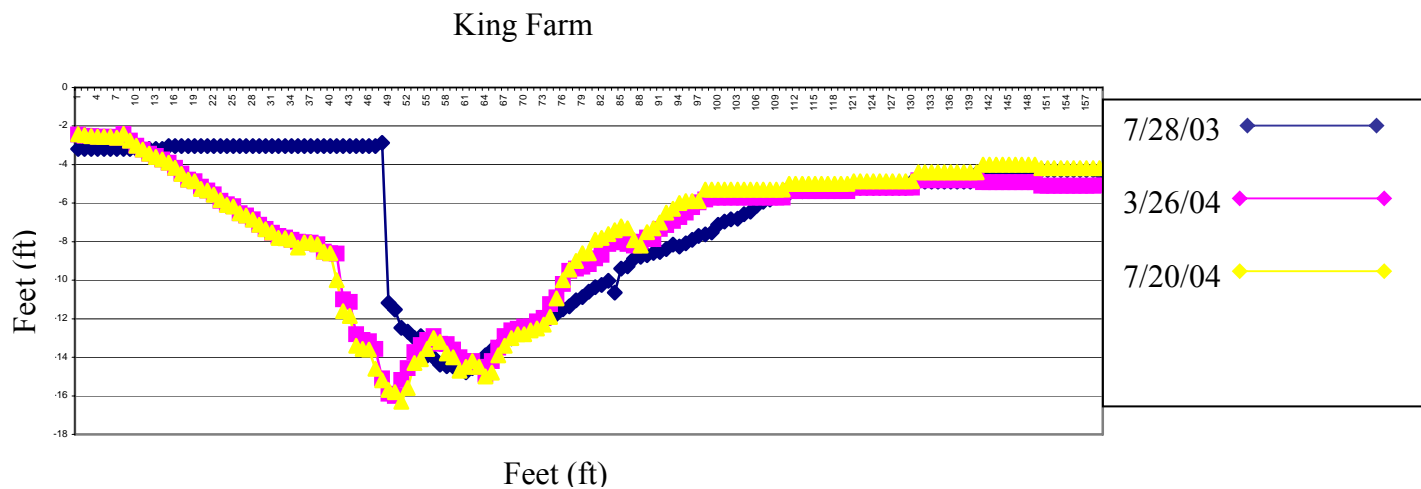


Figure 4. Profile of the King Farm reach before and after Hurricane Isabel. The second two profiles followed the implementation of CREP activities including a 3:1 pullback of the left bank. The horizontal scale of this profile is smaller than that in Figure 1-3.

BANK EROSION RATES

One quantitative measure of stream morphology is the bank erosion rate. This rate, in feet per year, estimates the amount of soil loss on a stream bank. To show the effect of stream bank restorations, we estimated bank erosion rate at the King, Reference Reach, and Turner properties in August of 2003. The rating was also done in March of 2004, approximately six months after Hurricane Isabel and the King property restoration, and again in July of 2004.

The bank erosion rate is based on two calculations: bank erosion hazard index (BEHI) and near bank stress rating. The BEHI considers the characteristics of a stream bank that vary with erosion rates. These characteristics include bank height ratio, ratio of root depth to bank height, root density, percent surface area of protected bank, bank angle, soil composition layering, and

bank material (Rosgen, *Channel Monitoring Methodology*). After analyzing these characteristics at each site, we used two forms created by David Rosgen at Wildland Hydrology to determine the BEHI value. These forms provide index values for each of the aforementioned characteristics, which are totaled and fall into ranges varying between Very Low and Extreme.

The near bank stress is a ratio of the shear stress at “near bank”, or one-third of the cross-sectional width, and the shear stress of the total cross-section. We calculated this ratio using the Near-Bank Stress Calculation and Bank Erosion Prediction form (Wildland Hydrology). Our calculations use the cross-sectional analysis and observed bankfull stations to determine the near bank stress rating. Again, this rating ranges between Very Low and Extreme. Once the BEHI and near bank stress rating are determined, they are compared to the North Carolina Erodibility Graph in order to obtain the bank erosion rate.

Table 1 shows the bank erosion rates for each site on the three dates of monitoring. The last column demonstrates the difference in bank erosion rate between August of 2003 and July of 2004.

Table 1: Bank erosion rate comparison.

Bank Erosion Rate (Feet per year)	August 2003	March 2004	July 2004	Difference (August- July)
Witmer	0.07	0.006	<0.006	(-) 0.064
Reference Reach	<0.006	<0.006	0.006	0.0
King	0.6	0.006	0.006	(-) 0.594
Turner	0.006	0.08	0.5	(+) 0.494

From this table, we can conclude that a bank with established vegetation and low bank angle, such as that at the Reference Reach, will lose minimal amounts of soil due to bank erosion. The Witmer property also showed significant improvement between August 2003 and July 2004. A decline in the bank erosion rate can be attributed to the restored 45-degree bank angle, and the development of vegetation on the restored bank over the last year. Therefore, as herbaceous cover and the planted riparian buffer mature, the root system provides for a more secure stream bank.

The restoration of the stream bank at the King property had a significant impact on the bank erosion rate, in that the bank angle declined from 95 degrees to 30 degrees. The change in bank angle, in addition to the development in planted vegetation on the restored area between March and July, both caused a half-a-foot per year decline in bank erosion.

The Turner property also demonstrated a significant change in that the bank erosion rate increased by approximately half-a-foot per year. As seen in the cross-sectional analysis of the Turner property, the right bank has lost approximately 5 to 6 feet between August 2003 and March 2004. This loss was not predicted by our bank erosion rate calculations, and may indicate a level of ambiguity in the analysis. However, since the bank receives the bulk of the energy diverted by the upstream debris and most of the vegetation has been lost due to severe undercutting, the bank erosion calculation has now increased to predict about half-a-foot per year loss.

Overall, the bank erosion rate calculations indicate a level of success in the stream restorations at the Witmer and King properties. The Turner property, the only property with an increase in bank erosion rate, has not received any restoration work and may be a further indication of the benefits of stream restoration and buffer enhancement.

CONCLUSION

The goals of CREP are to reduce bank erosion, sediment loss, and agricultural runoff. These goals are accomplished through stream bank restorations and vegetative buffers. Restorations act a short-term solution, while vegetation promotes success in the long-term.

By decreasing the slope of a bank, less soil is lost because the energy of the water dissipates over the flood plain. The bank restorations at the King and Witmer properties show immediate benefits in reducing soil loss during bankfull events. However, bank restorations are only one part of CREP. Without the vegetation buffer, these benefits will diminish over time.

In contrast, we expect the benefits of the vegetation to increase over time. Over the course of this study we found little correlation between vegetation and water quality. Currently, the one-to-two year old saplings at the King and Witmer properties do not have the root network to effectively meet the water quality goals of CREP. As the trees continue to mature, we anticipate seeing the full effectiveness of the program.

ACKNOWLEDGMENTS

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BROWNFIELDS: AN ECONOMIC CASE STUDY ANALYSIS OF VIRGINIA'S PROGRAM INITIATIVE

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ABSTRACT

Many U.S. business activities over the past century have contributed to site-specific soil and groundwater contamination, which ranges from light to severe. This contamination limits the future uses of property and becomes an important consideration after a business has closed. Liability issues arising from owning contaminated property can lead to millions of dollars in cleanup costs and legal fees for former, current, or prospective property owners. Idle and abandoned property undermines a municipality's tax base and slows economic growth.

In the 1990s, a handful of state governments developed voluntary remediation/cleanup programs, or VRPs, that were designed to promote the development of "abandoned, idled, or under-used industrial and commercial property," commonly referred to as Brownfields (EPA 2003). These programs usually provide a combination of liability and regulatory relief to property owners with a financial incentive for redevelopment.

This paper examines 10 redevelopment projects that participated, or are currently participating, in Virginia's VRP in order to evaluate the success of the VRP policy initiative. This research focused on examining the number of jobs created; changes in property value after cleanup/redevelopment; increased revenues from business activities; the severity of contamination; and the cost of cleanup at each site. Available data was examined with the purpose of identifying the environmental and economic effects of participating in the VRP by establishing project multipliers, through which the overall cost effectiveness of Virginia's VRP was measured.

INTRODUCTION

Years of unregulated waste disposal have allowed hundreds of acres of U.S. property to become contaminated. These contaminated sites are located in almost every city and many are well known, such as Love Canal in Niagara Falls, New York or the town of Woburn, Massachusetts. Still others are much more ordinary and remain relatively anonymous. People are not surprised to learn of contamination at chemical plants and petroleum refineries; however, regularly visited business sites such as dry cleaners, automotive repair and painting shops, gas stations, and print shops, are often contaminated as well.

The most contaminated sites galvanized the U.S. Congress to pass the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), which is better known as

Superfund, in 1980. This statute, and its amendments, is important in any discussion of Brownfields because its liability scheme creates a barrier for redevelopment projects.

Superfund is designed to force a responsible party, or parties, to pay for the cleanup and restoration costs associated with environmental contamination at a site. To make a responsible party pay, the Superfund statute adheres to a strict, joint, and several liability scheme. Under a strict liability scheme, a property owner or operator is liable if he/she owns or controls the property, regardless of when contamination occurs or if the waste disposal was considered appropriate or lawful at the time. Joint and several liability enables the Government to force parties to pay equal amounts regardless of the amount of hazardous waste each party contributed.

Cleanup efforts have focused on the worst sites first, thus leaving moderately contaminated property in a state of limbo. Such properties are not contaminated to an extent where they can be placed on Superfund's National Priorities List (NPL) but they are not considered "clean" enough for use. As a result, prospective property owners and potential developers are hesitant to acquire underused or idle property for fear that it may be contaminated and that they would become a responsible party—forced to pay a portion of the site investigation and the cleanup costs.

Although the liabilities associated with contaminated property are considered the biggest hurdle to site redevelopment, the costs associated with identifying and remediating site contamination make funding an equally burdensome issue. According to the U.S. Conference of Mayor's annual reports on Brownfields Redevelopment, funding was the number one impediment to site cleanup in 1998, 1999, 2000, and 2003 (U.S. Mayors 1998 *et al.*).

Following the passage of the 2002 Federal Brownfields law, the Virginia General Assembly passed the Brownfields Redevelopment and Land Renewal Act. This law, which became effective on July 1, 2002, included provisions that limited liability, provided incentives for identifying property as a Brownfield, and established funding mechanisms, through grants and loans, to communities needing assistance with property assessment and development issues. The key player in this state initiative is the Virginia Department of Environmental Quality (VDEQ) through its Voluntary Remediation Program (VRP). Established in 1995, the VRP's annual budget is approximately \$500,000 and supports four full-time staff and three part-time staff.

METHODS

This research sought to identify the effects of redeveloping contaminated property by participating in Virginia's VRP. To accomplish this objective, ten sites that have participated, or are currently participating, in the state's VRP were identified. The sites were selected in cooperation with VDEQ staff and needed to have a broad range of contaminants and distribution throughout the state. The ten sites are presented in Table 1, below.

Table 1. List of VRP sites investigated.

VDEQ Site No.	Site Name (Location)	Acres	Location Setting	Former Use Theme	Redevelopment Theme
179	Former Ferguson Landfill-Home Depot 4631 (Newport News)	13.8	Urban	Landfill	Retail Center
180	Advance Mining System (Tazewell)	11.6	Rural	Industrial	Industrial
192	Birches Crossing-The Nature Conservatory (Arlington)	1.5	Urban	Service Station	Office Building
195	Singer Furniture (Roanoke)	25	Urban	Industrial	Industrial
197	Helms Concrete-Alexandria Toyota (Alexandria)	13	Urban	Landfill	Retail Center
229	Aileen Facility (Flint Hill)	30	Rural	Industrial	Industrial
231	Deanwood Property (Roanoke)	4	Urban	Mfg. Gas Plant	Industrial
235	Port Warwick (Oyster Point of Newport News)	150	Urban	Industrial	Urban Village
284	South Jefferson Project, Site Number 1 (Roanoke)	7.5	Urban	Industrial	Office Park
289	Rocketts Landing (City of Richmond)	50	Urban	Industrial	Urban Village

For each site, a case history, or synopsis, was developed from publicly available documents and interviews with site participants. Integral to each site synopsis was a limited economic and financial analysis. Considerable effort was placed on identification/determination of several key elements for each site, including, but not limited to, changes in property value before and after participation in the VRP, changes in tax revenue, job creation from redevelopment, estimated tax revenue from redevelopment, redevelopment funding/investment initiatives, and environmental damages and benefits.

By examining the economic, financial, and environmental information, a measurement of results emerged for each site. Each case history's data was then be used to identify two types of multipliers—one that assesses the economics of redevelopment and one that assesses the environment after redevelopment. These data were then used to calculate the program's cost effectiveness and its overall multiplier—a summary of the total environmental and economic benefits as measured against the VRP's annual funding.

For the purpose of this investigation, five cost indicators and five benefit indicators were selected. Cost and benefit information was also further subdivided into fixed and future categories. Future data were then adjusted according to time expectations as defined by projects. For example, if a property developer still needs to acquire additional parcels over the next few years, then property values of the identified parcels were adjusted by their historical change in property value to derive an estimate of future purchase value. All future data were adjusted to calculate future values for the relevant number of years: future project benefits were estimated

for a period of 20 years.

In general, natural resources are defined by their physical properties or by their use value. In the physical state, natural resources have either an extractive or an *in situ* value. Economic literature generally categorizes natural resources by use and nonuse values. Use value acknowledges both consumptive and non-consumptive uses and, by contrast, nonuse values include intangibles such as bequest and existence values. For these reasons, the most complicated of the economic and financial indicators developed for this research are the environmental costs and benefits associated with each project.

The National Research Council, which was organized by the National Academy of Sciences, acting through its Committee on Groundwater Cleanup Alternatives, has determined that cleaning up groundwater contaminated with hazardous waste to a “pristine” state may be impractical (NRC 1997). For some aquifers it might be preferable to contain or isolate the contamination to the extent possible. Consequently, valuation of affected resources becomes a useful tool. Water quality degradation in areas where substitutes are unavailable can dramatically increase the value of the water; conversely, a contaminated aquifer with “few prized attributes not available from substitutes” would not warrant a premium valuation (NRC 1997).

DISCUSSION

During redevelopment, difficulties can be encountered in various areas, including institutional constraints, conflicts, or conceptual problems associated with the site redevelopment effort; however, the uncertainties associated with site remediation and technical analysis have not been found to be a substantial hurdle. The case studies that were developed demonstrate that although a complete accounting for the various components is often difficult to obtain, quantifying some components can provide an important assessment of not only the business implications, but the environmental implications as well.

A site located in a rural setting, for example, has different economic opportunities than one in an urban setting. Redevelopment of the two rural properties studied has been driven by a creditors committee and a county administrator, respectively, both of whom had assumed control of the property after the bankruptcy of the site’s owners. The majority of the urban remediation efforts have predominantly been initiated by the purchaser of the site property. The research population is small; consequently, this difference between project initiators cannot be considered a trend. This difference is unique and may highlight differing needs for assistance between rural and urban participants.

Redevelopment efforts at the two rural sites have focused on attracting a replacement industry and have taken years to locate businesses and to return the site to the tax rolls. Conversely, the urban sites have generally been targeted by businesses seeking a location in an area and have not needed to be courted. In the rural setting, the existing buildings or structures are tangible assets that can be marketed to companies seeking to reduce the overall development costs when relocating or expanding. In the urban setting, property scarcity creates pressure for property redevelopment. VRP participants and site developers noted during interviews that they purchased the site property because it was available in a “hot market” (Freeman 2003, Flynn

2003, Taylor 2003). Historical aerial photographs were examined and indicated that these sites are among the last properties in the area to be developed.

Unlike rural property, redevelopment of these urban sites focused on overcoming site limitations to establish new facilities. Two of the eight urban properties that were examined were former landfills. Two of the urban parcels were former landfills and would generally be considered marginal property, if the market forces hadn't created such a high demand for building sites. Both of these parcels were redeveloped as retail centers that required large parking surfaces—one became a Home Depot while the other became an automobile dealership—that created a physical barrier to the contamination. To overcome groundwater intrusion into a new underground parking structure at its world headquarter building, The Nature Conservancy installed a groundwater collection and treatment system beneath its new headquarters in Arlington, Virginia. This treatment system was necessary to develop the 1.5-acre parcel.

Figure 1 is a graphical depiction of the economic costs and benefits associated with each of the sites researched. The total benefits associated with Deanwood Property may be overstated because the current occupant of the building vacated the site prior to the data collection phase. The building owner will need to find a new tenant and will incur financial losses if the building becomes vacant. Three of these projects, Port Warwick, Rocketts Landing, and South Jefferson, have not been fully developed, which will ultimately affect the benefits and costs of each. In general, all the projects have had, or are expected to have, a positive economic outcome.

Before any discussion of environmental costs and benefits, it must be stated that this investigation did not seek to measure how many lives were saved from remediation at the various sites. Nor did this research seek to quantify the existence or bequest values of environmental cleanup. Environmental costs are based on natural resource damages principles and for the purpose of this research are assumed to be replaceable. Figure 2 illustrates the environmental costs, or damages, from site activities and the benefits of remediation at each of the sites studied. Site characterization costs seem to be the most poorly tracked cost information across the studied sites; consequently, reliance on these figures is cautioned. Furthermore, benefit information can appear to be skewed higher where property is undergoing, or will undergo, subdivision, such as the urban village sites. For example, the willingness-to-pay for cleanup at Port Warwick has been estimated at approximately \$500,000; this figure is known to be low as the true costs were not reported and are likely to be much higher. The change in property value for the portion of the site where contamination was located is estimated to be near \$36.8 million, which clearly demonstrates that the stigma of contamination has not been attached to this property. It is probable that similar benefits may be realized at the Rocketts Landing and South Jefferson projects.

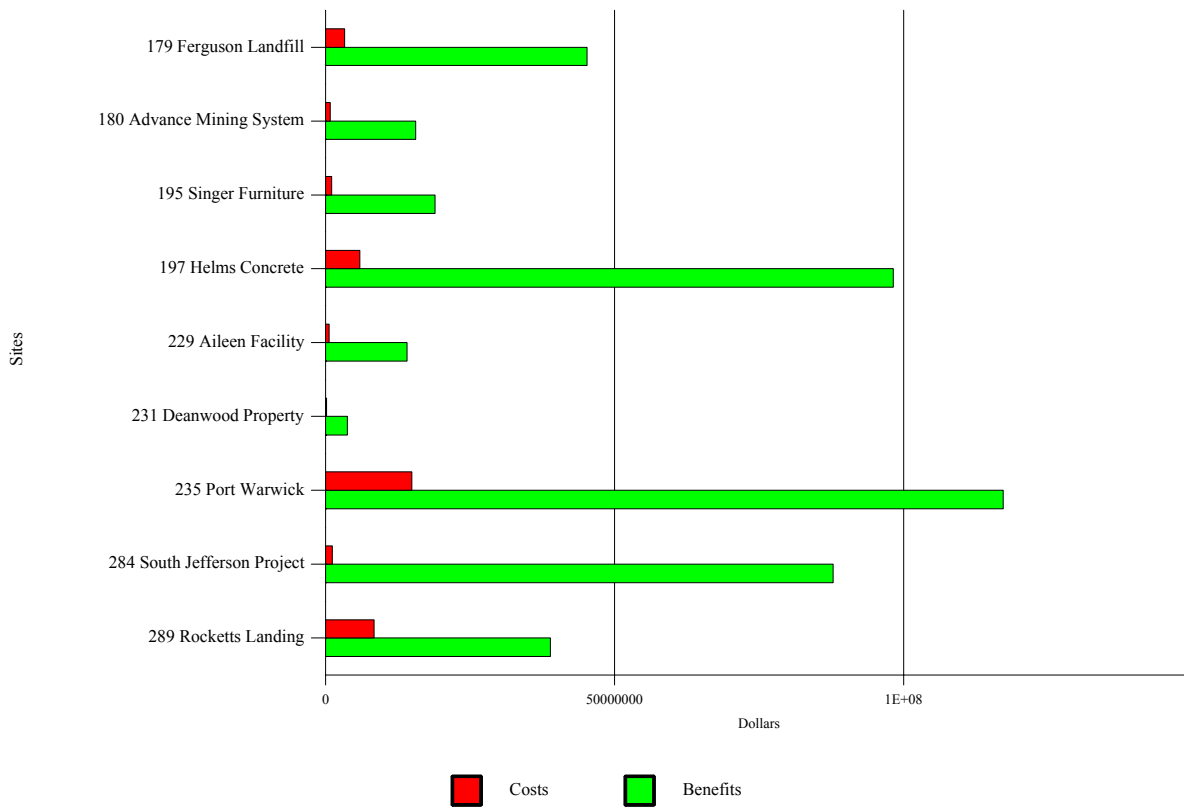


Figure 1. Project economic costs and benefits.



Figure 2. Project environmental damages and benefits.

It is also important to point out that two of the sites studied, Advance Mining Systems and the Aileen Facility, had negative total environmental benefits. This reflects a negative change in the property values at these two sites, which are both located in rural settings. Large environmental damages are found at the two landfill sites, Ferguson Landfill and Helms Concrete. These high costs are not unexpected given the site activities and demonstrate the expense of relocating a landfill. Similar high costs exist on the Deanwood Property and for Site Number 1 of the South Jefferson Project. Removal and replacement for all source materials at Rocketts Landing could be as high as \$30 million.

CONCLUSIONS

The redevelopment of idle or abandoned property through a Brownfields initiative is a viable solution to slowing urban sprawl. Studies have shown that for every one acre of Brownfields development, 4.5 acres of greenspace is saved (Wright 2002). Economic revitalization must include an effective way to transition property from uselessness to prosperity. Voluntary remediation is one vehicle needed to meet this goal. It is important to remember that Brownfields are inherently real estate transactions that have an environmental component. The voluntary remediation approach initiated by all states, and supported by the Federal government, seeks to bring equity to real estate transactions by creating a level playing field for stakeholders, including property owners, developers, and investors.

The VRP's goals of economic development and environmental cleanup are being achieved. The projects studied had an average project multiplier for economic development of about 21:1. There is a difference in multipliers for projects located in a rural setting as opposed to an urban setting. This difference was anticipated and does not appear to be a structural flaw within the program; however, the rural population of projects was not large enough to establish a structural trend. The program's goal of environmental cleanup can be measured by comparing environmental damages against environmental benefits. Overall the projects had an average environmental project multiplier of approximately 11:1. Based on this information, overall program multipliers have been calculated and are presented below in Table 2.

Table 2. Multiplier effects of VRP participation.

	Benefits	Multiplier
VRP*	\$654,390,234.11	1,308.78:1
Environmental**	\$35,611,296.30	71.22:1
Economic**	\$618,778,937.81	1,237.56:1

Notes: * As measured against the program's annual budget of \$500,000.

** As reported for the ten sites studied for this research.

This research indicates that the VDEQ's program is achieving, and probably exceeding, the agency's mandate of protecting human health and the environment. The voluntary nature of the program creates issues that warrant further discussion and evaluation. For example, the use of institutional controls prohibiting extraction of contaminated groundwater are useful in urban settings that have access to municipal water supplies. This approach, however, may have limitations in rural localities that depend on potable groundwater. Of the two rural sites examined, one site had groundwater contamination that was impacting a neighboring property

owner's well. Mitigation included treatment and the installation of a new well; however, the voluntary nature of the program limits the placement of groundwater restrictions on property not participating in the program even though the property owner is notified.

Site remediation cost information was difficult to obtain, which makes audits of individual projects difficult to conduct. In accordance with Virginia Code, program participants are required to submit an application fee that has some basis on expected remediation costs. The VDEQ is required to refund any monies due to the applicant, provided that the participant submits a detailed cost summary with the project's demonstration of completion report. Because the program is voluntary, the participant may waive its right to any refund and thus not provide documentation about costs. The individual case demonstrated that VRP participants have the potential to reap tremendous financial benefit. As a result of this benefit, and to ensure that an audit trail exists, program participants, at a minimum, should be required to provide detailed cost summary documentation just as they are required to provide laboratory data that supports environmental findings.

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Additional references were used in the full version of this paper, which is available upon request to the author.

PLANNING FOR LOW IMPACT DEVELOPMENT IN THE NORTHERN SHENANDOAH REGION OF VIRGINIA

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ABSTRACT

This project was funded by a Chesapeake Bay Small Watershed Grant through National Fish and Wildlife Foundation for the Northern Shenandoah Valley Regional Commission (NSVRC), under a contract with Engineering Concepts, Inc. of Fincastle, Virginia, with a subcontract to the Virginia Tech Center for Geospatial Information Technology (CGIT). The purpose of this study is to develop a Low Impact Development (LID) plan for the northern Shenandoah region of Virginia. LID is a strategy designed to decrease surface water runoff at the source by increasing infiltration, thereby preserving water quantity and water quality. Some common LID practices include bioretention, grass swales, green roofs, and narrow street designs. Recently, diminished water supplies due to rapid urbanization and drought have prompted many communities to consider LID for their area. The northern Shenandoah region of Virginia is a predominantly rural area that has experienced rapid urban growth along Interstate 81. The region also endured a five year drought from 1997 to 2002. Representatives from this region have expressed strong interest in using LID as a means to alleviate the impact of further urban expansion on their waters supplies. However, the northern Shenandoah region faces a unique challenge in implementing LID due to the nature of the karst and fractured rock that underlies the terrain. This study uses a Geographic Information System (GIS) to analyze data sources such as digital elevation models (DEMs), soil maps, land use maps, zoning maps, and geologic maps that

specifically locate karst, in order to determine which areas are best suited for LID and would be most appropriate for the region. This work will further encourage research and advancement of GIS studies as a tool for understanding the capabilities of stormwater management strategies such as LID. It will also contribute to local initiatives in the northern Shenandoah region to develop communities with sustainable design practices.

INTRODUCTION

Both water quality and water supply in urban areas are profoundly dependent on surface runoff and impervious surfaces. Surface runoff is the water that flows over the surface of the earth during a rainstorm. In highly urbanized areas, impervious surfaces greatly influence the amount and flow of surface runoff. Soon after rainfall, water flows quickly over rooftops, driveways, streets, and sidewalks, picking up pollutants and toxins along the way and depositing them in the nearest stream, increasing stream temperature and non-point source pollution.

In addition to supplying pollutants, impervious surfaces reduce the time of runoff concentration so that peak discharges are higher. Instead of reaching ground water reservoirs, the increased volume of water can cause flooding, and lead to erosion of stream banks which increases sedimentation and destroys habitat.

Impervious surfaces also contribute to the severity of drought. During a dry spell streams are replenished by the water stored by vegetation, soil, and groundwater reservoirs. However in an area of high impervious surfaces the potential maximum storage of these natural systems is lowered and streams run dry.

In recent years, rapid urbanization compounded with drought conditions have made stormwater management a critical issue. Planners and engineers are increasingly looking for better alternatives to many of the costly and inefficient means of controlling stormwater runoff. For example, in an effort to collect runoff from impervious surfaces in one site, planners are turning to strategies that concentrate development, as opposed to those that encourage large urban sprawls. "The best way to minimize the creation of impervious cover at the regional scale is to concentrate as much of it as possible in high density clusters in some subwatersheds (high levels of impervious cover-25% to 100%), so as to prevent other subwatersheds from exceeding the impervious threshold" (Center for Watershed Protection 2003). Additionally, engineers are relying less on costly end-of-pipe structural designs that usually do not provide opportunity for runoff to contact soils and vegetation (Roy *et al.* 2003), and are often poorly constructed and maintained. Instead, they are researching non-structural stormwater management tools that solve runoff at the source. In conjunction with urbanization, the costs and failures of many conventional stormwater designs have contributed to the development of LID.

LID is a stormwater management technique developed by Prince George's County Maryland in the 1990s. The concept is based on reducing stormwater runoff to predevelopment conditions by redirecting it away from impervious surfaces and increasing infiltration through soil and vegetation. The Environmental Protection Agency (EPA) recognizes four stormwater management practices as the basis for LID: bioretention, grass swales, vegetated roof covers, and porous pavements. Other LID techniques include rain gutter disconnect, rain barrels, rain

gardens, and urban forests. LID techniques are not intended to replace structural designs referred to as “Best Management Practices” (BMPs) (U.S. EPA 2000), but instead are intended to work in conjunction with them. These practices can be implemented into new development sites or urban retrofits. “Retrofit” is defined by the Center for Watershed Protection (CWP) as “structural stormwater management measures inserted in an urban or ultra-urban landscape where little or no prior stormwater controls existed.” In using LID, a retrofit would focus on inserting natural stormwater controls as opposed to “structural.”

Study Area

The northern Shenandoah region is located in northwest Virginia and extends from Shenandoah and Page counties to Frederick and Clarke counties, including Warren county and the city of Winchester (Figure 1). These predominantly rural counties are expected to face rapid urbanization as the Washington D.C. commuter corridor continues to expand (Orndorff and Harlow 2002). This rapid urban growth, in conjunction with recent droughts, has prompted communities to express concern over water availability and water quality conditions. The northern Shenandoah region is eager to implement sustainable development strategies such as LID in order to protect their resources. However, the unique karst terrain that dominates the landscape poses a significant threat to the success of many LID strategies.



Figure 1. Map of the study area.

Karst terrain is a geological formation composed of relatively soluble rocks such as limestone, dolomite, and gypsum. It is characterized by caves, springs, sinking streams, sinkholes, and solution valleys. The presence of karst enables groundwater to move rapidly through conduits and exits, making it highly vulnerable to groundwater pollution. Urban development of karst areas can threaten groundwater quality and the stability of the soil and bedrock, especially if sinkholes are abundant. Consequently, it is essential to understand the hydrologic network of the karst terrain when implementing urban development design strategies. Not only is it important to preserve fragile karstic features, it may also be necessary to avoid certain LID strategies such as rain gardens that promote infiltration. Increased infiltration over karstic terrain may collapse the land over sinkholes, enlarge conduits, or otherwise alter the hydrologic function of the

groundwater karst aquifers. At the same time, increased impervious surfaces in the surrounding basin can also increase stormwater runoff, thus impacting the groundwater system and warranting sustainable design strategies such as LID. The question becomes where and which LID practice should be implemented.

This study will use GIS and remote sensing to analyze data sources such as, soil maps, DEMs, zoning maps, land use maps, and geologic maps that specifically locate karst, in order to develop an LID management plan appropriate for the northern Shenandoah region.

METHODOLOGY

In this study, we identify suitable LID locations based on GIS data layers that either describe the character of the landscape, or a community's zoning regulations. The specific GIS data layers we used represent the most important variables impacting the suitability of LID sites based on previous LID studies. The relative importance of each factor is based on expert knowledge of LID.

The zoning layers were provided by the different counties and the city of Winchester. The soil data were provided by the Natural Resources Conservation Service (NRCS) and were recoded using a GIS so that the soil classifications reflect the permeability and hydrological function of the soil. Soil classes range from A to D with A being the most permeable and D being the least. The slope layers were derived from 10m digital elevation models (DEMs). The land use data come directly from the Virginia Gap Analysis project, a project initiated by the USGS to map the land use of the state of Virginia at the 1:100,000 scale. Finally, the karst data are based on a 1:250,000 scale map produced by the Virginia Department of Mineral Resources (DMME).

The specific LID practices chosen for this study were determined by the preferences of community representatives. While some LID practices such as bioretention and vegetative swales require specific landscape conditions in addition to flexible zoning standards, other practices such as reduced road widths and curb elimination are only restricted by zoning regulations. Consequently, an individual suitability map will be required for each LID practice. A single map may also describe the criteria required by multiple LID practices if the suitability requirements are similar. Table 1 shows which suitability factors are related to each LID practice.

After determining the suitability factors for each LID practice the next step is to evaluate the relative importance of each factor. In this study the importance of each factor is evaluated by a ranking system of 0 to 100, with 100 representing the most suitable and each rank being a multiple of 5. For example, in evaluating soil, soils classified as A are usually the most suitable and therefore ranked near 100, whereas soils classified as D are the least suitable and ranked between 0 and 20. The ranking assignments reflect the relative priority of each factor. Store and Jokimaki (2003) describe these as the sub-priority functions.

The next step is to combine all the sub-priority functions, or data layers that include rankings, into a final suitability map for each LID practice through arithmetic overlay analysis in a GIS. First, all the layers must be in raster format in order to technically analyze the combination of layers. Then, each sub-priority grid is multiplied by a weighted coefficient depending on how

important that layer is relative to other layers. Finally, the weighted sub-priorities are summed together using overlay analysis to result in a suitability map that incorporates soil, slope, land use, zoning, and karst. The suitability index is reflected in the combined rankings, a range from 0 to 100. As a final step the multiple rankings are reclassified using natural breaks, resulting in a suitability index in which values range from 1 (unsuitable) to 4 (most suitable).

Table 1. Site constraints of LID practices.

LID PRACTICE	SUITABILITY FACTORS				
	soil	slope	land use	zoning	karst
bioretention	x		x	x	x
biofilters, buffer strips	x	x	x	x	x
infiltration trenches	x	x	x	x	x
pervious pavement	x		x	x	x
vegetative swales	x	x	x	x	x
rain barrels, cisterns			x	x	
downspout disconnections				x	
reduced road widths			x	x	
curb and gutter elimination			x	x	
curb cuts				x	
surface roughness technology	x	x		x	
green roofs			x	x	

RESULTS AND DISCUSSION

This study is still in progress, so our results are forthcoming. However the implications of a LID site suitability map at the regional level can be discussed here. The goal of the project is to provide guidelines to those who choose to use LID in developing the northern Shenandoah region. The site suitability maps allow planners and developers to rule out areas that are not appropriate for LID. We recommend that LID practices not be implemented in the suitable areas until a site-specific assessment of the landscape has been completed. The quality of the original input layers presents some uncertainty in the final results. Therefore, the maps should only serve as a general guideline and a stepping stone for further LID site suitability maps at the local level in which field testing is highly recommended.

There are several sources of uncertainty that can compromise the predictions of a GIS model. Those relevant to this model are scale, quality of input data, temporal quality, boundary issues, choice of parameters, and categorization of suitable attributes. As GIS models become more complex and rely on many data layers from different sources, these types of uncertainties can limit the value of the model (Bolstad and Smith 1992).

Some of the uncertainties inherent in the LID model that warrant concern are exemplified by the karst data. David Hubbard (2001), the leading researcher behind the karst data provided by the DMME, notes that the karst data are useful in that they display the relative range of sinkhole development, however he emphasizes that the scale and resolution of the map make it inherently

problematic to use for site-specific management plans. The limited resolution of the maps can be explained by the inherent difficulties in mapping karst. Field checking of the karst data is very time consuming and remote sensing techniques are restricted by the type and amount of karst formations that are identifiable. Hubbard (2001) primarily relied on stereoscopic viewing of panchromatic aerial photos to locate sinkholes because they are visible surface features, unlike underground caves and springs. Thus, the karst data are limited to a small scale (1:250,000); and only locate sinkholes, making the karst data less reliable. The data are also temporally limited because the aerial photos on which they are based are from the 1980s, while karst formations are known to change frequently.

The soil data also contribute uncertainty to the LID model due to the boundaries that define the soil classes. The soil classes are defined by the permeability or infiltration rate of the soil. We identified four soil classes ranging from A (most permeable) to D (least permeable). The classes are based on factors that affect infiltration rates such as slope and soil composition, *e.g.*, clay and/or sand content. It was difficult to distinguish several of the soils that fell somewhere between classes B and C, therefore there is some ambiguity as to whether the classes we chose for these borderline soils are the most appropriate.

Clearly, many uncertainties arise when developing an LID model using spatial data that incorporate errors from various sources and at various scales. Similar uncertainties arise when developing any GIS model that addresses natural resources or environmental analysis due to the complexity and variability of natural processes. Fortunately, there are methods outlined in the literature that provide techniques for alleviating such uncertainties. For example, a common way to assess the uncertainty of a GIS model is to adjust the parameters of the model to determine a range of outcomes rather than just one. This procedure can enable an evaluation of the most probable outcomes. We intend to perform an uncertainty analysis for the karst data layer, and a sensitivity analysis for the soil data layers in order to address some of the ambiguity inherent in our weakest data layers.

We will evaluate the uncertainty in the karst data by comparing Hubbard's (2001) original sinkhole maps, identified on 1:24,000 maps, with an automated GIS technique to identify sinkholes. Comparing the results of both methods will enable us to evaluate the precision of our karst data. If available, we will also utilize LiDAR data to locate sinkholes in order to apply a third method to our comparative analysis. If all three methods output significantly similar results then we can eliminate some of the uncertainty in our karst data.

To address potential inaccuracies in the soil data, we will perform a sensitivity analysis. Many of the uncertain borderlines in the soil data layers are due to soil classes that did not clearly fall into one permeability category or the other. To test how sensitive the model is to these borderline soils we will reclassify them to their alternate permeability class. The difference in the two models will enable us to evaluate the uncertainty in our soil data and enumerate alternative outcomes to our model.

Despite problems with inaccuracies of data, using GIS to support and promote LID will dramatically improve our ability to preserve and protect our water resources during times of expanding urbanization. Data quality is an essential component of GIS analysis, although it is

not often mentioned in the GIS or LID literature. Our focus on data inaccuracies and scale issues in LID models will help other GIS managers and users to more appropriately and effectively apply available data to similar LID models. This study will also contribute to the local initiatives of the northern Shenandoah region to develop their communities with sustainable design practices.

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ESTIMATING STREAM LENGTH AND SINUOSITY USING GIS

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KEY WORDS: stream length, sinuosity, terrain analysis, DEM, NHD, resolution

ABSTRACT

Several GIS datasets were used to estimate stream length and sinuosity as a function of data resolution and type. Stroubles Creek, in Montgomery County, Virginia, was used as the test case for this preliminary analysis. Data used include USGS 10m DEMs, the National Hydrography Dataset (NHD), and two data sources from the Virginia Base Map Program (VBMP) 2002 aerial photography project – a vector stream feature (DTMhyd), and flow networks defined from the digital terrain model (DTM) data. Topographically defined flow networks were created from the DEM data at 10m, 30m, and 90m grid resolutions, and from the DTM data at 10ft, 30ft, and 90ft. Twenty four analysis reaches were defined from the 7.5min topographic map (DRG) that appeared to be consistent in character. A ‘valley’ distance was defined for each reach and used as the basis for calculating a sinuosity for each of the stream networks generated from the different source data sets. The DTMhyd provided the best representation of the stream channel since it was created by digitizing from high resolution aerial photographs, and in general, had the highest sinuosity values. Sinuosity for the derived stream networks decreased as data grid size increased. Sinuosity for the NHD data was similar to that of the high resolution grid flow networks and better than the low resolution flow networks. While the actual position of the stream network varies distinctly between data sets, all provided a reasonable representation of the general valley profile with the exception of the DEM 90m network. The DTM data from the Virginia VBMP data appears to offer improved digital representation of stream networks for specific applications.

INTRODUCTION

The length and sinuosity of channels are important parameters for hydrologic analysis and modeling. Sinuosity is defined as the ratio of straight-line distance (or valley length) to actual stream length, thus is a measurement of the degree to which a stream meanders down its valley. Sinuosity is a common parameter in stream characterization studies (Arya 2002) and restoration design (Rosgen 1994). The best estimate of site-specific stream parameters is through field surveying, but this process is very time consuming. Analysis using a Geographic Information System (GIS) and digital data sources can be a much simpler approach that may provide

reasonable estimates of sinuosity at desired locations. McCleary *et al.* (2002) computed slope and sinuosity for small streams in the west central part of Alberta using GIS derived values. In comparison with field measurements, the GIS results underestimated stream sinuosity for streams within a watershed area of 20 sq. km.

In this study, GIS analysis was used to calculate sinuosity using several different sources and resolutions of data that can be used to define the stream network in a GIS. This study is a preliminary analysis to explore the feasibility and limitations of calculating sinuosity using available digital data sources, and to examine the effect of data resolution on sinuosity. A secondary goal was to explore the accuracy of the DTM data from the Virginia Base Mapping Program (VBMP) (VGIN, 2002) for defining streams.

METHODS

Stroubles Creek in Montgomery County Virginia was used as the basis for the study. The analysis considered the main channel, from the outlet of the Duck Pond on the Virginia Tech campus (800 ha catchment), downstream to Route 685 which is near its confluence with the New River (6000 ha catchment). This study area falls on two 7.5minute quadrangle map sheets – Blacksburg and Radford North. Four sources of data were used to define the stream network, with three different resolutions used for the two DEM data sets. A brief description of the data used, the name used to refer to the data set throughout this paper, and a reference to the source of the data is given in Table 1. Data sets were projected, as required, to a common coordinate system of NAD-83, UTM zone 17N. Analysis was conducted using ArcGIS 8.3 with Spatial Analyst and 3D Analyst.

Table 1. Data sets used to create stream networks for Stroubles Creek.

Name	Description
NHD	National Hydrography Dataset – High resolution dataset, equivalent to 1:24000 scale. This vector data follows very closely the blue-line streams on the topo map (DRG) Source: http://nhd.usgs.gov
DTMhyd	DTM (Digital Terrain Model) data from VBMP – the vector hydrography (DTM breakline feature code 44) Source: http://www.vgin.vipnet.org
DTM10f DTM30f DTM90f	Flow networks derived from different resolution grids (10ft, 30ft, 90ft) using ArcGIS terrain analysis functions. The VBMP DTM data was used to first create a TIN in ArcGIS which was resampled to the specified raster data set resolution. Source: VBMP DTM data (VGIN 2002)
DEM10 DEM30 DEM90	Flow networks derived from different resolution grids (10m, 30m, 90m) using ArcGIS terrain analysis functions. USGS 10m DEM (Level 2) was resampled (Nearest Neighbor) for 30 and 90m grids. Source: http://www.gisdatadepot.com

NHD Data Preparation

The NHD high resolution data for the New River basin was accessed from the USGS in June, 2004. The features for Stroubles Creek were selected and extracted for display and analysis without further processing.

DEM Data Preparation

The 10 meter DEMs of Blacksburg and Radford North were used for the study area. The DEMs were converted from SDTS format into raster grid using Arc Toolbox, and merged to form a single data set. The 10m data were resampled to 30m and 90m resolutions using the center value (nearest neighbor) by setting the cell size to the desired resolution and calculating a new grid. Each of the grids were then processed using the routine procedures for generating a flow network of filling sinks, defining flow direction, and creating the flow accumulation grid. The Hydrology menu 'Stream network as feature...' function was used to create a vector feature for each grid flow network.

DTM (VBMP) Data Preparation

A total of 148 DTM data sets (tiles) were used in defining the study area. The line and point feature classes were extracted from the DTM data sets (dgn file format). The breakline feature ID 44 was extracted, merged and projected to create the DTMhyd data set. The hydrography feature in the VBMP DTM data is available as a derivative of the process of creating the DTM for orthorectification of the aerial photography. The breaklines and point data were input to 3D Analyst and a TIN surface model created. The TIN was then converted to DEMs at resolutions of 10, 30, and 90 feet (source units of the DTM data). The 30 and 90 ft resolutions were chosen for comparison with the DEM data resolutions of 10 and 30m, and the 10ft grid provided an assessment of a finer (approximately 3m) resolution grid. Following creation of the three raster data sets, stream networks were generated following standard procedures, and the resulting shapefiles reprojected to NAD-83 UTM 17N.

Data Samples and Analysis

Twenty six reaches along the stream were identified from the DRG that avoided major confluences, road crossings, or other obvious disturbance (Figure 1). Problem areas in the data sets such as 'fills' in the DEMs and breaks in the DTM breaklines were avoided in selecting the sample reaches and resulted in not using data from reaches 2 and 21. To provide a measure of 'valley length' for calculating sinuosity, a linear feature defining the general line of the stream valley was digitized using the DRG as a background at 1:12000 on-screen scale. The reach polygons were used to intersect the eight stream features and the 'valley' length feature. For two reaches (16, 26), the DEM90 path crossed the reach polygon obliquely and was judged to not be a reasonable representation of the reach length so these values were not used. The length of each stream feature was calculated in meters, and the lengths exported for tabulation and analysis in Excel.

A second sampling procedure was used as a way to evaluate analysis procedures. Seven points were located along the stream and circular buffers of 50, 100, 150, 200, 250, and 300 meters defined for each point. These seven sample locations with six sample sizes (100 – 600m diameters) were used to examine the effect of sample zone length on sinuosity calculations. By visual examination, the diameter of each sample zone was considered a reasonable estimate of the valley length with the exception of sample points P3, P4, and the outer ring of P2, for which reference distance was defined by manual measurement using the GIS measure tool. An example of the spatial configuration for one sample point is shown in Figure 2.

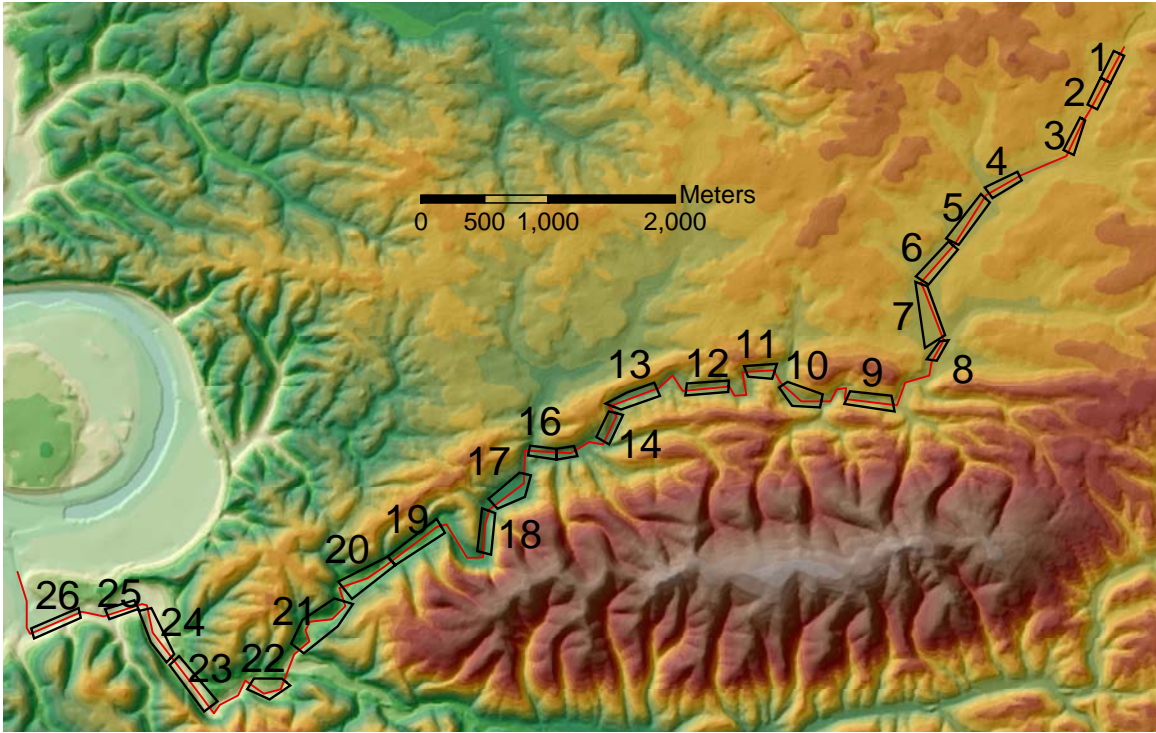


Figure 1. Sample reaches on Stroubles Creek used for evaluating sinuosity.

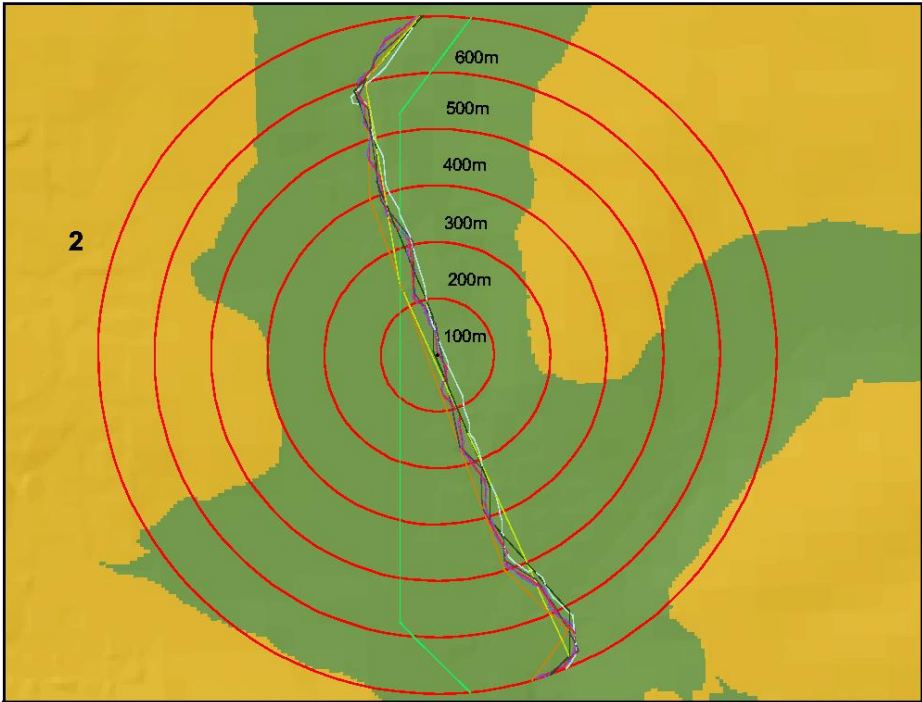


Figure 2. Point sample location P2 showing radial sample distances and flow paths for different data sets.

Analysis

Sinuosity was calculated for each data type by reach, using the ‘valley length’ vector as the base length. Comparison of the lengths for different stream reaches can also be made directly. In addition to the numeric analysis, visual comparison of the paths of the various derived stream networks indicates limitations and weaknesses of some data sets. In this analysis we assume that the DTMhyd data represents the best digital estimate of the stream geometry since it is developed by digitizing from high resolution aerial photographs.

RESULTS

Differences in the flow paths generated by the different data sets and differences in grid resolutions can be seen in Figures 2 and 3. As would be expected, similarities in flow paths can be seen between data sets having common backgrounds: a) DTMhyd and DTM grid-based flow networks, and b) NHD and DEM10 and DEM30. Since the creation of the TIN model is based in part on the DTMhyd as a breakline, strong similarity is expected between the DTMhyd feature and the derivatives of the TIN elevation model – the DTM10f, DTM30f, and DTM90f.

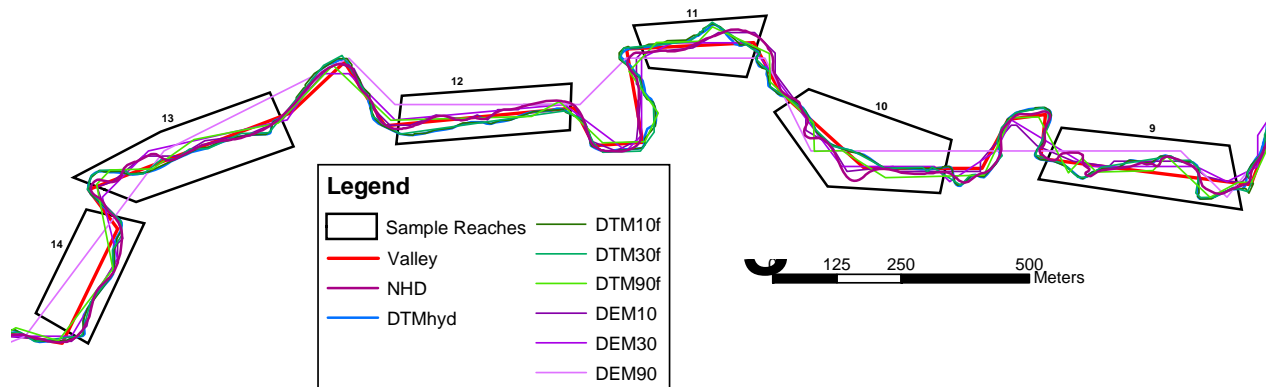


Figure 3. Section of Stroubles Creek showing six of the sample reaches and the stream networks from different data sources.

For both the DTM and the DEM data sets, the visual assessment of the stream paths and the comparison of the sinuosity data show, as expected, the generalization of stream features with increasing grid size in the base data. As expected, larger grid sizes result in greater smoothing of the derived flow paths, leading to shorter flow distances and simplification of the stream geometry. Note that while the 30m data sets (DEM30, DTM90f) still follow the general curvature of the stream valley, the 90m grid, DEM90, has significant smoothing and deviations from the stream valley shape. The limitations of the 90m DEM can be further seen in the fact that more than half of the sinuosity values for DEM90 (Table 2) are less than one, indicating that reach length is less than the ‘valley’ length thus demonstrating ‘short-cutting’ of the flow path.

Table 2. Sinuosity for 8 data sources for 24 reaches (Sorted by DTMhyd).

Reach ID	Valley dist (m)	Sinuosity (m/m)							
		DTMhyd	NHD	DTM10f	DTM30f	DTM90f	DEM10	DEM30	DEM90
9	371	1.25	1.27	1.26	1.25	1.25	1.03	1.01	1.11
8	174	1.25	1.29	1.22	1.15	1.04	1.06	1.10	0.65
17	365	1.24	1.12	1.24	1.23	1.03	1.08	0.99	0.98
4	276	1.17	1.06	1.12	1.07	1.01	1.04	1.04	1.00
15	158	1.16	1.08	1.13	0.98	1.06	1.04	1.01	1.01
5	428	1.13	1.06	1.06	1.03	1.00	1.05	1.02	0.99
11	229	1.12	1.04	1.14	1.10	1.09	1.04	1.00	0.91
20	443	1.11	1.05	1.12	1.12	1.04	1.00	1.00	1.04
13	383	1.10	1.09	1.10	1.12	1.05	1.03	0.99	0.92
7	444	1.10	1.03	1.08	1.05	1.03	1.03	1.01	1.09
19	478	1.09	1.04	1.11	1.09	1.03	1.05	1.04	0.99
6	368	1.08	1.04	1.05	1.02	1.02	1.02	1.02	0.99
3	286	1.06	1.02	1.09	1.02	1.02	1.02	1.02	1.00
23	462	1.06	1.03	1.06	1.07	1.03	1.04	1.03	1.01
16	215	1.06	1.06	1.08	1.05	1.07	1.06	1.09	0.15
1	238	1.06	1.01	1.05	1.03	1.12	1.03	1.00	0.99
14	258	1.06	1.03	1.05	1.05	1.01	1.02	1.01	0.94
25	244	1.05	1.02	1.05	1.05	1.05	1.04	1.02	1.01
22	342	1.04	0.97	1.03	1.05	1.04	1.00	1.02	0.95
12	337	1.03	1.04	1.04	1.01	1.04	1.06	1.00	0.98
24	435	1.03	1.03	1.04	1.03	1.02	1.05	1.02	1.03
18	335	1.03	1.01	1.04	1.03	1.04	1.02	1.02	1.04
26	404	1.02	1.01	1.04	1.03	0.86	1.03	1.02	0.28
10	351	1.00	1.16	1.00	1.02	1.09	1.08	1.05	1.06
Count of number of values by sinuosity range									
1.10 - 1.30	10	4	9	6	2	0	1	1	
1.05 - 1.09	8	6	9	8	6	8	2	2	
1.02 - 1.04	5	10	5	8	12	14	11	3	
1.00 - 1.01	1	3	1	1	3	2	8	5	
< 1.00		1		1	1		2	13	

Sinuosity values tabulated in Table 2 can be examined as a whole by considering the distribution of values. Recognizing that the DTMhyd data is the best graphical representation of the stream and that the other data sets incorporate smoothing, this data set is the reference in the absence of field survey data. The DTMhyd data has the highest values and the greatest number of high values of sinuosity for the sample reaches. The DTM10f stream network is closest in range and distribution of sinuosity values, and has the best correlation on a reach-by-reach basis with the DTMhyd data (Figure 4). The DTM30f has the next highest sinuosity values, followed by the NHD and DTM90f. Note that while the overall distribution of values may be similar, the actual values on a reach by reach basis may be different. The DEM10 results were unexpected in that no sinuosity value is greater than 1.1. The DEM10 network represents the valley well, with no values less than 1.0, but otherwise the values provide little discrimination between the reaches.

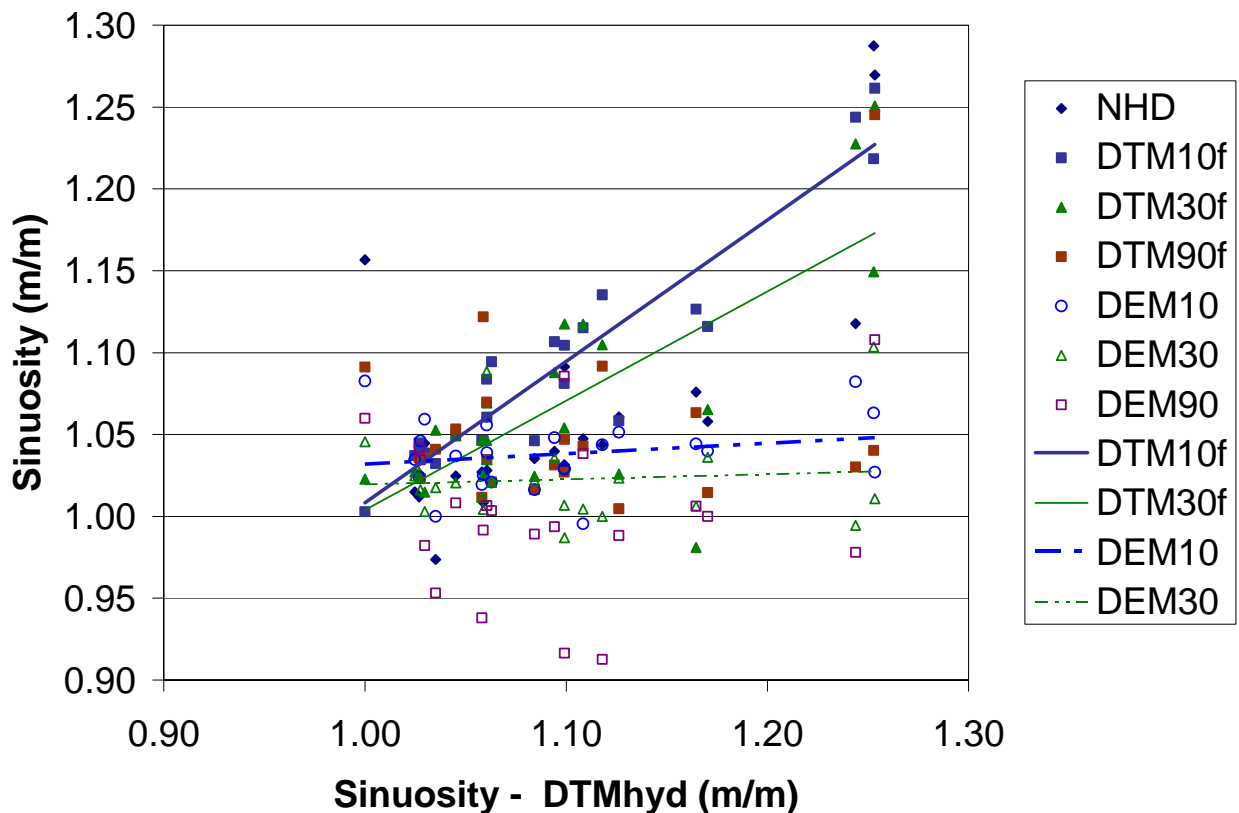


Figure 4. Sinuosity of six derived stream networks and the NHD plotted against the sinuosity of the DTMhyd (VBMP hydrography feature).

The DEM30 data represent the valley (rather than sinuosity) well, with only 3 values 1.05 or higher and 18 of the values falling between 0.99 and 1.02. With 13 of its 24 values less than 1.0, the DEM90 does not adequately represent even the valley shape for Stroubles Creek.

Figure 4 shows the sinuosity of the different stream networks plotted against the sinuosity of the DTMhyd. The assumption here is that the stream digitized from the aerial photographs is reasonably the best representation of the actual stream. In the figure, each stream reach (sample set) is a vertical line of points. In addition to the data, lines representing linear least squares fits to four of the data sets are shown. The lines are included to help illustrate the patterns in the data. Again, the strongest positive relationship is between the DTMhyd data and the DTM10f and DTM30f sinuosity. Other data sets have greater scatter in the correlation of the sinuosity values, and the DEM and DTM data series have increasing scatter in the data as grid size increases.

For the DTMhyd data, the sorted order in Table 2 indicates that for this stream, the sinuosity is not solely a function of the position along the stream. In the sorted sinuosity list, sample locations higher on the stream tend to be towards the top of the list (higher sinuosity), and sample locations in lower reaches tend to be towards the bottom of the list (Table 2), but there is significant variation, and no clear relationship between sinuosity and position along the stream.

Selections of the point radial sampling data are shown in Table 3. General results from this data set were consistent with observations from the larger data set. Data from the other four points and for the additional data sets had results that were similar in characteristic to the data shown here. The focus of this sampling procedure was to look at the effect of sampling distance, and a selection of sample points and data sets illustrate the range of responses observed. For P6, sinuosity generally increased with reach length for all data sets. For P5, sinuosity decreased as reach length increased for the DTMhyd data, but varied much less for the other data sets. The P0 data are typical of several points which had consistent values across reach length, while the basic range of values was different between the data sources. The conclusion from this data is that there is no consistent trend in sinuosity due to sample reach length, but rather, that sinuosity is a function of local data and is sensitive to the specific location and length of the sample zone selected. The point and radius approach to sinuosity calculation thus may be useful, realizing that results may be sensitive to the location of the point and the radius selected.

Table 3. Sinuosity values for sample points P0, P5, and P6 at six reach lengths.

Reach Dist (m)	Point P0			Point P5			Point P6		
	DEM	DTM	DTM	DEM	DTM	DTM	DEM	DTM	DTM
	10	30f	hyd	10	30f	hyd	10	30f	hyd
100	1.03	1.01	1.09	1.06	0.97	1.26	1.03	0.99	1.03
200	1.05	1.01	1.08	1.06	1.08	1.22	1.02	1.03	1.07
300	1.03	1.02	1.11	1.05	1.06	1.20	1.05	1.04	1.11
400	1.04	1.05	1.13	1.04	1.06	1.17	1.05	1.11	1.14
500	1.04	1.04	1.11	1.04	1.05	1.15	1.08	1.15	1.16
600	1.06	1.05	1.10	1.04	1.05	1.14	1.10	1.17	1.18

CONCLUSIONS

GIS data were used to evaluate stream sinuosity using various data sources for Stroubles Creek in south-west Virginia. The stream network generated from a gridded elevation surface that comes from the VBMP DTM data provided the best match of the stream digitized from the VBMP aerial photographs. Increasing grid size results in significant reduction in the length of the stream features. The 10 foot DTM grid produced a stream network that was most similar to the DTMhyd stream, while the 30 foot DTM grid had stream lengths comparable to the NHD data. The 10m DEM data had lower sinuosity than the NHD (from which it shares common background) and has weak correlation with the DTMhyd sinuosity.

For similarly sized grid data (DTM30f and DEM10; DTM90f and DEM30), the DTM data have higher sinuosity, reflecting the inherently higher resolution of the parent data (DTMhyd versus DEM10). The NHD had sinuosities only slightly higher than the DEM10 flow network and similar to the DTM90f. These results support the recommendation of Rosgen (1998) who says that sinuosity cannot be determined from 1:24000 scale topographic maps (representative of the NHD and DEM10), but can be estimated from large scale aerial photographs.

From these preliminary results, the VBMP DTM data show promise of offering a higher resolution data set for defining stream networks, using either the vector feature or derivatives

from generated elevation models. The hydrography feature from the DTM data provides a high resolution feature that should provide a good measure of sinuosity. Flow networks from 10m DEMs offer a well-defined stream network but without adequate definition for characterizing sinuosity. The 30m DEM offered little discrimination for sinuosity, but for this stream, provided a good representation of general valley shape. The 90m DEM was too coarse to adequately represent even valley shape.

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VULNERABILITY OF STREAMS TO EPISODIC ACIDIFICATION IN SHENANDOAH NATIONAL PARK

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One of the largest resource-management problems faced by Shenandoah National Park is air quality, in particular acid rain and its effects on streamwater quality. Paine Run, in Shenandoah National Park, is currently listed on American River's most endangered list due to the effects of acid deposition. Other streams in Shenandoah National Park likely are similarly threatened. Baseflow streamwater chemistry is influenced by bedrock geology, but during acidic precipitation events, other factors, such as topography, also influence streamwater chemistry.

The goal of this study was to develop a vulnerability map that predicts the frequency, duration, and intensity of episodic acidification events in Shenandoah National Park streams. Researchers at University of Virginia have continuous discharge data and discontinuous acid-neutralizing capacity (ANC) data for three streams in the park. We used a transfer function—the natural log of discharge regressed against ANC—to develop a time-series model of ANC of streams, which allowed us to predict ANC from hourly discharge data. We used the model to characterize the magnitude, frequency, and duration of ANC excursions from baseflow concentrations. Models were extrapolated to nearby catchments with limited data. Our vulnerability map will provide important information to better understand a critical stress to the park's aquatic ecosystem.

DEVELOPMENT OF THE NATIONAL WATERSHED BOUNDARY DATASET FOR VIRGINIA

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KEYWORDS: national watershed boundary dataset (NWBD), hydrologic unit, basin, sub-basin, watershed

ABSTRACT

An effort to develop and maintain a seamless system of hydrologic unit boundaries on the national level has been in effect since the 1970's. The initial attempt divided the United States into 21 Regions, 222 Sub-regions, 352 Accounting Units, and 2,149 Cataloging Units which are known as the first, second, third and fourth level units, respectively. Unfortunately the smallest of these cataloging units, around 448,000 acres in size, were too large for Virginia's applications. Therefore, Virginia went on to develop a more detailed system that would meet the needs of the state's environmental agencies. The first version of this system was developed in 1989 with revisions in June of 1995.

In 2001 the NRCS, USGS, EPA, and other federal agencies teamed with the Subcommittee on Spatial Water Data (part of the Advisory Committee on Water Information (ACWI)) and the Federal Geographic Data Committee to develop new standards for establishing the seamless 5th and 6th order units of the National Watershed Boundary Dataset (NWBD). These standards, which call for 5th order units that are 40,000 – 250,000 acres in size and 6th order units that are 10,000 (but no less than 3,000) – 40,000 acres in size, have prompted Virginia's latest effort to revise the hydrologic unit system in use since 1995.

This paper addresses Virginia's latest revision of the hydrologic unit boundaries. It gives a brief history of Virginia's hydrologic unit system and notes the significant differences between the previous and most recent version of the *Federal Standards for Delineation of Hydrologic Unit Boundaries*. It also outlines the updating process and highlights the major differences between the 1995 hydrologic unit boundaries product and the new NWBD dataset for Virginia. The goal of this paper is to inform interested parties of the latest updates to the state's hydrologic system and direct them to where the data can be obtained.

INTRODUCTION

In 1989 the Virginia Department of Conservation and Recreation (DCR), the USDA Soil Conservation Service (now the Natural Resource Conservation Service (NRCS)), and the Information Support Systems Laboratory at VPI completed the initial subdivision of the 4th order hydrologic units (cataloging units) in Virginia. The incentive to do so was the increasing use by these agencies of hydrologic unit codes to geospatially reference water quality activities and indicators. What was needed though was a level of hydrologic units that were at a finer resolution than the existing 4th order units. The 6th order units developed in 1989 became known as the 14-digit hydrologic unit system – a reference to the number of digits in the official unit defining codes (USDA and DCR and ISSL 1991). In Virginia a unique 3-character string was assigned to each unit, proceeding sequentially in each basin from outlet to source. This 3-character coding system in effect replaced the 14-digit codes for referencing purposes in the state.

The 1989 effort was updated in 1995 to comply with published federal standards on the mapping and digitizing of hydrologic units (USDA 1992, USDA 1995), which required some major modifications to be met. The 1995 product was also influenced by the increased use of hydrologic unit codes for referencing by the Virginia Department of Environmental Quality (DEQ). What was developed in 1995 immediately became the new official 6th order units for Virginia (USDA and DCR 1995) and was the basis for creating the original 5th order hydrologic units in Virginia. The 5th order units of this system were referred to as the 11-digit hydrologic unit system. The 6th order units were reassigned 3-character strings proceeding sequentially in each basin from source to outlet (the opposite of the 1989 system due to standards changes). The 6th order units have been used extensively over the past 9 years to catalogue all types of data at DCR and by federal and other state environmental agencies as well.

In 2001 the NRCS, USGS, EPA, and other federal agencies teamed with the Subcommittee on Spatial Water Data (part of the Advisory Committee on Water Information (ACWI)) and the Federal Geographic Data Committee to develop the new *Federal Standards for Delineation of Hydrologic Unit Boundaries* (USDA 2002). The new standards are for establishing seamless 5th and 6th order hydrologic units for the entire U.S. The digital product resulting from the delineation and capture of these new units is the National Watershed Boundary Dataset (NWBD). There are several major differences between the new standards and those proceeding them.

(1) They call for smaller 5th and 6th order units. Sixth order units of the 1995 product averaged over 54,000 acres in size. The new requirements by order are in Table 1.

(2) Addition of specific attributes to polygons and arcs in the final digital product. This includes unit modifiers (dam, karst, drainage ditches, *etc.*) and types (standard, frontal, water, *etc.*), unit names, line source, official unit codes at the 4th, 5th, and 6th order, and the official codes of the 5th and 6th order units downstream.

(3) Water side delineations of frontal units are defined to the toe of the shore face. For Bay waters this boundary has been set at a depth of 10 feet. For the Atlantic this boundary has been set at a depth of 30 feet. Both are based on research regarding where wave action first affects the shoreline.

(4) Unit coding for 5th and 6th order units has changed from requiring 11 and 14 digits to requiring 10 and 12 digits respectively.

(5) Order names have been changed from those established in the 1970s to a new system as shown in Table 1.

(6) The Atlantic Ocean out to the three nautical mile territorial limit is now partitioned into 5th and 6th order hydrologic units.

Table 1. New versus old hydrologic unit system references.

ORDER	NEW DIGITS	OLD DIGITS	NEW NAME	OLD NAME	UNIT SIZE
1	2	2	Region	Region	Avg. 177,560 sq. miles
2	4	4	Sub-Region	Sub-Region	Avg. 16,800 sq. miles
3	6	6	Basin	Accounting Unit	Avg. 10,596 sq. miles
4	8	8	Sub-Basin	Cataloging Unit	Avg. 703 sq. miles
5	10	11	Watershed		Range: 40,000 - 250,000 acres
6	12	14	Sub-Watershed		Range: 10,000 - 40,000 acres

METHODS

DCR began developing Virginia's portion of the NWBD in 2001. As part of the development process, proposed 6th order units of the NWBD were delineated so as to preserve as much of the 1995 boundaries as possible. This was done in order to make the transition between the two systems less complicated. Accordingly, many NWBD 6th order units are subsets of the 6th order units in the 1995 system. Modifications to the previous boundaries were still occasionally necessary, however, in order to be in compliance with the new standards or even just to fix previous flaws.

All delineations were recaptured from heads-up (on screen) digitizing, including those previously existing in the 1995 system. The lines were captured from digital raster graphics (DRG) of the latest 7.5-minute USGS quadrangle maps and from NOAA charts. Updated jurisdiction boundaries were also captured during this process. After completion of the 6th order unit linework, the 5th order system was created by aggregating 6th order units.

The next step in the process involved coding the new units. In accordance with the official coding scheme for the NWBD, the 5th order units were assigned a ten-digit code, with the first eight digits being the code for the sub-basins (4th order units), and the last two a unique sequential number based on drainage flow. Sixth order units were given a twelve-digit code,

with the first ten digits being the code for the watersheds (5th order units), and the last two a unique sequential number based on drainage flow. Both the 5th and 6th order units were sequentially coded beginning upstream and progressing to downstream units. The general coding order is main stem followed by tributary, with no unit flowing into a unit with a lower reference code.

The reuse of the same 3-character string structure to uniquely code 6th order units of both the 1989 and 1995 products unfortunately produced a couple of years of questionable unit referencing. Unit coding had reversed between those systems, such that a watershed coded “A33” in 1989 became “A01” in 1995. It was not easy to catch unit referencing in 1995 and 1996 however that mistakenly continued to use the 1989 code set. Due to that experience, and the knowledge that more than two digits would be needed to code all of the NWBD 6th order units in a basin as previously defined, DCR developed a new 4-character internal coding scheme for the 6th order units of the NWBD. This 4-character code will replace the 3-character code of the past. The first two characters of the new code are based on the main stream name in the basin, or portion of the basin, where the unit is located (Table 2). The two digits that follow these codes are a numbering scheme based on the drainage flow, using the same criteria mentioned above.

Table 2. Basin portion of the new internal 6th order unit codes.

AO - Atlantic Ocean	JL - James, Lower	RL - Roanoke, Lower
AS - Albemarle Sound Coastal	JA - James-Appomattox	RD - Roanoke-Dan
BS - Big Sandy	JR - James-Rivanna	TC - Tennessee-Clinch
CB - Chesapeake Bay	NE - New	TH - Tennessee-Holston
CU - Chowan, Upper	PU - Potomac, Upper	TP - Tennessee-Powell
CM - Chowan-Meherrin	PL - Potomac, Lower	YA - Yadkin
CL - Chowan, Lower	PS - Potomac-Shenandoah	YO - York
JU - James, Upper	RA - Rappahannock	
JM - James, Middle	RU - Roanoke, Upper	

In concordance with the generation of linework and coding for the NWBD, a master file was created to use for attributing the hydrologic units. DCR determined and coded the information needed for all of the optional and required attribution for the NWBD polygons and arcs. In addition to the attributes described by the standards, DCR has also created a file listing most major streams for each 6th order unit and a file noting the extent of the main-stem of the major stream per 6th order unit. This information will further assist users with watershed planning, particularly those without geographic information system (GIS) capabilities.

RESULTS

The final hydrologic unit product arising from compliance with the March 2002 NWBD standards contains 1246 6th order units and 315 5th order units, barring further tweaking. This is a significant change from the 494 14-digit units and 211 11-digit units of the 1995 products. There are also a number of improvements in the NWBD that arise from recapturing hydrologic

units using new geographic information technologies, from past experiences developing and using hydrologic unit systems, and from the opportunities that arise from a true multi-state effort.

(1) There was a more precise delineation and capture of hydrologic unit boundaries. Units of the NWBD were captured from heads-up digitizing on DRGs of the 7.5 minute topographic quadrangle maps and NOAA charts versus from the paper versions of those maps. The ability to zoom and pan made this process more precise at least in regards to capturing linework from these sources.

(2) Linework and labeling were coordinated with all surrounding states so as to make seamless and sequentially coded units across all state borders. This effort was attempted in 1995 but was only successful at the 5th order. The NWBD units will be seamless between all states at all orders. While this is in part due to this being a stated goal in the NWBD standards, it successfully occurred in this version because all states were updating their 5th and 6th order units (to the NWBD) simultaneously.

(3) The first two versions of 6th order hydrologic units developed for Virginia delineated units within the established 4th order units but, with one minor exception, did not affect the delineation of the 4th order units except to recapture them more precisely. Although the NWBD standards requested a continuance of that practice the final product includes a few significant modifications and redefinition of established 2nd and 4th order units. These requested changes (linework and coding), which affect multiple states, have not yet been officially endorsed. They are being requested to fix the more glaring problems created by imposing 5th and 6th order units from the new standards onto larger units developed many standards ago.

The new internal coding scheme, cross-referenced to the previous 3-character coding scheme, is shown in Table 3.

Table 3. Internal coding changes for 6th order units.

1995 UNITS	NWBD UNITS	DRAINAGE
A01-A34	PL01-PL74	POTOMAC RIVER, LOWER
B01-B09	PU01-PU20	POTOMAC RIVER, UPPER
B10-B58	PS01-PS87	POTOMAC RIVER-SHENANDOAH RIVER
C01-C16 and R01	CB01-CB47	CHESAPEAKE BAY/CHESAPEAKE BAY COASTALS
D01-D07	AO01-AO26	ATLANTIC OCEAN COASTAL
E01-E26	RA01-RA74	RAPPAHANNOCK RIVER
F01-F27	YO01-YO69	YORK RIVER
G01-G15	JL01-JL59	JAMES RIVER, LOWER (TIDAL)
H01-H22, H33-H39	JM01-JM86	JAMES RIVER, MIDDLE (PIEDMONT)
H23-H32	JR01-JR22	JAMES RIVER- RIVANNA RIVER
I01-I38	JU01-JU86	JAMES RIVER, UPPER (MOUNTAIN)
J01-J17	JA01-JA45	JAMES RIVER- APPOMATTOX RIVER
K01-K13	CM01-CM32	CHOWAN RIVER-MEHERRIN RIVER
K14-K36	CU01-CU70	CHOWAN RIVER, UPPER
K37-K38	CL01-CL05	CHOWAN RIVER, LOWER
K39-K42	AS01-AS20	ALBEMARLE SOUND
L01-L41	RU01-RU94	ROANOKE RIVER, UPPER
L42-L74	RD01-RD77	ROANOKE RIVER- DAN RIVER
L75-L82	RL01-RL24	ROANOKE RIVER, LOWER
M01-M03	YA01-YA07	YADKIN RIVER-ARARAT RIVER
N01-N37	NE01-NE88	NEW RIVER
O01-O14	TH01-TH46	TENNESSEE-HOLSTON RIVER
P01-P16	TC01-TC35	TENNESSEE-CLINCH RIVER
P17-P24	TP01-TP19	TENNESSEE-POWELL RIVER
Q01-Q14	BS01-BS35	BIG SANDY RIVER

DISCUSSION

Unfortunately, naming standards of the proposed NWBD system will cause confusion because some of these terms are now used to express geographic units with a different extent. The way we now reference river basins, for instance, differs from the meaning attached to the proposed system's use of "basins". Likewise, calling every unit of the new 5th order hydrologic unit system a "watershed" will make it difficult to indicate which are true watersheds and which are not (DCR 2003). If this portion of the standards is unchanged, DCR will likely refer to hydrologic units by their order instead of name and would encourage other state agencies to do the same.

DCR submitted Virginia's portion of the NWBD to the NRCS for certification in September of 2004. Upon certification the official form of the boundary dataset will be made available on the USDA's Geospatial Gateway. DCR provides a second version of the Virginia NWBD - one that

contains the internal coding scheme and additional related files. It is available in both shapefile and coverage format, along with fully qualified FGDC compliant metadata, by contacting the authors.

Whereas maps were produced for past versions of the 6th order units in both a variable scale form on 8.5 by 11 inch pages (USDA and DCR and ISSL 1991, USDA and DCR 1995), and on large stock at 1:126,720 (1 inch equals 2 miles), no such mass produced products are envisioned for displaying the boundaries and codes of the NWBD. It is anticipated that internet map services from DCR of the boundaries over DRGs and the increased use of GIS by users will be sufficient to help the vast majority of those who wish to associate actions and entities to hydrologic unit codes of the NWBD to be successful.

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BENTHIC MACROINVERTEBRATE RESPONSE TO ALTERED FLOW REGIMES

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ABSTRACT

Relationships between benthic macroinvertebrate community health and hydrologic and hydraulic variables were investigated on fifteen streams in three physiographic provinces of Northern Virginia, USA. Differences in hydrological characteristics of streams before and after watershed development were calculated using the Indicators of Hydrologic Alteration methodology and USGS stream flow data. Hydraulic characteristics of stream flow and morphology were modeled with the HEC-RAS hydraulic model using topographic surveys of the stream channels. Principal Component Analysis was used to identify significant relationships between physical and biological variables.

Changes in Flow Predictability, Minimum Flows, and Flashiness were indicated as most significantly related ($P < 0.05$) to benthic macroinvertebrate health. The hydraulic variables Hydraulic Radius, Channel Width Divided by Drainage Area, Froude Number, Shear and Stream Power were also significantly related to macroinvertebrate health. The study suggests that stream protection regulatory efforts should concentrate on measures that reduce the size and number of less-than-bankfull flows.

INTRODUCTION

It has been well documented that converting land from a natural forested state to a developed condition causes a wide array of negative impacts on aquatic habitat, biodiversity, and hydrology (Sovern and Washington 1996, Kemp and Spotila 1997). This study attempted to identify specific aspects of hydrographs of altered streams and determine which aspects of post-development hydrology are responsible for degradation of the stream benthos. Many studies indicate a relationship between increases in imperviousness, residential density and other factors on one hand and decreases in aquatic ecosystem health on the other. Paul and Meyer (Paul and Meyer 2001) listed increased peak discharge and decrease in lag time, increased bankfull discharge, more frequent minor floods, reduced evapotranspiration, decreased infiltration, and reduced baseflow as important factors. Geomorphologic effects include channel enlargement, reduced sinuosity, and eroded streambanks and bed. Estimates of a threshold level for imperviousness in a watershed beyond which the receiving streams ecosystem will suffer range from 5% to 30% (FitzHugh 2001), but the mechanism of this effect, at whatever level it is fixed, is as yet unexplained.

METHODS

Archived streamflow data from USGS gaging stations were used to establish the hydrologic characteristics of fifteen sampling stations in the Potomac River basin. Stream reaches were selected to include a mix of severely degraded, moderately impaired, and healthy streams, as indicated by the macroinvertebrate data.

Hydrologic parameters included the 38 ecologically significant hydrologic factors identified by Richter *et al.* (Richter *et al.* 1999) in their Indicators of Hydrologic Alteration (IHA) methodology, and listed in Table 1. The IHA method organizes the USGS data into pre-development and post-development regimes and compares parametric and non-parametric statistics between the two groups. The pre- and post-development dates were selected for each watershed based upon historical land development information obtained from the local jurisdiction's regulatory agencies, and represent a break between historical land use, either forest cover or agricultural use in most cases, and the advent of significant conversion of the watershed to urban and suburban development. Hydraulic parameters were generated from HEC-RAS numerical models for each stream reach. A portion of each stream near the invertebrate data collection site was topographically surveyed to establish stream cross-section and profile information for input to HEC-RAS. Four of the fifteen USGS sites were eliminated from the HEC-RAS simulation study. Cameron Run, Four Mile Run, Accotink Creek, and South River have been so altered by channelization that any relationships between geomorphologic features and ecosystem health are suspect. For this study, Manning's n was estimated using the examples and protocols of the USGS (Barnes 1977).

The HEC-RAS simulations yielded data on velocity, stream power, Froude number and other hydraulic variables. Simulations were run for two flow conditions: the baseflow condition and the bank-full flow condition. Bank-full flow, generally corresponding to the 1.5-year flood event, was calculated by performing a log-Pearson Type III regression on annual maximum flow values (USGS 1982.). Table 2 lists the variables calculated by HEC-RAS. Stream topographic surveys were conducted according to the procedure developed by the U.S. Forest Service (Harrelson *et al.* 1994).

Macroinvertebrate indices were calculated from bioassessment data obtained from the Virginia Department of Environmental Quality and the Fairfax County Stream Protection Strategy Baseline Study (Fairfax County Storm Water Management Branch 2001.) Indices of macroinvertebrate community health included: Taxa Richness, EPT Richness, Percent EPT, Percent Trichoptera w/o Hydropsychidae, Percent Coleoptera, Family Biotic Index, Percent Dominance, Percent Clingers + Plecoptera, Percent Shredders, and Percent Predators. Both agencies use modified versions of the EPA Rapid Bioassessment Protocol (Barbour *et al.* 1999) for collecting and analyzing benthic macroinvertebrate and fish data. In all cases the macroinvertebrates were classified to the genus level and these data were used to calculate a number of stream health metrics. The measured hydrologic and biotic parameters are similar to those used in previous studies (Clausen and Biggs 1997, Richter *et al.* 1999).

Table 1. Summary of 38 hydrologic parameters used in the Indicators of Hydrologic Alteration and their characteristics.

Hydrologic Variable Class	Hydrologic Parameter
Measures of Annual Variation	Annual coefficient of variation
	Change in flow predictability
	Change in flow constancy divided by change in flow predictability
	Change in percent floods in 60-day period
	Change in flood-Free Season
Magnitude of monthly water conditions	Change in mean flow for each calendar month
Magnitude and duration of annual extreme water conditions	Annual 1-day minima
	Annual minima, 3, 7, 30, and 90-day means
	Annual 1-day maxima
	Annual maxima, 3, 7, 30, and 90-day means
	Number of zero-flow days
7-day minimum flow/mean for year (base flow)	
Timing of annual extreme water conditions	Julian date of each annual 1-day maximum and minimum
Frequency and duration of high and low pulses	Number of low and high pulses within each year
	Mean duration of low and high pulses within each year
Rate and frequency of water condition changes	Means of all positive and negative differences between consecutive daily values
	Number of hydrological reversals

Table 2. Hydraulic variables created by field hydrographic surveying of stream profiles and cross sections, and modeling with HEC-RAS. These parameters were modeled for the bank-full flood (1.5 year flow) and base flow conditions for each reach.

Hydraulic Variable Class	Hydraulic Parameter
Measures of Stream Geomorphology	Channel Width divided by Watershed Area
	Mean Flow Area
Measures of Stream Energy	Hydraulic Radius
	Mean Velocity
	Mean Froude Number
	Shear Stress
	Stream Power

Principal Component Analysis was used to address the degree of variance in the samples explained by each of the factors studied, and to suggest relationships between hydrologic and hydraulic parameters and macroinvertebrate community health indices. The principal component loadings indicate which components best explain the most variation. Non-parametric statistical tools were used to describe and summarize the data, since Komolgorov-Smirnov analysis of all data sets indicated that many variables do not exhibit normal distribution.

Table 3 lists the sites for which macroinvertebrate, hydraulic, and hydrological data were gathered. The prerequisites for each site were the presence of a USGS gaging station with a long period of record, a nearby sampling site with previously collected macroinvertebrate data, and reasonable accessibility to allow the use of topographic surveying equipment.

Table 3. Sampling sites used in the study, with USGS station numbers and the period of record for each station.

Stream	USGS Station No.	Period of Record
Accotink Creek	1654000	1947-2001
Cameron Run	1653000	1955-2000
Cub Run	1656960	1927-1987
Difficult Run	1646000	1935-2001
Four Mile Run	1652500	1951-1999
Goose Creek	1644000	1909-2001
Hogue Creek	1613900	1960-2001
Mattaponi River	1674000	1942-2001
Middle River	1625000	1927-2001
North River Burkettown	1622000	1926-2001
North River Stokesville	1620500	1946-2001
Opequon Creek	1615000	1943-1997
Quantico Creek	1658500	1951-2000
Rapidan River	1667500	1930-2001
South River	1626000	1952-2001

RESULTS AND DISCUSSION

The principal components, or axes, form “super-variables” that differentiate sites or variables along gradients formed by the PCA data reduction. This procedure helped to identify general trends in ecologically relevant relationships between abiotic and biotic factors illuminated by PCA. Biotic and abiotic variables identified as possibly significant in the PCA were tested using the Kendall’s tau statistic, a non-parametric test of covariance.

Principal Components Analysis of the three data sets reduced the dozens of variables to those that explained a significant portion of the variance in the sample. Inspection of the PCA loadings allowed the elimination of a number of variables that had low eigenvalues from further analysis. In all, the PCA analysis allowed elimination of approximately half of the possible correlations, reducing considerably the computational requirements and increasing the chances of identifying relationships with ecological significance. The remaining variables in the hydrological, biological and hydraulic data groups were analyzed by calculation of the Kendall’s tau statistic for each correlation between abiotic and biotic data pairs.

Kendall’s tau analysis indicates that measures of flashiness, increases in flooding, reduction in base flow, decrease in flow predictability year to year all have negative impacts on stream health as expressed by macroinvertebrate sampling indices. Similarly, hydraulic factors including

velocity, hydraulic radius, Froude number, stream power and shear are positively related to macroinvertebrate community health. Channel width divided by drainage area and mean flow area are negatively related to macroinvertebrate community health.

Figure 1 presents a ranking of the relative importance of each variable class. The importance of

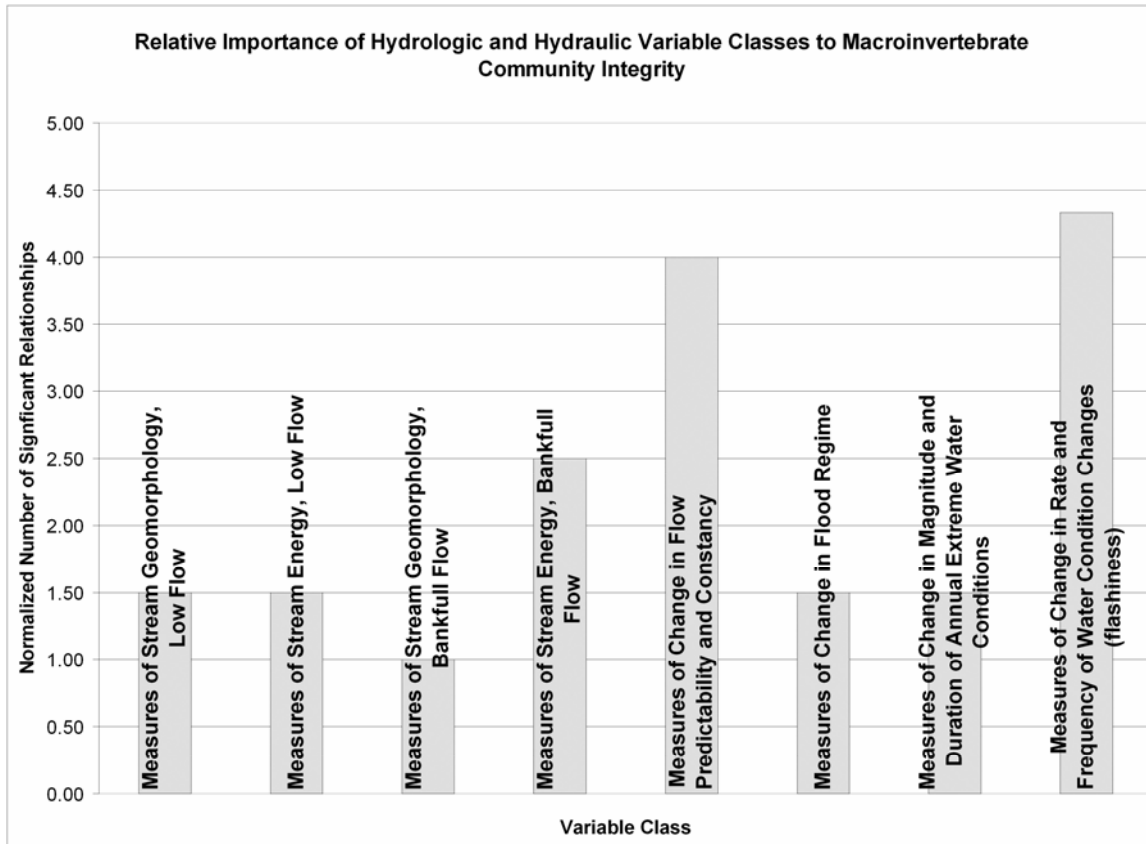


Figure 1. Ranking of the relative importance of each of the variable classes. While measures of flashiness ranked highest, each variable class contained some variables that are significantly related to macroinvertebrate health.

each class was calculated as the number of total significant relationships between all members of the class and all biological variables. This number was normalized by dividing the number of significant relationships for each hydraulic or hydrologic variable class by the total number of variables within the class.

The measures of stream geomorphology at low flow include mean flow area and hydraulic radius. Both are positively correlated with measures of biodiversity, including taxa richness and EPT richness. Many previous studies (Boulton *et al.* 1992, Clausen and Biggs 2000, Dole *et al.* 1997, Gore 1989, Hart and Finelli 1999, Leopold *et al.* 1964, McQuaid and Norfleet 1999, Okay 1998, Schueler and Claytor 2000) have noted that the increased flows and increased number of flows above baseflow caused by increases in watershed imperviousness often cause widening of

the channel. High values of hydraulic depth translate to deep, narrow channels with a gradient of microhabitats within the stream channel for benthic macroinvertebrates. Low hydraulic radius often indicates a stream channel that has been eroded laterally, with shallow flow spread over a broad streambed.

Flow predictability ranks high in the plot of the relative importance of hydrological influences on macroinvertebrate community integrity, the second highest of the variable classes in the study. This result is in accord with those of Poff and Allan (Poff and Allan 1995) and Poff and Ward (Poff and Ward 1989), who found that flow predictability, the year-to-year similarity in the annual hydrograph, was positively correlated with both fish and macroinvertebrate biodiversity metrics.

Changes in flood regime and changes in the magnitude and duration of annual extreme conditions were less important in determining measures of macroinvertebrate health and diversity, but remain an important variable class related to macroinvertebrate health indices. This study uncovered decreases in the richness and diversity of macroinvertebrate taxa with low pollution tolerance values in those streams with altered flood regimes. Poff and Ward (Poff and Ward 1989) cited smaller body size, and accelerated and asynchronous development as adaptations to more frequent flooding.

The variable class with the highest relative importance in determining macroinvertebrate community structure was the measures of change in rate and frequency of water condition changes, or flashiness. The variables that comprise this class describe a variety of aspects of less-than-bankfull flows. The mathematics of flood routing dictates that a given amount of precipitation falling on a relatively impervious watershed will produce not only more frequent minor high flow events, but higher short-term maxima as well.

These results indicate that less-than-bankfull flows have important impacts on stream health. The hydrologic variable indicating the rate of rise in water level (rise rate) and the rate of fall in water level (fall rate) are significantly related to a total of seven biological variables, and the change in high pulse count is related to six biological variables. These hydrologic variables measure the steepness of the hydrograph without regard to the total volume of flow or to the frequency of increased flow. A number of studies have related increases in watershed impervious surface with increases in flashiness and runoff, and other studies relate impervious surface to declines in macroinvertebrate community health (Schueler 2000.) This study ties direct measures of flashiness to reduction in benthic macroinvertebrate indices.

Further research, with finer scale topographic, bathymetric and geomorphologic information generated prior to analysis, might produce interesting results. In particular, the hydraulic parameters are highly dependent on the geomorphological form of the stream channel. A study that uses finer resolution of stream geomorphology might allow more rigorous establishment of relationships between the effects of altered hydrology and biological integrity metrics. A larger number of sampling sites might also indicate whether the effects observed in this study are non-linear. In the analysis of the relationships in this study there are hints of incipient curves in scatter plots of some of the related variables. This study calculated variables only up to the bankfull flood flow, since this flow is geomorphologically most important. Extension of the

calculations of hydraulic factors into higher flows could show effects of large flow events, with the possibility that the data will show relationships more complex than the linear ones assumed in this study.

The results of this study indicate that LID and similar technologies are important steps in the right direction. Flashiness, increased short-term flows, and reduction in groundwater contribution to streams are the stressors identified in this investigation. All of these impacts can be ameliorated significantly by engineering efforts to increase infiltration and evapotranspiration.

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THE IMPACT OF LOW IMPACT DEVELOPMENT

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ABSTRACT

Low Impact Development (LID) has become the *mantra* for urban Stormwater Management (SWM) especially in highly urbanized, and fast urbanizing, areas of Northern Virginia. Many of the Northern Virginia jurisdictions are moving towards requiring developers to adapt LID techniques in order to control the quantity and quality of stormwater leaving their developments. The deteriorating water quality of streams and other water bodies, especially the Chesapeake Bay to which most of the Northern Virginia jurisdictions contribute flows, is the driving force behind the push for LID.

The traditional Stormwater Management strategy has been to remove stormwater from residential and commercial areas as fast as possible after a rainstorm. The idea is to avoid unseemly puddles and marshy areas that can be breeding grounds for mosquitoes, including disease carrying vectors. Hence it was felt that a well managed urban area should have a well designed storm sewer infrastructure leading excess water away from inhabited areas to stormwater detention and retention ponds, which in turn will also delay the excess water reaching streams thus mitigating flooding and erosion damages in water courses. Storm sewer systems and regional ponds became as essential a part of urban infrastructure as drinking water supply and sanitary sewer infrastructure.

Low Impact Development, in contrast, prescribes keeping excess stormwater near homes and inhabited areas detained in rain gardens, infiltration trenches, porous parking lots *etc.* LID also prescribes that instead of curb and gutter, excess water flowing from streets should be conveyed to and through grassy swales.

LID in effect broadly transfers the responsibility for stormwater management from jurisdictional governments to home owners and/ or homeowners' associations (HOAs) and commercial property managers. In these days of budgetary retrenchment, local governments are attracted to potential for resulting capital and maintenance cost savings in reduced SWM infrastructure. It has also become a rallying cry for environmental activists who want to keep the streams clean and the Bay protected.

However this transformation does raise several important questions. Does mosquito breeding increase near places of human inhabitation by adapting LID strategies, especially in the highly impermeable soils of Northern Virginia? Will West Nile virus spread more due to LID? Can homeowners and HOAs maintain LID facilities? Is the shift in cost burden from government to people fair? Standard regulations and manuals put out by VDOT, VA DCR *etc.*, have not been

updated to clarify when and how LID should be used, and when LID may be inappropriate. This has created confusion among Plan Reviewers, with uneven and arbitrary review procedures causing delays in Plan approvals at a time when the real estate market is booming, causing considerable angst among developers and consulting professionals assisting them.

This paper explores the impact of LID from different points of view and urges careful consideration of several concerns by a professional group consisting of representatives of regulators and policy makers at federal, state and local levels, developers, engineers and home owners, to develop consistent policies, regulations, procedures and manuals that clearly integrate LID into mainstream stormwater management strategies and practices.

INTRODUCTION

Storm water management has become an important part of urban municipal responsibility, just as much as safe water supply and wastewater management. Traditionally the strategy has been to collect storm water, via roof gutters *etc.*, from its source, residential, institutional and commercial areas, and carry it through storm sewers, usually laid parallel to water mains and sanitary sewers, to regional retention/ detention ponds, and release the excess from the ponds at a rate closer to pre-development rate of flow to the nearest streams and other water bodies. The main attempt has been to eliminate/ minimize unseemly puddles and aggregations of water that could be breeding grounds for mosquitoes, near habited areas. Also the attempt is to reduce excessive flow rates in stream channels thus reducing erosion. In the process pollutants typical of storm water excess, are trapped and removed from the runoff reaching streams thus minimizing nutrient loadings to streams, lakes and bays. In Virginia much of this strategy is driven by the desire to protect the Chesapeake Bay.

Over the years, storm water management has become a major activity accounting for significant cost and manpower resources of local governments. Partly this cost is recovered through dedicated levies on developers and property owners. However the cost burden has been increasingly felt by the jurisdictions, especially as budgets become tight and revenue resources dwindle.

LOW IMPACT DEVELOPMENT (LID)

Low Impact Development (LID) is based on a strategy exactly opposite to the traditional storm water management. LID advocates disposing of storm water excess, defined as storm water generated due to development in excess of the undeveloped condition, on site. The idea is to match the pre and post development hydrographs of outflow from any development. Suggested techniques include: Bioretention/ Rain gardens, Landscape Island Storage, Porous Pavements and Parking lots, Road side grassy swales, Rain barrels, Catch basins/ seepage pits, Roof top detention, Increasing flow paths, Down spout disconnection, Connecting downspouts to storm chambers placed under the lawns *etc.*

In some anecdotal cases, these techniques have been shown to reduce cost of storm water management infrastructure in terms of reduced need for storm sewers and regional ponds *etc.* In certain locations where soil infiltration characteristics are favorable, it has been shown that LID

techniques reduce overall cost of site development. Lower storm overflow hydrograph peaks and reduced pollutant loads have been demonstrated in certain locations. Prince Georges County, Maryland has been at the forefront of adapting LID techniques. Several other jurisdictions across USA are at various stages of adapting LID strategies.

ANALYSIS OF CURRENT SITUATION

Quite clearly LID has a place in improving storm water management. However, it may not be the universally effective panacea under all circumstances.

The possibility for reduced infrastructure cost in terms of eliminated or decreased storm sewer network and regional ponds has attracted the attention of local governments looking for alternative methods of storm water management at lesser cost. Several local governments nationwide, including some in Virginia, started promoting utilization of LID techniques, also commonly referred to as Best Management Practices (BMPs). Environmental enthusiasts picked up on the anticipated benefits of reduced pollution and reduced infrastructure construction, and started demanding that jurisdictions adapt, and even mandate LID. Several jurisdictions, especially in Northern Virginia encourage use of LID and have been insisting that developers show that they incorporated LID techniques during plan review process. Fairfax County, the largest jurisdiction, is considering incorporating LID requirements in the proposed “Plan Amendments.” Other jurisdictions are at various stages of requiring LID in new development proposals. U.S. Army Corps of Engineers, Norfolk District published, in October 2003, a detailed 25 page Checklist prescribing LID facilities to ‘enhance’ site design. The trend is to require that the post-development outflow hydrograph from a new development match the pre-development conditions.

The speed with which LID is being adopted, and the impending mandates, by regulatory authorities has caused concern in those directly impacted such as developers and site designers. These concerns are being expressed in public and private forums, including some in which the writer participated in several jurisdictional committees in Northern Virginia where vigorous discussions took place on the pros and cons of LID.

A review of available literature on LID, shows description of various techniques and principles of design and installation. Pictures of several installed facilities are also shown on some websites.

However, not much in terms of data on monitoring the performance of installed facilities is available. Also adaptability of LID techniques in poorly draining soils is not well developed or fully understood. Even though some anecdotal cases of cost savings are shown in some of the literature, there is no detailed guidance on how to develop comprehensive cost estimates for LID facilities or how to perform generally acceptable cost-benefit analyses.

CONCERNS THAT NEED TO BE ADDRESSED

It is clear that for LID to become an integral and essential part of stormwater management strategy, these and other relevant concerns have to be addressed. Certainly mandating LID

without addressing genuine and real concerns only leads to avoidable confusion and consternation. Hence it is urged that across the board mandates of LID be held off until further thoughtful and thorough consideration is given to all aspects of LID and this innovative technology is fully integrated into the established science and practice of stormwater management.

In order to assist the process of such consideration, this paper attempts to highlight some of the concerns that need to be investigated further to integrate LID into the mainstream stormwater management practice.

1. LID facilities such as rain gardens, disconnecting downspouts or connecting them to storm chambers under the lawn, construction of swales, rain barrels *etc.*, incorporated into a home plan will place the responsibility for maintenance of these facilities on the homeowner.

Clear guidelines for maintenance of these facilities should be developed and home owners must be trained to ensure proper and timely maintenance.

What are the consequences if the homeowner cannot maintain the facilities properly or fails to maintain at all?

What is the cost of maintenance of these facilities? It must be properly developed and disclosed to the homeowner before the sale of the home.

2. Responsibility for maintenance of some of the LID facilities such as porous pavements, parking lots, grassy swales as alternatives to curb and gutter on sub-division roads, larger rain gardens including those installed on median strips *etc.*, will fall on Homeowners' Associations (HOAs). Typically the developer of a sub-division forms the HOA and transfers it to an elected body of homeowners after the properties in a sub-division are sold out.

In these situations, individual homeowners must be made aware of the maintenance responsibilities that they will inherit individually as well as through the HOA. Individual homeowners as well as HOA officials should be trained in maintenance routines.

The cost of maintenance should be well developed and disclosed to homeowners and HOAs. Sufficient provision should be made, through perhaps an escrow account established by the developer, to cover the cost of maintenance of common facilities after the responsibilities are transferred from the developer to the HOA.

3. While anecdotal references are made to potential cost savings through LID, in literature including websites promoting LID, a thorough and well understood methodology, based on generally accepted principles, should be developed to prepare cost estimates for each type of LID facilities, and cost-benefit analyses to justify incorporation of LID facilities.

4. As discussed above, the traditional stormwater management strategy has been to get rainfall excess away from habited areas to avoid unseemly puddles that could breed

mosquitoes and spread disease, LID is based on the principle of detaining/ retaining storm water excess near habited areas. Does this increase exposure to disease? Specifically will it spread the dreaded West Nile Virus known to have occurred in certain Virginia jurisdictions?

Will the risk of disease increase if LID facilities already installed in a neighborhood are not properly maintained, or their maintenance is completely neglected?

5. Incorporating LID facilities into home plots and subdivisions presumably reduces the need or greatly reduces the cost of traditional stormwater management infrastructure such as storm sewers and regional ponds. This will result in cost savings for jurisdictional governments, which traditionally pay for the construction and maintenance of such facilities. But as noted above, LID facilities increase the maintenance responsibilities and resultant costs on the homeowners and HOAs.

Is this cost shift fair? Should it result in reduction of taxes imposed on homeowners? If so, by how much?

6. While there is great interest, and some movement, in various jurisdictions to encourage, and even mandate LID, the existing guidelines and regulations are not consistent with that objective.

For example, the Fairfax County Public Facilities Manual (PFM) favors regional stormwater detention/retention ponds, as opposed to localized facilities, as the preferred stormwater management strategy. Other jurisdictions have similar stipulations. As discussed above, LID advocates localized detention strategies. While LID advocates doing away with curb and gutter and letting rainfall excess drain through grassy swales on the sides of subdivision roads, VDOT regulations require curb and gutter.

These inconsistencies and contradictions have caused confusion and delays in the review of plans much to the chagrin of developers anxious to complete their projects swiftly, particularly in the fast moving real estate markets of Northern Virginia.

All regulatory agencies at the federal, state and jurisdiction level should get together and revise all relevant regulations to be consistent and unambiguous. Publications such as the State Stormwater Management Manual, VDOT Design Manuals, U.S. Corps of Engineers Wetland regulations, PFM and similar jurisdictional publications should be streamlined to stipulate consistent and clear regulations.

7. Currently there is wide variability in the understanding of LID principles and their application, among persons responsible to review plans in the jurisdictions, state and federal agencies causing inconsistencies and delays in plan approval.

Once the regulatory agencies develop consistent policies, regulations and guidelines, there is an important need to train all concerned including reviewers, designers, builders and people who will end up maintaining LID facilities.

8. LID facilities are generally designed for short duration storms (one year or two year frequency). Traditional stormwater infrastructure including storm sewers and storm water ponds are usually designed for 25-year storms or better.

Under this scenario, can LID completely replace traditional stormwater management strategies? If not, perhaps a hybrid system combining both strategies will evolve. In that case, will LID be justified from cost, operational and maintenance complexity points of view?

9. How will LID facilities designed for one year or two year frequency storms perform in higher intensity storms?

10. If an LID facility is saturated in a first storm, how will it perform in a subsequent storm occurring soon after the first storm? If that results in overflow of stormwater excess, where will that excess go? Will that cause puddles? Overflowing roads and paths? Spread of disease?

11. How will LID facilities which are designed on the presumption that stormwater excess will be soaked or infiltrated through them, perform in soils which do not drain well.

LID literature indicates such situations can be handled with a system of underdrains which will collect the stormwater which will then be disposed through channels that will delay their path to streams and other water bodies.

Clearly such arrangements will increase capital and maintenance costs to LID facilities. Will such increased costs be justified by the presumed benefits? Is it possible for homeowners and HOAs to handle the increased maintenance responsibilities and resultant cost burden?

12. While LID is being introduced through persuasion or mandates in newer developments, for it to have any major impact on overall stormwater management, older developments built using traditional stormwater infrastructure need to be retrofitted with LID. The science of such retrofits and its economics need to be studied and developed if LID is found to be an effective strategy.

13. LID policies and methodologies should be coordinated with other regulations such as TMDLs, wetland regulations *etc.*

14. How do LID facilities fit into the traditional laws and ordinances dealing with storm flow paths and easements between neighboring properties?

15. Some of the regulatory agencies, including several jurisdictional governments, are developing policies and regulations on LID. However, it is important that all agencies at all levels – federal, state and local, coordinate and develop consistent and practical policies, regulations, procedures and manuals that can be easily used by the industry without fear of contradictions and confusion.

16. Many designers and builders are concerned about liability issues that might arise by using LID facilities that are not yet completely tried and tested.

Once the regulatory agencies coordinate and develop consistent policies, regulations and guidelines, they must also clearly define liability issues and limits. Until this happens it is unrealistic to expect widespread use of LID.

CONCLUSION AND RECOMMENDATION

Low Impact Development has proved to be a useful strategy for stormwater management. However, important and valid concerns persist as described above. A professional group drawn from regulatory agencies at various levels – federal, state, and local, as well as academics, designers, builders and home owners’ representatives, should be formed to carefully consider all aspects of LID including concerns described above and develop consistent policies, regulations and procedures to integrate LID into overall stormwater management strategies.

ADVANCED BIORETENTION TECHNOLOGY

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ABSTRACT

Bioretention filtration technology has been used over the last ten years to treat urban runoff. Contaminated runoff passes through a biologically active engineered aerobic filtration media containing plants, soil and microbes where pollutants are removed through a complex array of physical, chemical and biological processes. The latest advancement in bioretention technology has come from the commercialization of the technology by Americast, A Division of Valley Blox, Inc. working with the University of Virginia's Civil Engineering Department. Major improvements have been accomplished by optimizing flow rates, increasing treatment capacity, standardizing the filtration media and simplifying analytical methods for sizing and determining the annual pollutant load reductions to meet Total Maximum Daily Load (TMDL's) pollutant load allocations. Monitoring data shows this advanced system can treat over 90% of the total annual volume of rainfall with maximum pollutant removal rates reaching 95% for total suspended solids, 82% total phosphorus, 76% total nitrogen and 91% heavy metals (measured as Cu). The high pollutant removal efficiency is primarily due the multiple treatment processes inherent in the plant / soil / microbe treatment media.

Additionally, bioretention is a unique among BMP's due to its many ancillary benefits making use of typical landscape plants with low maintenance costs, enhanced aesthetics, improved habitat value, and easy / safe inspection. The "at-the-source" treatment strategy of bioretention is highly adaptable for many urban settings to achieve multiple stormwater management water quality and quantity goals including combined sewer overflow control.

BACKGROUND

Bioretention has been defined as filtering stormwater runoff through a terrestrial aerobic plant / soil / microbe complex to capture, remove, and cycle pollutants through a variety of physical, chemical, and biological processes. The multiple pollutant removal mechanisms of this technology make it the most efficient of all BMP's. The word "Bioretention" was derived from the fact that the biomass of the plant / microbe complex retains, degrades, uptakes, and cycles

many of the pollutants / contaminants of concern including bacteria, nitrogen, phosphorus, heavy metals, and organics such as oil / grease and polycyclic aromatic hydrocarbons (PAH). Therefore, it is the “bio”-mass that ultimately “retains” and transforms the pollutants - hence “Bio-retention.”

Treatment technologies using soils, sand, organic materials, microbes and plants have been used in both water and wastewater treatment. For example, wastewater effluent spray irrigation on fields and meadows has been successfully used for centuries throughout the world (Shuval *et al.* 1986). These systems have been shown to be both economically and environmentally sustainable (Feigin *et al.* 1991).

Bioretention was first developed by Prince George’s County, Maryland’s Department of Environmental Resources (PGCDER) in the early 1990’s (Coffman *et al.* 1993). The PGCDER design manual provides basic Bioretention planning, design and maintenance guidance. The practice was originally developed to allow use of sites’ landscaped and green space to filter and treat runoff. The original design was essentially an enhanced infiltration technique where the filtered water was allowed to infiltrate into the ground.

Since the introduction of bioretention, the success of the practice has been mixed primarily due to the lack of detailed specific design and construction standards. This lack of specificity has led to wide variations in the soil / filter mix, infiltration rates, plant materials, and sizing resulting in costly reconstruction and maintenance repairs. Americast’s advanced design features have eliminated all of the past problems and liabilities of conventional bioretention designs and greatly improved its performance, reliability, and ease of construction and maintenance.

ADVANCED BIORETENTION SYSTEM PHYSICAL DESCRIPTION

The system consists of a concrete container, a 3 inch mulch layer, 1.5 to 3.5 feet of a unique soil filter media, an observation / cleanout pipe, an under-drain system and an appropriate type of plant *i.e.*, flowers, grasses, shrub, or tree.

Stormwater runoff drains directly from impervious surfaces through an inlet structure in the concrete box and flows through the mulch, plant, and soil filter media. Treated water flows out of the system via an under-drain connected to a storm drain pipe or other appropriate outfall. The advanced system can also be used to control runoff volumes / flows by adding storage volume beneath the filter box for either infiltration or detention control (*e.g.*, a gravel infiltration trench area beneath the box).

The concrete container and treatment media are below grade with the only features visible being the top concrete slab, tree grate, plant, and inlet opening. This system looks very similar to an ordinary tree box except that it is specially designed to treat runoff (Figure 1).



Figure 1. Filterra™ advanced bioretention system installation.

POLLUTANT REMOVAL PROCESSES

Pollutants are captured, cycled, degraded and removed by a wide variety of complex physical, chemical, and biological processes as the contaminated runoff flows onto and through the mulch / plant roots / soil / microbe / plant treatment system. Suspended solids are removed through sedimentation as runoff is allowed to pond above the filter media with filtration of pollutants as the runoff passes through the media. Organic compounds are removed by chemical complexing with the organic constituents of the media, microbial degradation, filtration, and sedimentation. Nitrogen is captured through physical and chemical means and removed through nitrification, denitrification, and plant uptake. Phosphorus is removed through adsorption, sedimentation, precipitation and plant uptake. Heavy metals are removed through sedimentation, organic complexing, precipitation, adsorption, and plant uptake.

The pollutant removal mechanisms operate in two distinct time scales. The first time scale occurs during the storm event when pollutants come into contact with the media and are captured instantaneously through sedimentation, filtration, adsorption, absorption, infiltration, and chemical precipitation. The second time scale is between storm events. Pollutant removal and cycling occurs in a matter of hours, days, and weeks through biological degradation, biological uptake, and volatilization. The filter media is designed to capture pollutants during the storm event while biological processes degrade, metabolize, detoxify, and volatilize the pollutants during and between storms.

The difficulty with removing pollutants in urban runoff is that they occur in a wide array of organic and inorganic forms and in various particle sizes from gross solids to dissolved molecules. Each of the various pollutant forms and particle sizes can require different processes and mechanisms for capture and treatment. Bioretention's complex media structure provides for an array of physical, chemical, and biological treatment processes to handle a wide variety of pollutants.

TREATMENT CAPACITY

The treatment capacity is dependant on the overall pollutant removal capabilities of the treatment media and the hydraulic properties of the media. Many of the pollutant removal processes were mentioned above. The hydraulic properties of importance are the flow rate through the media and the volume of runoff it can treat. Both the pollutant removal and hydraulic capacity of the system have been measured though monitoring conducted by the University of Virginia (UVA Mixed Media Study 2001). Based on these measured values, a performance curve can be developed for various pollutants (Figure 3). This curve shows that pollutant removal capabilities vary with the ratio of media's surface area to contributing drainage area. Increasing this ratio will increase the pollutant removal rate up to the maximum removal capacity of the media to capture and process the pollutants.

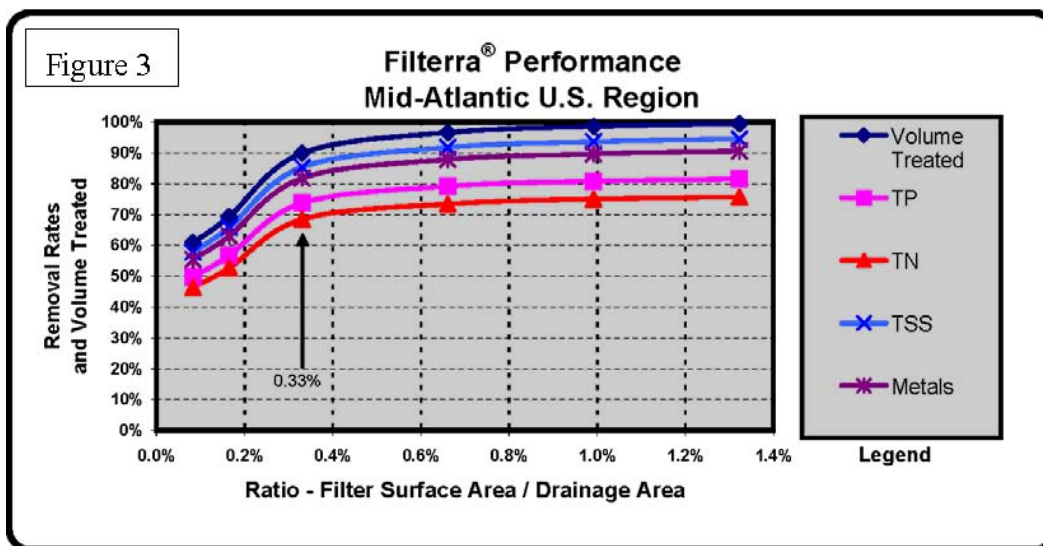


Figure 3. Pollutant removal of advanced bioretention system.

Based on test data and rainfall distribution of the Mid-Atlantic region of the U.S., the optimum media surface to drainage area ratio is about 0.33% or 36 square feet of media / 0.25 acres of contributing drainage area. Using the 0.33% ratio, the system will treat approximately 90% of the annual volume of runoff and can achieve maximum expected pollutant removals of 95% for total suspended solids, 82% total phosphorus, 76% total nitrogen, and 91% heavy metals (measured as Cu). The 0.33% ratio will vary from region to region as rainfall intensities varies. An explanation of the hydrology and hydraulic method for sizing the system is provided below.

HYDROLOGY AND HYDRAULIC ANALYTICAL METHOD

Americast has developed a unique and sound analytical method to determine the appropriate media surface area needed to achieve the desired treatment levels. This approach can be used for all filter media's. The sizing is based on appropriately matching the media's flow rate to the unique rainfall / runoff characteristics or probable rainfall intensity distribution of a region. Once rainfall intensity distribution and the media flow rates are know the total annual volume of runoff treated by the media can be determined.

For the Mid-Atlantic region, 50 years of rainfall data was analyzed from Reagan National Airport from which the probability and frequencies of all rainfall intensities (inches/hour) were determined. Knowing the frequency / distribution and the media flow characteristics, one can determine the annual volume of runoff that can be treated and the optimum surface area for any given drainage area. The performance chart for the Mid-Atlantic U.S. region (Figure 4) summarizes the rainfall intensity distributions, predicted pollutant removal rates, and volumes treated for 36 sq. ft. of media surface area with a ¼ acre drainage area. The MS Excel based performance spreadsheet will automatically calculate the filter media surface area needed to treatment goals of any given drainage area. If other pollutant removals are required or certain annual pollutant load reductions are needed, the spreadsheet can also calculate the surface area needed.

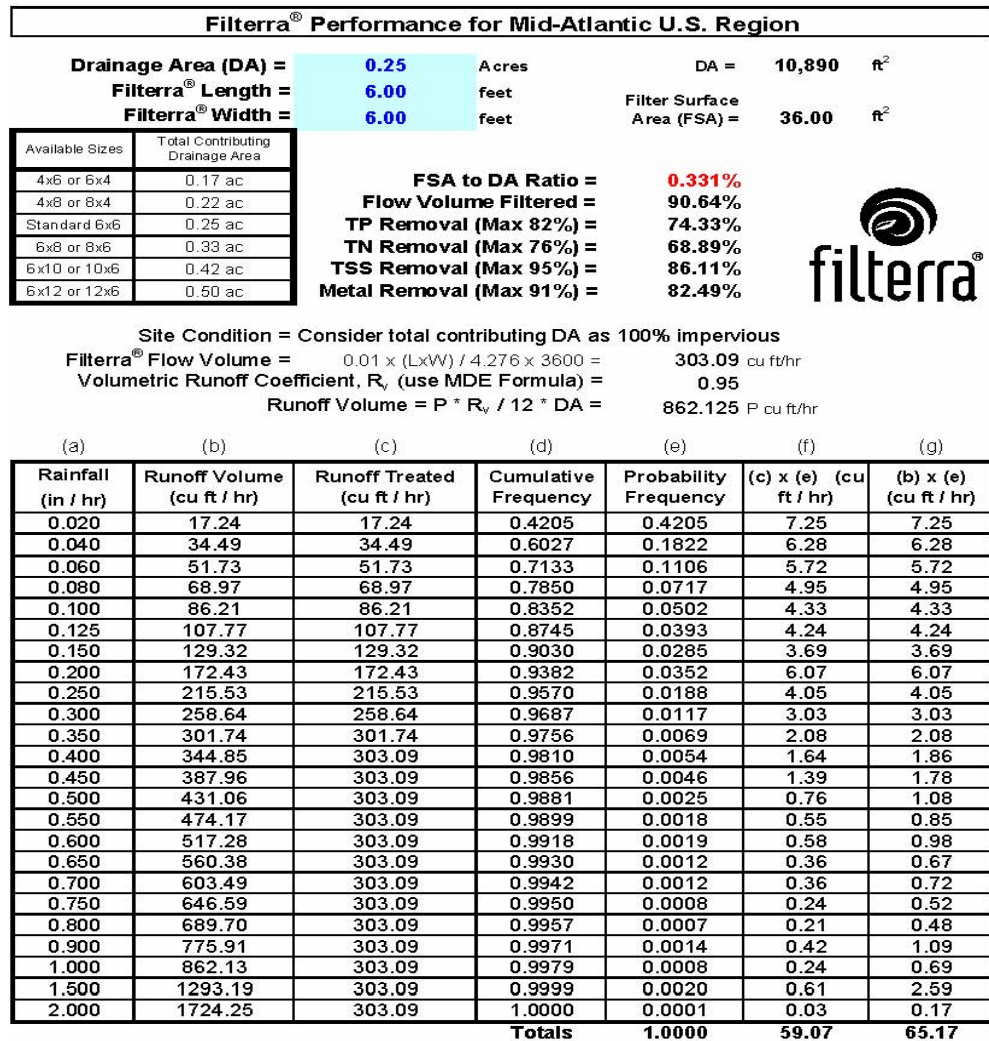


Figure 4. Sizing chart.

Calculating the annual pollutant load removal is determined by simply multiplying the percent annual volume treated by the maximum pollutant removal percentage for each pollutant. These values can be found in the performance chart above.

Example: Annual volume treated = 90.64 %
 Maximum TSS Removal = 95%
 Annual TSS Removal = $\frac{(90.64\%)(95\%)}{100}$ = 86.11%

UNIQUE DECENTRALIZED PLACEMENT

Another unique feature of the advanced design, sizing, and placement is that it utilizes a distributed design approach fundamental to the innovative Low Impact Development technology (LID). This design philosophy promotes at-the-source controls; off-line configuration of the units, treating relatively small drainage areas (less than ½ acre), and a more uniform distribution of controls throughout the site. This is opposed to conventional end-of pipe and in-line treatment approach used for most BMP designs. The LID approach reduces the effective hydraulic and pollutant load to each unit thereby increasing performance and reducing maintenance burdens. Controlling runoff as close to the source as possible also eliminates problems common to conventional BMP's such as concentrated high flows that cause erosion and re-suspension of pollutants or expensive control structures to store, split, or divert high flows. Using small drainage areas ensures that runoff flows and velocities are always very low.

EASE OF DESIGN

Americast's recommend media surface area to drainage area ratio of 0.33% for the Mid-Atlantic region is adequate to meet current state and Federal NPDES pollutant removal requirements. Americast offers a variety of precast concrete box sizes to meet most of site design needs. As long as you follow the LID design principles by distributing the units and keeping the drainage area to surface area at about 0.33%, the designer can just place the proper size filter box to match the drainage area, see Figure 5.

Available Sizes	Total Contributing Drainage Area
4x6 or 6x4	0.17 ac
4x8 or 8x4	0.22 ac
Standard 6x6	0.25 ac
6x8 or 8x6	0.33 ac
6x10 or 10x6	0.42 ac
6x12 or 12x6	0.50 ac

Figure 5. Filter surface area to drainage area.

OFF-LINE / BYPASS DESIGN

Another unique design feature of Americast's advanced bioretention system is its off-line design configuration. This design strategy improves treatment and avoids the possibility of resuspension of particulate matter. It is important that the site designer confer with Americast on the proper location of the unit to ensure that the site grading and placement is correct (Figure 6).

The site designer must also plan for the by-pass of high flows. Although the system will treat about 90% of the total rainfall events / volume, occasionally the flow capacity of the treatment media will be exceeded causing the unit to go into bypass mode. The bypass flows must be safely conveyed to a nearby inlet or other appropriate discharge point. Sump conditions must be avoided. If the unit is placed in a sump, bypass mode will result in flooding around the unit and cause re-suspension of the debris collected in the unit.

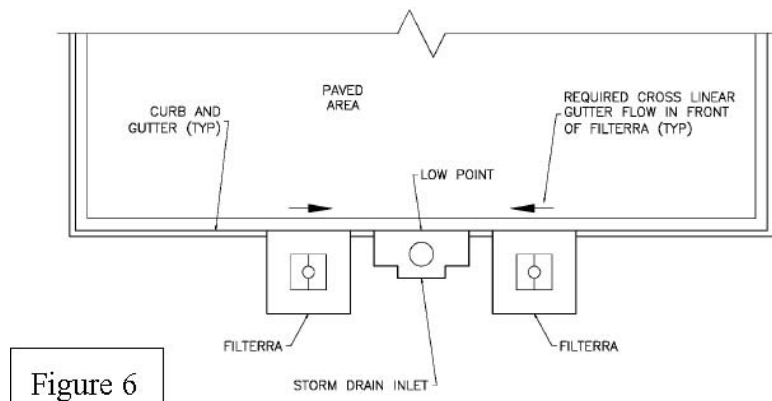


Figure 6

Figure 6. Offline design configuration.

CONCLUSIONS

Americast's commercialization of an advanced bioretention system has resulted in a number of improvement to the technology that will provides a much more cost effective system that occupies much less space, is easy to design and construct and provides more reliable treatment.

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QUANTIFYING INFILTRATION, RUNOFF, AND POTENTIAL FOR NON-POINT SOURCE POLLUTION IN MODEL URBAN LANDSCAPES

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KEY WORDS: sediment, surface water quality, turfgrass, urban forest, wood mulch

ABSTRACT

The contribution of non-point source pollution to degrading surface water quality is considerable throughout Virginia and beyond. While research on agricultural best management practices in nutrient management and nutrient and soil stabilization has made progress in reducing agricultural contributions to nutrient and sediment loading of watersheds, little is known about how land covers of different vegetation representative of urban areas (*e.g.*, bare soil versus turfgrass lawns versus urban forest) influence the potential for non-point source pollution. Using an innovative experimental design, manipulated rainfall was applied to plots with landscape covers of bare soil, shredded wood mulch, turfgrass, and simulated urban forest. Runoff was collected and analyzed for volume and sediment mass. The data obtained were used to quantify the influence of land cover type and precipitation level on volume and quality parameters of infiltration, leaching, and runoff. Preliminary results indicate the turfgrass treatment reduces water and sediment runoff more than the other landscape treatments considered here. Additional data on nitrogen and phosphorous levels will provide more complete information to better inform land use policy and best management practices of the influences of urban landscapes on water quality.

INTRODUCTION

The impact of urban landscapes on regional water quality continues to gain national attention. The Chesapeake Bay Foundation's *State of the Bay 2003* reports that overall water quality in the bay declined in the past year, largely as a result of increased nitrogen, phosphorus, and sediment loading to surface waters from urban landscapes after record heavy rainfalls (Chesapeake Bay Foundation 2003). These trends are mirrored nationally in nearly every watershed impacted by rapid urbanization. Urban community planners, real estate development professionals, commercial land managers, urban foresters, and state and federal agency personnel who oversee land use and its impact on environmental quality are challenged by the lack of data regarding the influence of land cover types on the potential for precipitation to infiltrate into soils versus contributing to runoff of sediment and nutrients to local watersheds. Here, we contribute to refining best management practices for planning, development, and management of urban landscapes by quantifying the influence of land cover type in conjunction with manipulated precipitation on volume and quality parameters of precipitation infiltration, leaching, and run-off.

Nutrient and sediment runoff from urban landscapes contribute to watershed impairment via adverse consequences to the biological and the physical environment. For example, N and P loading contribute greatly to increased algal growth. Total N and P concentrations in surface waters as low as 1 mg L^{-1} and $25 \text{ } \mu\text{g L}^{-1}$, respectively, can produce algal blooms to the serious detriment of aquatic ecosystems (Baird *et al.* 2000). Algal blooms, in turn, shade beneficial aquatic vegetation, reducing these stands of food and shelter for many other aquatic organisms. Increased algal growth also contributes to increased biological oxygen demand and reduced oxygen supply, which may be compounded by increased water temperatures from loss of shading canopy vegetation. Oxygen deficiency reduces the capacity of aquatic ecosystems to support biological activity. Excessive N loading to watersheds can also be a human health issue. $\text{NO}_3\text{-N}$ can be deleterious in potable water sources, leading the U.S. Environmental Protection Agency to set 10 ppm as an upper limit for $\text{NO}_3\text{-N}$ in drinking water (Erickson *et al.* 1999).

Some land uses and vegetation cover types have been shown to influence nutrient runoff. For example, land under cultivation showed greater losses of N and P and greater erosion than forested areas (Mathan and Kannan 1993). Rai and Sharma (1998) found increased losses of N and P and increased sediment erosion as forested land was converted to agricultural land over a three year period, suggesting that forest cover is needed for ecological sustainability of the watersheds studied. Turfgrass plantings, particularly those associated with golf courses, have also been studied. Intensively managed turfgrass, as found on golf courses, often contributes to N and P runoff (Linde and Watschke 1997, King *et al.* 2001). Fertilized turfgrass can contribute significantly more N runoff than unfertilized plots (Gross *et al.* 1990). Although the contribution of both N and P to nutrient runoff in managed landscapes is often highly dependent on fertilization management, transport of P is usually closely tied to sediment erosion. Consequently, nutrient loading of surface waters from urban and suburban landscapes is highly variable due to many factors including soil type, vegetation cover, and management practices (Petrovic 1990).

In efforts to reduce nutrient runoff from agricultural lands, research on best management practices in nutrient management and nutrient and soil stabilization have made progress in reducing agricultural contributions to nutrient and sediment loading of watersheds. For example, the implementation of riparian buffer strips with various types of vegetation to absorb or filter nutrients and sediments in runoff can reduce the deleterious impact to adjacent streams (Srivastava *et al.* 1996, Schmitt *et al.* 1999, Arora *et al.* 2003). In managed urban and suburban landscapes, turfgrasses have been employed in similar roles as vegetative buffers, but this practice is not used consistently. For example, homeowners and professional landscape managers are just as likely to fertilize the buffer strips as other stands of turfgrass, defeating the purpose of the buffer strip. Also, while it is generally believed that turfgrass provides effective buffer vegetation, little research has addressed alternatives to turfgrass in urban landscapes as a means of runoff reduction (Erickson *et al.* 1999).

Research addressing the influence of land cover types in urban landscapes to nutrient and sediment runoff necessitates controlled experimental designs. An innovative design that utilizes alternative landscape land covers in combination with manipulated precipitation volume was used to quantify impacts on runoff versus infiltration of precipitation, potential for sediment erosion, and transport of nutrients from landscapes. Specifically, we compared land covers of

bare soil, mulch, turfgrass, and urban forest. The data produced from this research will be able to be incorporated into databases utilized in modeling of land cover influences on runoff pollution potential. In addition, the results of this research will inform landscape trade professionals, providing additional knowledge to reduce the environmental impact of urban and suburban landscapes.

METHODS

Research plots were located on a N/NE facing slope at Kentland Farms (37.19°N, 80.58°W), near Blacksburg, VA. Plots were constructed over a period of four weeks in May 2004. A weather station on-site was used to monitor rainfall amounts. Six rainfall events during July and August of 2004 are reported on in this paper.

An arrangement of replicated triplicate plots (Figure 1, n for each treatment = 3) was used to impose precipitation treatments. Each triplicate plot was made up of three 8 ft. x 8 ft. (2.44 m x 2.44 m) plots that all received the same landscape treatment. Following the down slope of the hillside, 6 in. (15.24 cm) d. PVC pipe cut length-wise in half to form troughs was used to cover and collect 50% of precipitation falling on the middle plot and move it and deposit it over the lower plot. The troughs were held 15.24 cm above the ground using bent steel wire to prevent interference with any growing vegetation. The use of these troughs creates three manipulations of precipitation across the triplicate plot, with 100% of actual precipitation falling in the upper plot, 50% falling in the middle plot, and 150% falling in the lower plot. All triplicate plots were arranged in one row, with spacing between each plot of 2.44 m.

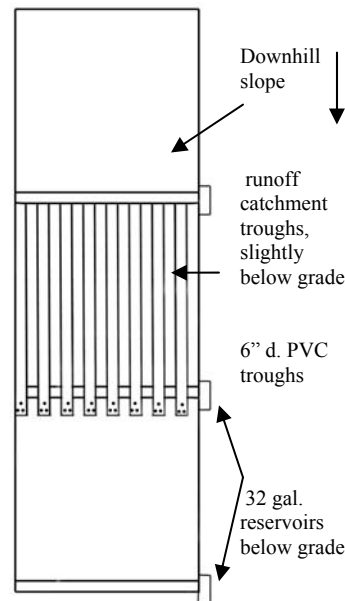


Figure 1. A triplicate plot to provide urban landscape experimental areas with 100% (top), 50% (middle), and 150% (bottom) of natural precipitation.

Landscape treatments used were bare soil, turfgrass, shredded wood mulch, and simulated urban forest. Turfgrass treatments made use of existing turfgrass, while all other plots received an

initial application of Roundup® to kill existing turfgrass. Dead vegetation was removed mechanically from the surfaces of non-turf grass plots, but soil was left undisturbed. Nothing further was done to bare soil plots; mulched plots received a 3 in. thick layer of shredded wood mulch. Each 8 x 8 simulated urban forest plot was planted with four 3.175 cm diameter pin oak (*Quercus palustris*) trees and covered with a 3 in. thick layer of composted leaf mulch to simulate forest litter layer.

Runoff from each 8 x 8 plot was collected along the bottom end of the plot by a 6 in. (15.24 cm) PVC trough, like those described above, sunken just below the surface, with each end capped. A 7.62 cm wide strip of metal flashing along the bottom of each plot was used to ensure runoff entered the trough. Each trough was slopped slightly to drain runoff to one end cap, which was drilled to allow runoff to drain out into collection reservoirs. Collection reservoirs for each 8 x 8 plot were constructed by sinking a 32-gallon (121.13 L) garbage can into the ground. Five gallon (18.93 L) buckets placed inside each can collected runoff and allowed easy removal for analysis following a rain event.

Runoff collected from each plot for each rain event was filtered at 63 µm to collect sediment using standard brass sieves. Sediment collected was rinsed onto pre-weighed filter paper, allowed to drain, and dried at 70°C for 2 days. Filter papers were weighted again and sediment mass determined. Water runoff volume was measured using graduated beakers. JMP statistical software was used to run statistical analysis of data.

RESULTS

Data was collected from six rain events with varying precipitation amounts during July and August of 2004 (Table 1). The rainfall volumes that fell over an 8 x 8 plot were calculated based on the actual rainfall amount, the amount that fell on the upper 8 x 8 100% plots.

Table 1. Rainfall event amounts and volumes per 8 x 8 plot.

Rainfall Date	Rainfall Amount (mm)	Rainfall Volume on 8 x 8 plot at 100% (L)
3 July 2004	21.8	129.88
12 July 2004	12.4	74.00
19 July 2004	6.9	40.78
23 July 2004	13.0	77.02
2 August 2004	23.9	141.96
6 August 2004	15.8	93.63

For each rain event, replicates of each landscape treatment at each precipitation manipulation were averaged. The results were plotted to compare the averages of water and sediment runoff. The plots indicated similar general results for both water and sediment data. A representative plot of water runoff indicates that the turfgrass treatment produced less runoff than the other landscape treatments, while the mulch and urban forest treatments produced the most runoff (Figure 2). Sediment runoff was much more variable, but some trends were fairly consistent between events. Most noticeable was that turf generally produced less sediment runoff than other landscape treatments (Figure 3).

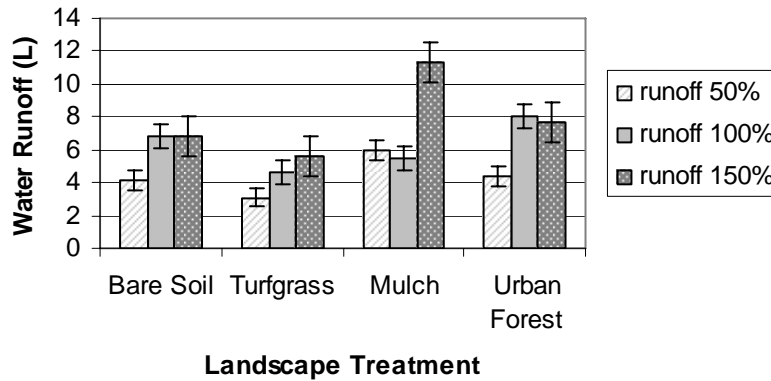


Figure 2. Water runoff by landscape treatment and precipitation level. This is a representative plot using data from the 6 Aug rain event.

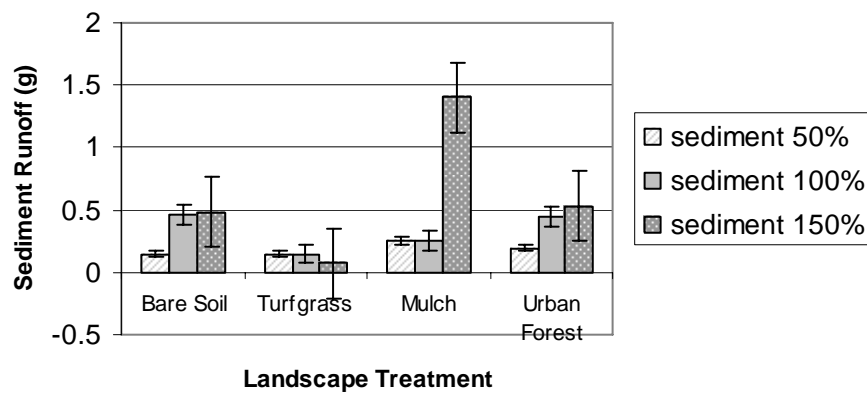


Figure 3. Sediment runoff by landscape treatment and precipitation level. This is a representative plot using data from the 2 Aug rain event.

Because rainfall amounts for each event were different, data from each event was analyzed with statistics separately. ANOVA tests were performed on both water volume and sediment mass runoff that was collected for each event (Table 2). Results of ANOVA tests on water runoff volume indicate significant differences among landscape treatments except for the 23 July rain event. ANOVA tests on sediment runoff masses were less consistent between events. Only the second three rain events indicated there were significant differences in the sediment runoff among landscape treatments.

Table 2. P-values from ANOVA tests for each rain event.

	3 Jul	12 Jul	19 Jul	23 Jul	2 Aug	6 Aug
Water Runoff	*0.0079	*0.0200	*0.0409	0.0578	*0.0001	*0.0067
Sediment Runoff	0.8132	0.1722	0.4010	*0.0011	*0.0132	*0.0017

*Indicates significant difference among landscape treatments, $\alpha=0.05$.

To compare the ability of the different landscape treatments to reduce water runoff over all six rain events (representing a range of precipitation volumes), the water runoff volume was divided by the amount of precipitation received for each 8 x 8 plot for each rain event, where a smaller number indicates less runoff. Averages of these ratios by precipitation treatment indicate the

turfgrass treatment produces the least runoff (Table 3). Averaged across all three precipitation levels, the mulch treatment was shown to produce the greatest amount of water runoff.

Table 3. Averages of ratios of runoff volume to precipitation volume for each landscape.

% Precipitation	Landscape Treatment			
	Bare Soil	Turfgrass	Mulch	Urban Forest
50%	0.102	0.081	0.136	0.104
100%	0.092	0.050	0.070	0.095
150%	0.055	0.048	0.078	0.059
Average	0.083	0.060	0.095	0.086

DISCUSSION

The results of this research, to date, point to the turfgrass treatment as being more capable of absorbing water and sediment runoff compared to the other three treatments. First, the plots of water runoff from each rain event consistently show the turfgrass treatment producing less runoff than other treatments as seen in the representative plot (Figure 2). Although less clear, sediment runoff plots also indicate turfgrass as producing the least sediment runoff of all the treatments (Figure 3). However, the sediment runoff level is most likely based on the water runoff volume, providing an explanation for why the turfgrass treatment produced the least sediment runoff. The ratios of the runoff volume to the precipitation volume provided a second set of evidence indicating the turfgrass produced less runoff than other landscape treatments with a ratio approximately 0.02 to 0.03 units smaller than the other treatments for the average of all precipitation treatments. The ANOVA results confirm the difference seen in the amount of water runoff from the turfgrass treatment, indicating there were significant differences among the four landscape treatments in five out of the six rain events collected (Table 2).

Beyond the differences in the runoff from turfgrass, there were other interesting trends in the data. At first glance it was surprising that, almost always, the bare soil did not produce the greatest volume of runoff (Figure 2). Rather, the urban forest and mulch treatments produced the greatest runoff. This suggests there are other factors influencing the runoff. Considering the spacing of rain events and weather conditions, there was enough time in between events for the bare soil plots to dry out, while the mulch and tree canopies maintained higher soil moisture contents. Because of this difference in the soil moisture contents, the bare soil plots were most likely able to absorb greater amounts of water than the mulch and urban forest treatments.

When looking at both the plots of the water runoff from each rain event and the runoff-to-rainfall ratio, it can be seen that there are different trends in the amounts and ratios of water runoff at different precipitation levels. Looking at turfgrass first, this landscape treatment appears to have a fairly linear increase in runoff as precipitation increases (Figure 2). In contrast, the mulch treatment shows what appears to be an exponential relationship, such that large increases in runoff do not occur until the amount of precipitation has increased greatly. Opposite of this trend is the more logarithmic trend that appeared in the urban forest and bare soil treatments, with large increases in runoff at first while leveling off as precipitation levels increase.

Looking at the ratios in Table 3, there is a different set of trends visible. Among all four landscape treatments, as precipitation level is increased, the amount of runoff per unit of precipitation decreases. More specifically, as precipitation level increased, the bare soil and urban forest treatments showed an exponential decrease in the ratio, while the turfgrass and mulch treatments indicated a logarithmic decrease in the ratio. Despite these interesting trends, seen in both Figure 2 and Table 3, no conclusions can be made about the trends at this point. There are many factors that could have caused these trends to occur that were beyond the scope of this project to date.

At this point in this research project, we have started to see interesting results on water and sediment runoff. These preliminary results indicate turfgrass has a greater ability to absorb water, and thus, reduce runoff of water and sediment. This suggests that turfgrass may be the most efficient of these landscape treatments at reducing runoff. However, we are still in the process of collecting and analyzing data and have much to consider beyond runoff water volumes and sediment masses. Of key interest to us is nitrogen and phosphorous runoff from the different landscape treatments, as it is critical that the runoff of these nutrients is controlled due to the environmental and public health risks that increased levels of these nutrients pose. Additional work on this project and future research projects will need to address these issues.

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STREAM DISCHARGE MEASUREMENT USING A LARGE-SCALE PARTICLE IMAGE VELOCIMETRY PROTOTYPE

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KEY WORDS: LSPIV, discharge, measurement, stream

ABSTRACT

New technology for the development of stage-discharge relationships has been pursued due to concerns about safety, accuracy, and costs of traditional discharge estimation methods. Large-Scale Particle Image Velocimetry (LSPIV) is a recent application of flow visualization to measure stream discharge. The LSPIV system tracks the movement of ‘tracers’ through successive video images using statistical correspondence. A laboratory prototype was developed and tested with good results. The Froude number was found to significantly affect the accuracy of the LSPIV prototype. Additionally, oblique camera angles above 30 degrees negatively affected the accuracy. The LSPIV discharge measurements were found to be equivalent to a Marsh-McBirney flow meter in the laboratory application. Additional field work will be completed in the fall of 2004.

INTRODUCTION

Open channel flow is a component of the hydrologic cycle where measurements representing large geographic extents are economically feasible and can be achieved with reasonable accuracy (Hersch 2002). Good water management is therefore founded on accurate open channel flow measurements. Flow information is necessary in many diverse applications including water supply management, pollution control, irrigation, flood control, energy generation, and industrial use (Hersch 2002). Flow information is dependent on obtaining field measurements in sometimes challenging conditions.

Stream gauging stations have been used as the standard method of measuring open channel flows for over 100 years (Costa *et al.* 2000). A gauging station is generally an automated system that measures the depth of flow in the channel (stage) and estimates flow using a stage-discharge relationship. If a calibrated control structure is not installed in the stream, the channel bathymetry is surveyed and flow is measured using current meters, pressure sensors or, in difficult measuring conditions, Acoustic Doppler velocimetry (York and Oberg 2002). Flow measurements, across a large range of conditions, are used to develop a channel specific stage-discharge relationship. Flow monitoring is generally labor intensive and a costly endeavor (Grant 1997).

Due to funding constrains, state and federal agencies can employ only a limited number of field technicians for stream flow measurements (Melcher *et al.* 2000). As a result, the technicians can

make only a limited number of individual flow measurements. In order to develop rating curves with a high level of accuracy, much of the technicians' time is devoted to a limited number of streams. Consequently, the capacity to expand the number of gauging stations is severely limited.

New technology for the development of stage-discharge relationships has been pursued due to concerns about safety, accuracy, and costs of traditional discharge estimation methods during high flow conditions (Grant 1997). The U.S. Geological Survey (USGS) is investigating technologies for direct, continuous, non-contact measurement of open channel discharge (Melcher *et al.* 2000). Research by Costa *et al.* (2000) showed that it is possible to measure discharge with non-contact methods that maintain accuracy levels equivalent to those of conventional methods. Cruetin *et al.* (2003) demonstrated that large-scale particle image velocimetry (LSPIV) might be a viable method for collecting discharge measurements accurate enough for rating curve development. The USGS and other researchers have concluded that LSPIV is a promising technology for non-contact remote flow measurement (Bradley *et al.* 2002, Melcher *et al.* 2000).

Particle image velocimetry (PIV) is a system capable of measuring velocity fields by collecting and analyzing recorded images of the flow field. The PIV system tracks the movement of 'tracers' through successive images using statistical correspondence. Cross-correlation algorithms divide the image into small interrogation areas; each producing one displacement vector. The velocity is the ratio of the particle displacement divided by the elapsed time between images. PIV technology is an offshoot of classical flow visualization and laser speckle velocimetry (Adrian 1991).

Use of LSPIV for flow measurements in low-order streams has several advantages. LSPIV is not as labor intensive and does not present the safety concerns of the conventional methods during high flow events. Another, promise for LSPIV is remote monitoring applications (Bradley *et al.* 2002, Cruetin *et al.* 2003), which could also reduce labor and data management costs (Melcher *et al.* 2000).

A limited amount of LSPIV field research has been conducted. For example, research by Bradley and Kruger (2002) found that discharge measurements determined with LSPIV are as accurate as those measurements using conventional methods. Bradley and Kruger (2002) compared discharge measurements from LSPIV and current meter methods for a stream in Iowa. The study found differences between current meter and LSPIV discharge measurements were within the estimated standard error of the current meter. In a more recent study, Creutin *et al.* (2003) measured discharge in a 70-m cross-section of the Iowa River. Their results indicate that LSPIV is capable of constructing a rating curve accurately and quickly with a stationary camera.

Previous applications of LSPIV to flow monitoring systems have been limited. Additionally, no work using LSPIV has been conducted in the streams of Virginia. Semi-automated LSPIV systems show promise as an inexpensive non-contact method for flow measurement in low-order streams within small watersheds in Virginia. More efficient rating curve development could increase the number of streams that are monitored, and thereby help the Commonwealth better

manage its water resources. The goal of this study was to evaluate the accuracy and feasibility of using LSPIV technology to measure flow in the low-order streams in Virginia.

METHODS

Development of LSPIV for measuring stream discharge has the ultimate goal of being either commercially viable or adopted for use in large monitoring projects. In either case, a LSPIV system must be accurate, inexpensive, and easy-to-operate. The scheme used in this study for the development of LSPIV follows a logical progression: assimilate current knowledge, develop methods and acquire equipment, conduct laboratory and field experiments for ‘proof-of-concept’, and refine the methods to decrease costs and increase usability. This paper reports on various stages of the progression up to and including field proof-of-concept experiments.

Laboratory Experiments

The objective of the laboratory experiments was to become familiar with the equipment and methods necessary for LSPIV discharge measurement and evaluate the accuracy of LSPIV under controlled conditions. Subsequently, a prototype was tested under various conditions in the laboratory. The experiments were conducted in a one foot-wide re-circulating flume (B-16 Hydraulic Demonstration Channel Series 6201). Three treatments were compared: control (flume manometer), Marsh-McBirney (Model 2000) flow-meter, and laboratory LSPIV prototype. Different conditions were investigated in the form of the independent variables: Froude number, camera angle, and number of tracer particles. A total of 120 repetitions were necessary, collected in two experiments, both following a split-plot statistical design. This split-plot design minimized the error term, giving 80 degrees of freedom (personal correspondence with Dr. Golde Holstein).

The experiments were designed to first test the effect of the number of tracer particles (seeding density) on the accuracy of the LSPIV prototype. Five levels of seeding densities were used corresponding to an average of 1, 2, 3, 4, or 5 particles per interrogation window. Additionally, two levels of camera angle (0 and 15 degrees) and Froude numbers (0.05 and 0.30) were used to test for interactions with the number of particles. The next experiment examined the effects of Froude number and camera angle on the treatments and evaluated the treatment differences. To accomplish this objective, the seeding density was fixed to test the effect of four levels of Froude number (0.05, 0.15, 0.25, 0.35) and camera angle (0, 15, 30, 45 degrees) using 80 repetitions. The two experiments were conducted following the same procedures and using the same equipment.

The methodology developed in this study is comprised of several steps, as shown in Figure 11. Initially, site selection determines an optimum location for the LSPIV device (in field applications). The images are recorded and saved during the image acquisition step. The images can be altered to improve results using image enhancement. Next, the images must be registered to the physical location and transformed to remove spatial distortion. In the image-evaluation step, the surface velocities are determined. If some velocities are incorrect, a validation process is conducted to replace the erroneous data. The correct velocity values must be averaged with space and depth and then a discharge can be estimated. The data can then be transferred to a

remote location for further use. Each step follows a distinct set of routines, often requiring specialized equipment.

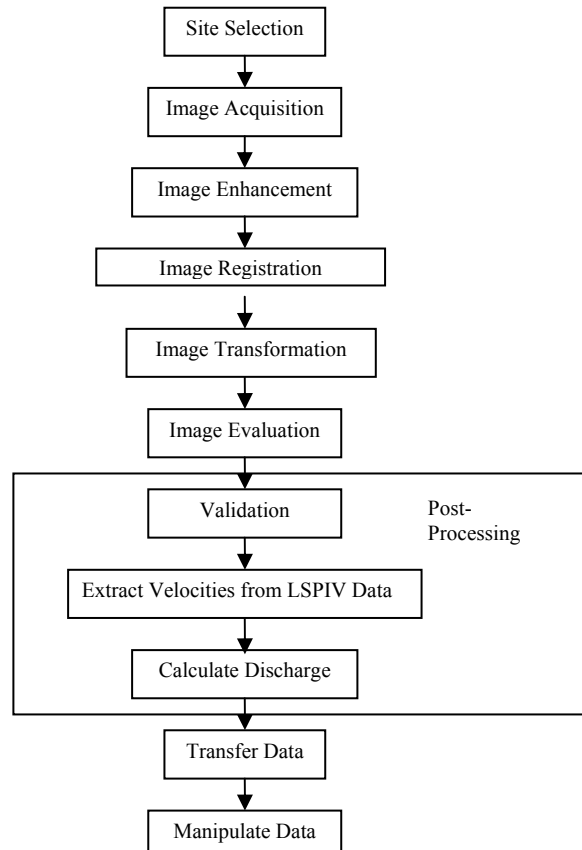


Figure 1. Flowchart showing the generalized steps involved in LSPIV. These steps are developed specifically for LSPIV application to low-order stream discharge measurement.

As an initial step, a laboratory-scale LSPIV prototype was constructed after a literature review was performed to identify best methods and materials. The selected camera is a Pelco monochrome CCD camera (Model# MC3651H-2) with extra-wide lens (1.4 mm). This is an analog output which then is digitized via an ImagingSource® Video to FireWire Converter (Model# DFG/1394-1). This information was accessed as 8-bit black and white images using the image acquisition toolbox (Version 1.2) within Matlab (Release 14). A custom-written software program controlled the frame-rate and number of pictures; in the laboratory 25 images were collected at 25 Hz.

A methodology was developed to reduce error in collecting the discharge measurements. A custom-made stand was attached via clamps to the flume. Marks on the stand allowed good repeatability in the configuration, necessary to keep a constant field-of-view. Additionally, the camera mount was marked to the desired angles and leveled for each measurement. The Froude number was controlled by changing the depth with the tailgate. The flume manometers were calibrated before each measurement. The Marsh-McBirney flow meter was also calibrated for each experiment. A constant lighting source was provided with two 65-W fluorescent lights. The arrangement of the equipment within the flume (Figure 1) was designed to minimize

disturbance to the measuring devices (flow meter probe and surface of LSPIV image field) and ensure maximum accuracy with all measuring devices.

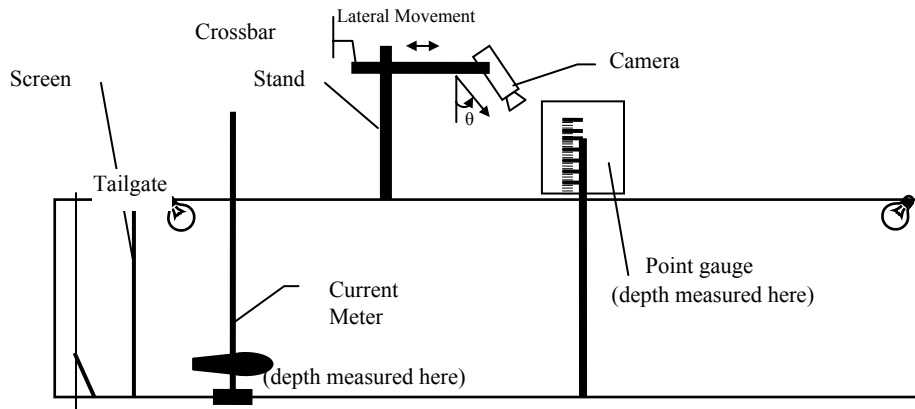


Figure 2. Laboratory setup for testing the LSPIV prototype.

The computation time to perform the image processing was large, about 50 hours. The image enhancement was slightly more time consuming than the image evaluation. An optimization of number of images and processing parameters (*e.g.*, interrogation window size) was necessary to maximize accuracy and minimize computation time. Additionally, two computers with Pentium 4 processors and 512 MB RAM were networked together. This setup allowed dual processing, as the DPIV software and Matlab programs could be run simultaneously. Use of dual processors significantly increased computation speed.

The images were enhanced to improve contrast and adjusted to the physical coordinate system with custom-written programs. The images are distorted by the lens and oblique camera angles. Therefore, a grid was used to convert camera positions to physical locations, such that velocity could be reliably measured. The same Matlab program included a histogram-equalization filter to improve contrast and cropping routines to minimize area for evaluation. The evaluation algorithms were run within DPIV (Version 1.0) used in collaboration with Dr. Pavlos Vlachos from the Mechanical Engineering department at Virginia Tech. The program was designed for aerodynamic research and is as robust as those used in previous LSPIV research. The input parameters for the evaluation of algorithms were manipulated through a Graphical User-Interface (GUI). The program includes window-offset methods to increase accuracy and validation/replacement routines to remove erroneous vectors. The image acquisition, evaluation, and averaging of user-defined parameters were optimized through a series of preliminary tests. After the laboratory tests were completed the prototype was modified for field conditions.

Field Experiments

The second objective of this study required the development of a LSPIV prototype for field conditions. As of publication, the prototype had been modified for field use and an operating procedure has been developed. The prototype requires a user-input of the water depth, cross-section survey, and ground-reference points (GRP). These data were obtained with Leica 400 survey equipment. The GRP allowed for the image to be transformed to physical coordinates. The custom-written programs can then output a discharge for a wide variety of flow conditions.

The field tests will be completed in September 2004 on low-order streams in the Blacksburg area.

RESULTS AND DISCUSSION

The laboratory results indicate that LSPIV is capable of accurately measuring discharge in certain situations. Initially, the effect of the seeding density on LSPIV accuracy was investigated. After the data were collected, a statistical analysis was conducted using SAS (Version 6.03). The relative error was calculated with the manometer as the control, using the equation:

$$\% \text{RelativeError} = \frac{[LSPIV(cfs) - manometer(cfs)]}{manometer(cfs)} \times 100$$

In the zero degree/low Froude number configuration, the relative error of the LSPIV prototype (where negative indicates under prediction) had a median of -15.4 % (Table 1). The data showed a trend of increasing accuracy with increased seeding density.

Table 1. Relative error of LSPIV (averaged between two replications) versus control for different seeding densities and Froude number/camera angle configurations.

Number of Particles Used	Low Froude number and 0 degree camera angle	High Froude number and 0 degree camera angle	Low Froude number and 30 degree camera angle	High Froude number and 30 degree camera angle
350	-37.8%	-54.1%	-55.1%	-23.3%
700	-15.4%	-49.1%	-36.5%	-17.9%
1050	-15.5%	-49.8%	17.7%	-16.0%
1400	-27.0%	-44.6%	15.8%	-15.0%
1750	-1.9%	-44.6%	14.8%	-16.9%

The zero degree angle/high Froude number configuration resulted in the largest error, with a median of -46.6 %. This could be a result of the combination of higher velocities and a smaller field-of-view which produced displacements that occurred outside of the 128² pixel interrogation area. Pixels are the basic unit in digital images; the images collected had a resolution of 480 X 640 pixels. Displacements outside the interrogation window can cause severe under-prediction of surface velocities. Additionally, out-of-plane losses are increased with increasing Froude numbers and could reduce surface velocity estimates. The 30 degree/low Froude number configuration produced the most accurate LSPIV discharges. The relative error ranged from -70.0 % to 28.9 %, with a median of 2.31 %. In this configuration, flow measurement accuracy increased with increasing seeding density. Finally, the 30 degree/high Froude number configuration had relative error values ranging from -28.9 % to -3.2 %, with a median of -19.2 %. In this configuration there was no trend in flow measurement accuracy with higher seeding densities.

A value of 0.95 was selected as the minimum level of statistical significance prior to the experiment. This was further defined to better interpret the data: 0.95 to 0.99 – moderately statistically significant, 0.99 to 0.9999 – very significant, and >0.9999 – very highly significant.

The first experiment determined that the interaction of Froude number, camera angle, and seeding density very highly affected the treatment results. The seeding density was found to have a moderately significant effect on LSPIV accuracy (p-value of 0.110). The trend of increasing accuracy with increasing seeding density is in agreement with findings of other researchers (Raffel *et al.* 1998).

For the second set of experiments, the seeding density was fixed at the highest level. This allowed investigation into four levels of Froude number and camera angle (Table 2). Using a camera angle of zero degrees the LSPIV prototype showed relative errors ranging from -28.6 % to 0.0 %, with a median of -8.20 %. In these experiments the LSPIV underpredicted the flow at low Froude numbers and overpredicted it at high Froude numbers. These results are likely caused due to the use of static input parameters. For example, the surface velocity correction factor was kept constant for all Froude numbers and corresponding depths. The surface correction factor should increase with decreasing depth, as the vertical velocity distribution becomes more uniform. This type of scaled input parameters were not used, and may be a source of error.

Table 2. Percent relative error of LSPIV prototype versus control flume manometer for four levels of Froude number and camera angles.

Froude number	0 degree camera angle	15 degree camera angle	30 degree camera angle	45 degree camera angle
0.05	8.9%	16.3%	-17.3%	-14.8%
0.15	-3.2%	0.6%	-16.7%	-21.7%
0.25	-14.6%	-3.5%	-25.3%	-25.0%
0.35	-28.3%	-7.8%	-22.9%	-22.2%

The 15 degree configuration resulted in the smallest relative errors for the LSPIV prototype. The error ranged from -15.74 % to 24.4 %, with a median of -2.74 %. The results show similar trends to the zero degree configurations in that it over-predicts the flow at lower Froude numbers and under-predicts it at higher Froude numbers. The LSPIV prototype under-predicted discharge at all Froude numbers in the 30 degree configuration. The error for this configuration ranged from -41.5 % to -5.7 %, with a median of -18.0 %. The 45 degree camera angle resulted in even poorer accuracy; relative error ranged from -27.3 % to 9.1 %, with a median of -21.77 %. The under-prediction at the 30 and 45 degree camera angle configurations may be caused by out-of-plane losses, especially at higher Froude numbers. Another possible source of error was due to the physical pixel changing with depth. Therefore, the physical pixel size had to be determined for each Froude number/camera angle combination, which resulted in another potential error source.

The statistical analysis of the data led to several important results. The Froude number (ranging from 0.05 to 0.35) very highly affected the results in all three treatments. Additionally, Tukey's multiple comparisons tests were performed and showed that all levels of Froude number significantly affected the LSPIV accuracy. Camera angle was also found to very highly affect the accuracy of the LSPIV prototype measurements. Multiple comparisons of the camera angle indicated that the flow measurements were not affected at angles below 30 degrees. However,

accuracy dropped significantly at camera angles above 30 degrees. Finally, the treatments were compared to test the accuracy of the LSPIV prototype and current meter against the control (flume manometers). The results indicated that both treatments means were statistically different than the control. Although, error using the LSPIV method was likely the cause of some of the differences, other factors may have contributed. For example, the control produced a discharge measurement with only two significant digits, versus three significant digits for both the LSPIV prototype and the current meter. Additionally, the discharge were sufficiently small (average \approx 0.15 cfs), where this lack of significant digits may have significantly affected the resulting flow computation. Finally, using multiple comparisons, it was found that there was no significant differences between the accuracy of flow measurements using the current and the LSPIV prototype, indicating that LSPIV could be a viable alternative for measuring stream flow.

CONCLUSIONS

There is a need for improved discharge monitoring techniques to reduce costs and thus increase the number of streams being monitored. LSPIV is an emerging technology for measuring discharge in streams and rivers. The preliminary results of this study have led to familiarization with equipment and procedures necessary for efficient discharge calculation. The laboratory proof-of-concept experiment produced good results. The following conclusions could be made:

- The Froude number does affect the accuracy of the Marsh-McBirney flow meter and the LSPIV prototype. Therefore, future applications may wish to use an adaptive method to determine input parameters based on flow conditions.
- The LSPIV resulted in poor flow measurements at camera angles above a 30 degree oblique angle. Therefore, field applications should use camera positions that reduce the oblique angle below 30 degrees. However, a zero degree camera angle may cause out-of-plane losses and reduce velocity measurements.
- There were no statistically significant differences in flow measurements made by the current meter and the LSPIV prototype in laboratory experiments.

Results could likely be improved by using an adaptive method to determine input parameters based on various flow conditions and using a control (*e.g.*, weir) with more accuracy than the manometer. However, the information gathered from the laboratory experiments will prove valuable in future field applications. A field proof-of-concept experiment will be conducted in fall 2004. Additionally, there is a need to automate the flow measurement process to produce a commercially viable system.

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IMPROVING SUSPENDED SEDIMENT LOAD ESTIMATION THROUGH CONTINUOUS TURBIDITY SENSING

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Transport of sediment into the Chesapeake Bay remains a significant water-quality issue with major implications for the overall health of the Bay ecosystem. Accurately estimating the suspended sediment concentrations and loads that are delivered to the Bay, however, remains an elusive goal. Although manual sampling of suspended sediment concentrations produces an accurate series of point-in-time measurements, robust extrapolation to unmeasured periods (especially high-flow periods) has proven difficult. Sediment concentrations typically have been estimated using regression relations between individual sediment sample concentrations and associated discharge values; however, suspended sediment transport during storm events is extremely variable and it is often difficult to relate a unique sediment concentration to a given stream discharge. With this limitation for estimating suspended sediment concentrations, innovative approaches for generating detailed records of suspended sediment concentrations are needed.

One promising new technology for improved suspended sediment determination involves the continuous monitoring of turbidity as a surrogate for suspended sediment concentrations. Turbidity measurements are theoretically well correlated to suspended sediment concentrations because turbidity represents a measure of water clarity and suspended sediments directly influence this measurement of clarity. To evaluate the use of turbidity as a sediment surrogate in large river systems, continuous turbidity monitoring has been initiated on the James and Rappahannock Rivers, as part of an ongoing water-quality monitoring project. These two rivers were selected because they represent major sediment contributors to the Chesapeake Bay. Initial results suggest that turbidity data will provide an improved method for generating a continuous record of suspended sediment concentrations, relative to the classical approach that uses discharge as a sediment surrogate. Further components of this approach include: presentation of the data on the internet in real-time; using the unexplained variance from the regression equations to quantify uncertainty in the estimated suspended sediment concentrations; and possible incorporation of these sediment data into a model of the Chesapeake Bay Watershed.

SYSTEMATIC SAMPLING OF BENTHIC MACROINVERTEBRATES IN SOUTHERN BLUE RIDGE STREAMS

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KEY WORDS: benthic macroinvertebrates, systematic sampling, biomonitoring, sedimentation

ABSTRACT

A systematic longitudinal transect (SLT) sampling design was recently developed for stream monitoring in the Chattahoochee National Forest (northern Georgia). This study aimed to evaluate biological information gained from the SLT design for developing biomonitoring techniques in mountain streams affected by sedimentation. Among eight streams, the SLT design produced a high number of rare taxa (68% of total taxa) with relative abundances less than 0.5%. However, four of those rare taxa showed significant relationships with substrate conditions, while three abundant and widely distributed taxa (*Ephemerella*, Orthoclaadiinae, and *Paraleptophlebia*) were not related to substrate conditions, suggesting those abundant taxa may be well-adapted to sandy streambeds. Seventy-four percent of the total number of individuals collected was associated with fast-water habitats, whereas slow-water habitats were comprised of both low total abundance and taxa showing no relationships with the sedimentation variables. Those findings suggest that sampling efforts concentrating on fast-water habitats (*i.e.*, those comprised of cobble and large gravel) may provide effective biological information to discern macroinvertebrate assemblage changes due to sedimentation. Additionally, assemblages separated by one kilometer showed low within-stream similarities, with the average taxon coefficients of variation per stream ranging from 83 to 132%, suggesting reach-scale physical conditions along relatively short, longitudinal stream distances should be considered when conducting regional, comparative bioassessments.

INTRODUCTION

A variety of sampling designs exist for collecting freshwater biota as part of biological assessment and monitoring programs (*e.g.*, USEPA Rapid Bioassessment Protocols; Barbour *et al.* 1999). For sampling benthic macroinvertebrates, habitat stratification is generally the method used and can involve a variety of modifications. For example, a stream reach can be marked with a particular habitat selected as the sample unit within that reach (*i.e.*, single habitat approach). Similarly, a multi-habitat approach includes sampling streambed habitats in proportion to their overall occurrence to form a composite stream sample or pooled habitat samples. In a review of contemporary field procedures, Voshell *et al.* (1989) found that 92% of lotic studies use some type of habitat stratification sampling design, which continues to be an effective sampling design for the development of biocriteria for state freshwater monitoring programs (Stribling *et al.* 1998, Gerritson *et al.* 2000, Pond and McMurray 2002).

An alternative strategy seldom used in stream studies is systematic sampling, which involves sampling at regularly spaced intervals (Krebs 1998). When associated with benthic macroinvertebrate sampling, a systematic design can include (1) sampling systematically within a reach, or (2) systematically selecting reaches. Unlike stratified sampling that aims to reduce sample variation associated with a particular habitat in order to facilitate stream comparisons, systematic sampling in streams can potentially produce a relatively larger range of community data due to the randomness of sampling locations and the associated variety of encountered streambed habitats.

A systematic longitudinal transect design (SLT) was recently developed for biological surveys conducted in the Chattahoochee National Forest (CNF, northern Georgia). Initial benthic macroinvertebrate sampling conducted by the USFS Center for Aquatic Technology Transfer (CATT) team was performed using a stratified, multi-habitat sampling design. However, the overall effectiveness of that type of sampling (in regards to biomonitoring) was considered potentially weak due to the massive volume of sample material obtained, attributable to the high physical streambed heterogeneity in low-order, mountain streams. Also, because of the large sample volume, a disproportionate subsample size was produced. The SLT design was developed to allow randomization and systematic standardization of sampling locations in order to combat problems associated with physically heterogeneous, mountain streams.

This study summarizes current findings regarding benthic macroinvertebrate assemblage and streambed condition information produced from the SLT design. Specifically, we reported macroinvertebrate assemblage traits (*i.e.*, taxa accumulation, taxa richness, abundance and diversity) and investigated (1) macroinvertebrate relationships with streambed sedimentation, and (2) within-stream macroinvertebrate assemblage variability among SLT reaches separated by one kilometer.

METHODS

Sites selected for macroinvertebrate sampling in April 2003 were in low-order, wadeable streams located in the eastern CNF (Southern Blue Ridge ecoregion). Stream systems in the region are typically dendritic and contained by landscapes consisting primarily of mica shists, mica gneiss, and aluminous shists geology, with highly erodible soils (USEPA 1999). Watersheds in our study were completely forested and relatively undisturbed with no roads or modern land-use activities. However, all streams in the region were affected by widespread historical timber harvests occurring in the early 1900s and physical stream conditions today are partially attributable to those activities (Dollof 1995, Harding *et al.* 1998).

At each site, macroinvertebrates were systematically sampled at 33 transects perpendicular to the channel, spaced at 3m intervals along a 100m reach. A random number table, with the number range corresponding to the approximate, maximum stream width, was used to select the distance from the left bank for sampling. A D-frame net (approximate sample area: 0.09m², mesh size: 500µm) was positioned along each transect and the streambed was agitated directly upstream of the net for 15 s to dislodge organisms. Because of the heterogeneous nature of the streambed and the systematic and randomized sampling locations, a variety of streambed conditions were

encountered. Therefore, a varying degree of streambed agitation and net movements were used during sampling, with the goal of equalizing the volume of water and benthos available for capture. Each macroinvertebrate sample was preserved separately with 70% ethanol and all macroinvertebrates were counted and identified to genus level using standard identification keys. Chironomidae were identified to subfamilies Orthocladiinae, Chironominae, or Tanypodinae or to the tribe Tanytarsini. All other Chironomidae were identified to the family level if no subfamily or lower level identification could be reached.

The variables *% deposited sediment* and *dominant substrate type* were visually estimated within a 0.25m² metal frame placed on the streambed at each sampling location. From the visual estimations, a variety of substrate variables were developed to summarize physical streambed conditions attributable to sedimentation (Table 1) and to investigate relationships of those variables with macroinvertebrates. In addition, the habitat type (fast-water and slow-water), water depth, and flow velocity was recorded at each macroinvertebrate sampling location in order to relate sedimentation-sensitive macroinvertebrates to their associated streambed habitats. Overall, a total of 264 macroinvertebrate samples were collected among eight streams, with an equal number of measurements per habitat variable and visually-estimated substrate variable.

To investigate within-stream variation in macroinvertebrate assemblages among SLT reaches separated by one kilometer, we used a dataset from sampling conducted in spring 2001 and 2002 by the CATT team. Sampling methods were similar to those described above, with the exception that transect samples were composited and then subsampled to obtain \approx 200 organisms.

Table 1. Streambed substrate variables visually estimated within a .25m² frame at each macroinvertebrate sampling location and the variables developed from those estimations.

<i>variables</i>	<i>developed variable</i>	<i>abbreviation</i>
1. <i>% deposited sediment</i>	a) average deposited sediment	DS
	b) proportion of total transects with greater than 50% deposited sediment	T>50
	c) average fast-water deposited sediment	F DS
	d) ratio of deposited sediment (slow-water / fast-water)	S/F DS
2. <i>dominant substrate</i>	a) proportion of transects with fines (< sand) as dominant substrate	% T fines
	b) proportion of transects with cobble, boulder or bedrock as dominant	% T cbb

Linear regression was used to investigate individual taxon log(x+1) abundance relationships with streambed substrate variables in Table 1. Taxa with non-normal distributions (*e.g.*, several non-zero abundance values) were not evaluated. Substrate variables reported as percent values were arcsine square root transformed and substrate variable ratios were square root transformed prior to analysis. Regressions were performed with Sigma Stat Version 3 (SPSS, Inc., Richmond, CA).

Analysis of macroinvertebrate assemblages among reaches within streams and separated by one kilometer was performed by calculating taxon abundance coefficients of variation (CVs) for the 20 most dominant taxa per reach and then calculating average taxon CVs per stream. CVs were calculated from raw abundance means and standard deviations using the column (*i.e.*, taxa) summary command in PC-ORD Version 4 (MjM Software Design, Glenden Beach, OR). Additionally, a nonmetric multidimensional scaling (NMS) ordination was performed, using

$\log(x+1)$ macroinvertebrate abundances and Sorenson (Bray-Curtis) distance measures, to examine within-stream differences in macroinvertebrate assemblages.

RESULTS AND DISCUSSION

A total of 115 taxa were collected among the eight sites. Taxa richness ranged from 59 to 73 with an average of 65.5 per site. Total number of individuals varied between 1467 and 2794, with a total streambed area sampled of 2.97m² (.09m² x 33 D-frame samples). The total area of streambed sampled with our systematic design is slightly higher than the average of 1.7m² reported in a survey of sampling methods used by U.S. state agencies (Carter and Resh 2001). Macroinvertebrate densities among streams averaged 720.8 individuals/m² (range; 492.9 – 940.7).

A variety of rare taxa removal criteria is used in analysis of macroinvertebrate assemblage patterns. For example, commonly used methods include (1) removal of species that occur as <0.5% of the sample abundance, (2) removal of taxa occurring in less than n samples, and (3) taxa with fewer than n individuals. An additional method is to remove taxa that contain fewer than n non-zero values, as implemented in PC-ORD. With some methods, caution must be taken to avoid removing important taxa that occur as highly abundant singletons (*i.e.*, only 1 occurrence over the entire sample). Our dataset contained 115 taxa, of which 68% (77 taxa) and 28% (32 taxa) were considered rare based on the criteria of (1) and (3)($n < 5$) above, respectively. In one of the apparently few studies using a systematic sampling design, Snyder *et al.* (2002) identified 64 rare taxa (42%) occurring at fewer than 4 sites in Delaware streams. Roy *et al.* (2003) eliminated 24 rare taxa, out of 91 total taxa, as a result of sampling riffle, pool, and bank habitats in streams located in north-central Georgia. Although using a multi-habitat approach allows sampling a variety of habitats and a relatively large streambed area, sampling is stratified nonetheless, which reduces the number of rare taxa collected when compared to a random, systematic design.

A total of 97 slow-water and 167 fast-water habitats were sampled over the eight streams. Results of ANOVA showed that those habitats differed significantly in both depth ($P < 0.001$) and flow velocity ($P < 0.001$). Fast-water and slow-water habitat types also varied in dominant substrate type and occurrence over the range of transects, with notable differences in the number of transects per habitat type comprised of sand or cobble (Figure 1). Overall, a total of 17,378 individuals were collected among the eight streams, of which 74% were associated with fast-water habitats. Across all transect samples, 88 taxa were collected from slow-water and 101 taxa were collected from fast-water habitats.

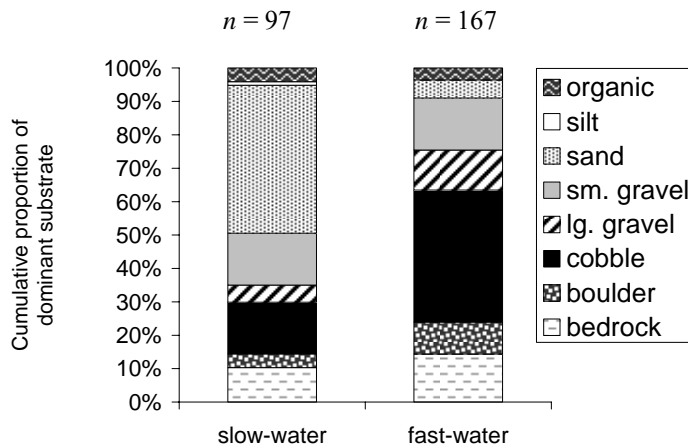


Figure 1. Proportion of dominant substrate type occurring across two habitat types.

One of three taxa was consistently the single most dominant taxon among the 8 streams: *Ephemerella*, *Leuctra*, and the Chironomid subfamily Orthocladiinae. Among two sites, *Ephemerella* and Orthocladiinae were the dominant single taxon comprising 20 and 22% of the total abundance, respectively. Proportional abundances of the 10 most dominant taxa varied from 62 to 81% with an average of 71.4% per site. Although sites were dominated by a few very abundant taxa, Simpson's diversity index was high and varied little among sites (≈ 0.97 to 0.98) due to the high number of taxa present at each site.

Among a list of the 15 dominant taxa occurring at each site (total taxa = 34), three were collected from all 8 sites: *Ephemerella*, Orthocladiinae, and *Paraleptophlebia*. The ubiquitous distribution of those three taxa among sites suggests strong adaptation, and possibly preference, for deposited sediment conditions generally present at our sites. Of the dominant taxa, 14 were collected from only one site, suggesting that unique conditions may persist within a particular stream and provide resources to accommodate a very abundant, yet uncommon, particular taxon.

Fifty-two taxa, excluding rares, had abundances at least 50% larger in fast-water habitats when compared to slow-water habitats. Only two taxa with overall abundances over 5 individuals were found in greater abundances in slow-water habitats, *Eurylophella* and *Ephemerella*. A number of taxa with abundances greater than 50 individuals were collected from both slow- and fast-water habitats including *Tallaperla*, *Amphinemura*, *Leuctra*, *Ephemerella*, *Paraleptophlebia*, *Baetis*, *Optioservis*, *Oulimnius*, *Hexatoma*, Chironominae, Tanytarsini, Tanypodinae, and Orthocladiinae. However, those taxa all had greater abundances in the fast-water habitats.

Results of regression analysis showed that 11 taxa had significant relationships with at least one streambed substrate variable (Table 2). Of those taxa, six were among the 15 most dominant taxa in at least one stream. However, the three highly abundant and widely distributed taxa (*Ephemerella*, Orthocladiinae, and *Paraleptophlebia*) showed no relationship with substrate variables, further suggesting that those taxa are strongly adapted to sandy streambeds. Tanypodinae comprised the highest overall relative abundance (4.3%) of those taxa shown to be sensitive to substrate conditions (Table 2). Sixty-two taxa occurred in lower abundances than *Isoperla*, which was the least abundant taxa significantly related to substrate conditions. Those 62 taxa had proportional abundances less than 0.16%, suggesting a potential cutoff point for rare taxa of $\approx 0.1\%$ when investigating assemblage patterns in CNF mountain streams.

All taxa that were significantly related to at least one substrate variable were found in greater abundance in fast-water habitats as determined by abundance ratios from both habitat types. Seven taxa had fast-water/slow-water abundance ratios greater than 4:1. *Doliphilodes* had 14 times more individuals in fast-water habitats and showed significant relationships with a deposited sediment range in fast-water habitats (Table 2). Oligochaeta and Tanypodinae were significantly related to substrate variables and occurred in high abundances over both fast- and slow-water habitats. Analysis of abundance relationships with substrate conditions for those individual taxa showed that slow-water abundances for Tanypodinae were not significantly related to the streambed variable % *T fines* ($P = 0.965$), yet fast-water abundances were significant ($r^2 = 0.72$, $P = 0.008$), showing a slightly higher r^2 value when compared to the associated value in Table 2 that resulted from combining abundances from both habitat types. However, Oligochaeta relationships were only significant when habitat-type abundances were combined. Those findings suggest that deposited-sediment sensitive taxa are more associated with fast-water habitats and those taxa occurring in slow-water habitats may be less affected by a gradient of deposited fine sediment, due to the natural occurrence of sand in many of the slow-water habitats (Figure 1) and pre-adaptations of organisms that allow greater tolerance to deposited fine sediments.

Table 2. Linear regression statistics showing significant relationships of individual taxa and streambed substrate variables. r.a. = overall relative abundance.

Taxon	r.a.	r^2 / P -value					
		% <i>T fines</i>	<i>T > 50</i>	<i>S/F DS</i>	<i>F DS</i>	% <i>T cbb</i>	<i>DS</i>
Oligochaeta	1.78	(-) 0.54 / 0.036	*	*	*	*	*
Perlidae	1.08	*	*	0.56 / 0.034	*	0.71 / 0.005	*
Acroneuria	0.39	*	*	*	(-) 0.51 / 0.048	*	*
Isoperla	0.17	0.57 / 0.031	*	*	*	*	*
Remenus	0.59	*	*	(-) 0.60 / 0.024	*	*	*
Leucrocuta	0.31	(-) 0.61 / 0.022	(-) 0.72 / 0.008	0.64 / 0.016	(-) 0.70 / 0.006	*	(-) 0.74 / 0.004
Diplectrona	3.53	*	*	(-) 0.66 / 0.014	0.67 / 0.013	*	*
Doliphilodes	0.70	*	*	0.50 / 0.049	(-) 0.51 / 0.045	*	*
Polcentropus	0.45	*	*	*	*	0.53 / 0.038	*
Tanypodinae	4.31	(-) 0.70 / 0.009	(-) 0.59 / 0.024	0.60 / 0.023	(-) 0.77 / 0.004	*	(-) 0.57 / 0.030
Rhyacophila	0.86	(-) 0.50 / 0.049	*	*	*	*	*

Average coefficients of variation for the 20 dominant taxa abundances per stream ranged from 83 to 132%, suggesting high within-stream variability of macroinvertebrate assemblages among systematic reaches separated by one kilometer. Additionally, NMS produced a three dimensional ordination that explained 79% of the variation in the original sample space, with systematic reaches showing low within-stream similarities (Figure 2).

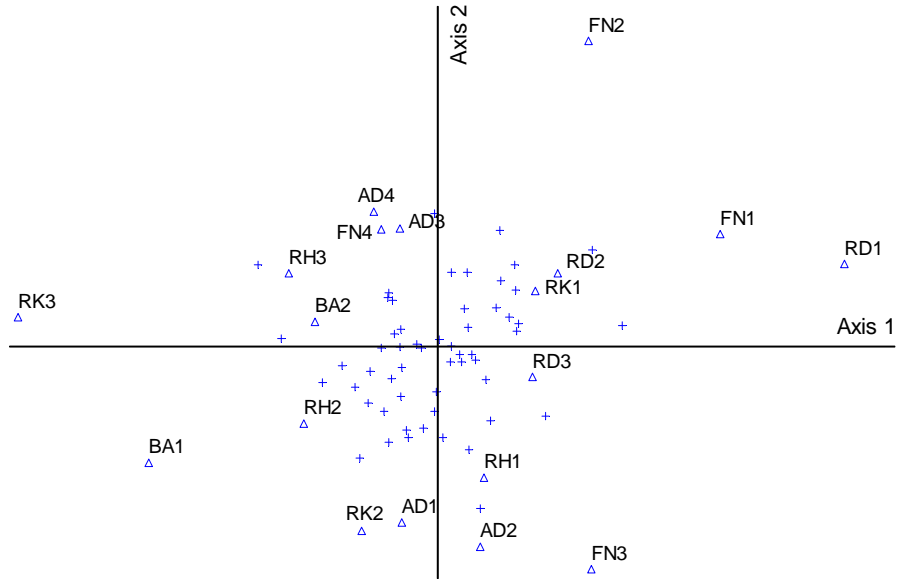


Figure 2. Nonmetric multidimensional scaling ordination of macroinvertebrate assemblages collected from systematic reaches in streams separated by one kilometer. Abbreviations are streams, number represent individual reaches within a particular stream. + = taxa

High physical variability exists within mountain catchments (*e.g.*, geology, vegetation, valley slope and channel confinement). Those physical landscape conditions influence streambeds and provide a variety of physical habitat templates used by macroinvertebrate assemblages differing in both taxonomic structure and overall function (Huryn and Wallace 1987). Because the SLT design incorporates both systematic (*i.e.*, equal intervals along the streambed) and random (*i.e.*, randomized location across transects), it may effectively reflect heterogeneous streambed habitat templates and lead to high-within stream estimates of assemblage variation as shown in Figure 2. In contrast, sampling macroinvertebrates among particular habitats using stratified sampling aims to lessen habitat dissimilarities among reaches, which may lead to less accurate estimates of within-stream assemblage variability.

In brief conclusion, the SLT design provided a comprehensive list of taxa occurring along stream reaches affected by a range of persistent deposited sediment conditions, which could benefit synoptic biological surveys designed to produce thorough taxonomic lists. Due to the large number of rare taxa collected and the number of highly abundant taxa showing no relationship with sedimentation variables, overall biological assemblage patterns remain vague. However, abundances of a small number of taxa were shown to vary significantly with sedimentation variables, and that information provides a starting point for the development of overall assemblage information (*i.e.*, metrics) responsive to sedimentation. By separating transect samples into slow-water and fast-water habitat types, we were able to associate our few sensitive taxa to particular habitat conditions, which will provide further direction in the development of effective sampling strategies. Future studies that involve dividing those general habitat types into substrate types and sub-habitat conditions (*e.g.*, dominant and subdominant substrate, % deposited sediment) may reveal relatively smaller scale macroinvertebrate assemblage patterns and provide further information for refining macroinvertebrate sampling designs. In addition,

investigations regarding the distribution and ecology of the dominant taxa occurring among our sites (e.g., the ubiquitous distribution of *Ephemera*, *Paraleptophlebia* and Orthoclaadiinae) could potentially show that macroinvertebrates have adapted towards distinct functional and structural assemblages along sandy streambeds shaped by historical logging disturbances.

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A NEW WATER QUALITY MONITORING AND ASSESSMENT TOOL FOR VIRGINIA'S FRESHWATER STREAMS

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KEY WORDS: statistical analysis; random survey design; water quality monitoring; and multi-metric index.

ABSTRACT

The Virginia Department of Environmental Quality's (VDEQ) biological and ambient water quality monitoring programs have historically used a targeted approach for monitoring the Commonwealth's aquatic resources. This is necessary for verifying regulatory compliance of pollution sources and tracking local pollution events. However, it is difficult to translate the data into estimates of water quality conditions across an entire state or river basin. Consequently, in 2001, VDEQ began a five-year probabilistic monitoring program (ProbMon) for non-tidal streams. ProbMon incorporates a random survey design that allows VDEQ to produce an accurate assessment of chemical, physical, and biological conditions in 1st through 5th order streams. This is the first survey that will provide estimates of the status of Virginia's aquatic resources with statistical confidence. Three years of data are presented for statewide benthic macroinvertebrates (n=159), physical habitat assessments (n=157), and chemical data (n=173).

INTRODUCTION

In response to the need to evaluate water quality of entire river basins or over the whole state, VDEQ added probabilistic monitoring (ProbMon) to its biological and chemical monitoring program in 2001. The goal is to provide an accurate assessment of regional chemical, physical, and biological conditions. Specifically, the ProbMon survey provides 1) estimates of the geographic coverage and extent of aquatic resource conditions with known confidence; 2) estimates of the current status, and a basis to determine trends and changes in indicators of aquatic resources with confidence; 3) statistical summaries and assessments of aquatic resources; and 4) a description of associations between indicators of natural and anthropogenic stressors and the condition of aquatic resources.

The focus of ProbMon is on non-tidal perennial streams. The station locations have been selected randomly to allow the expression of water quality conditions in statistical terms. That is, a point value can be generated with an estimate of its precision. Data will be collected from approximately 300 stream locations over a five-year period. The survey is evenly spread over the period 2001-2005, with approximately 60 locations sampled each year, to incorporate wet, dry, and normal precipitation years in the database.

METHODS

Data Collection

In 2001, field teams measured the habitat and benthic communities in the spring and fall, and stream chemistry in the fall. During subsequent years, chemical data was collected in the spring along with habitat and benthic macroinvertebrates in the spring and fall. Water and sediment samples and field parameters are collected according to VDEQ Standard Operating Procedures (VDEQ 2003). The Environmental Protection Agency's (EPA) Rapid Bioassessment Protocols (RBP) were used to evaluate physical habitat and guide benthic macroinvertebrate sampling (Barbour *et. al.* 1999). EPA's Relative Bed Stability methods were added in year 3 (Kaufmann *et. al.* 1999). In all, 79 chemical and physical parameters were measured at each site along with dissolved oxygen, temperature, pH, and specific conductance.

Station Siting

ProbMon employs a random survey design to select stream sample sites based on EPA's Environmental Monitoring and Assessment Program (EMAP) (Stevens 1997). A grid of hexagons was placed over the Commonwealth of Virginia (Figure 1). This grid ensures randomization and spatial distribution of sampling locations. The base density is one grid point per 640 km². This was intensified to allow regional analyses. The 640 km² hexagons are subdivided into 7 hexagons of 90 km² each. Next, the 90 km² hexagons are subdivided into 7 hexagons that cover 13 km² each. Finally, within the 13 km² hexals there are 7 hexagons that cover 1.8 km² of land surface. The sample areas were the 13 km² hexals whose edges are defined by the 1.8 km² hexagons (Figure 1). Thus, a 7 x 7 x 7 fold enhancement is used to randomly select stream reaches (Olsen 1999).

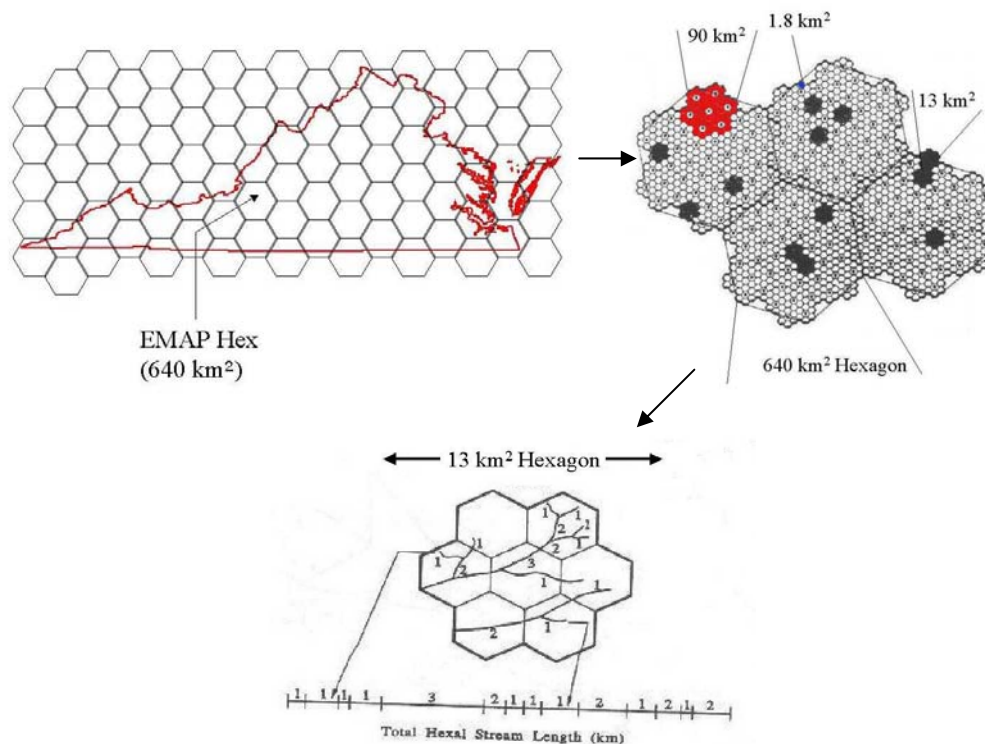


Figure 1. ProbMon station siting using random survey design.

In ProbMon, Strahler stream order is used to assign the probability of selection to each stream segment to avoid over selecting the more common stream sizes. For example, 1st order streams make up 65% of Virginia stream kilometers and are four times as common as 2nd order streams. Thus 1st order streams are four times as likely to be selected by a simple random design. High order streams occur less frequently and could potentially be underrepresented to the point that their statistics would be meaningless. Thus, the frequency of occurrence of each stream order was used to weight that order and ensure that all stream orders have an equal chance of being selected.

Data Analysis

Boxplots were created with STATISTICA 5.1 to permit the comparison of ProbMon variables by Strahler order. Many environmental parameters are not normally distributed, thus the median was consistently used to compare results across stream orders. Another method of analysis involved the manipulation of the data to generate the cumulative distribution function (CDF) for key variables (A. Olsen, personal communication 2000). This function is most useful when displayed as a CDF curve. CDF curves estimate the probability that a variable is less than or equal to some value. Furthermore, the likelihood that a variable would be less than a particular threshold may be interpreted from the CDF curve. It can also provide the probability that a variable would be above a threshold or within a certain range.

More information regarding survey design and data analysis is available in the 2001 ProbMon Report and EPA's EMAP website, <http://www.epa.gov/nheerl/arm>.

The Draft Stream Condition Index (SCI; Tetra Tech, Inc. 2003) is a multi-metric index based on historical benthic macroinvertebrate community data. It is applied to the ProbMon dataset as a potential assessment tool and to provide further evaluation of its Ecoregional validity.

RESULTS

First-year results from the ProbMon survey are presented in the 2001 ProbMon report (VDEQ 2003). The results presented in this paper are based on data collected in the spring of 2001 through fall 2003. The map shows ProbMon sampling sites layered with Virginia Ecoregions (Figure 2).

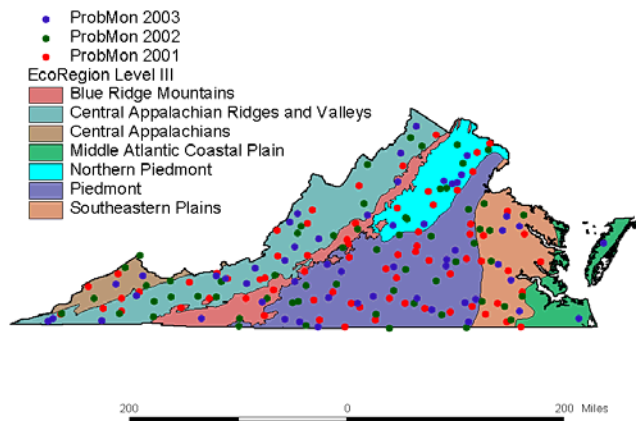


Figure 2. Map of ProbMon station locations by ecoregion.

Ecoregions are grouped as follows: Piedmont Ecoregion = Piedmont and Northern Piedmont Ecoregions; Mountain = Central Appalachians, Central Appalachian Ridges and Valleys, and Blue Ridge Mountains; Coast = Mid-Atlantic Coastal Plains and Southeastern Plains.

Water Chemistry

Fecal coliform bacteria results are compared to VDEQ's bacteria standards in Figure 3. 12% of stream kilometers (+/- 8% margin of error) exceeded the instantaneous standard or 400 cfu/100ml. Conversely, over 80% of stream kilometers met fecal coliform standards.

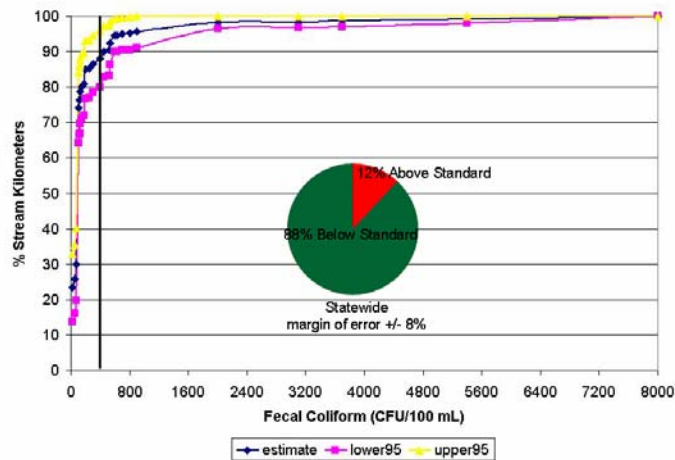


Figure 3. CDF curve and pie chart of fecal coliform bacteria by ecoregion. The 95% confidence intervals are shown above and below the CDF curve. The vertical bar represents the instantaneous standard for fecal coliform bacteria (400 colony forming units per 100 ml water). The pie chart represents percentages of stream kilometers that are above or below the instantaneous standard.

Currently, nutrient criteria are not included in the Commonwealth's Water Quality Standards. To gauge ProbMon data, limits were applied at 0.05 and 0.1 mg/l for total phosphorus in the Mountain and Piedmont Ecoregions to produce the pie charts in Figure 4. The aforementioned thresholds were chosen based on the Mid-Atlantic Highlands Streams Assessment (EPA 2000). The majority of streams have low total phosphorus according to the CDF curves. Based on the 0.05 mg/l limit, 97% of Mountain Ecoregion streams and 79% of Piedmont Ecoregion streams were considered *good*.

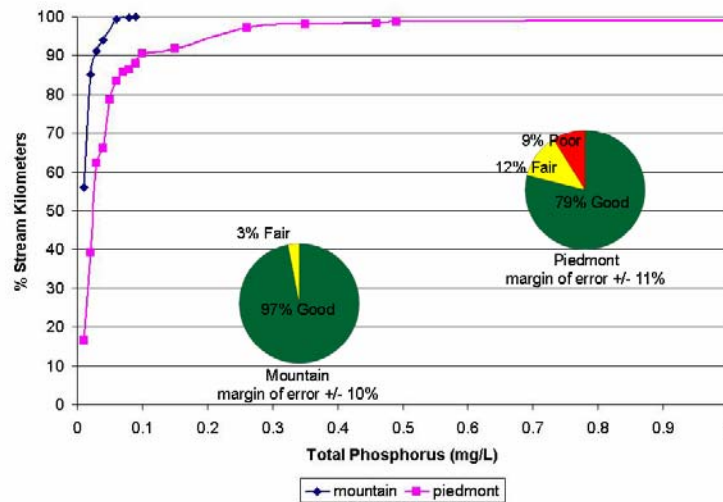


Figure 4. Total phosphorus CDF curves and pie charts for the Mountain and Piedmont Ecoregions. *Good*, *fair*, and *poor* categories are designated as less than 0.05 mg/l, 0.05 to 0.1 mg/l, and greater than 0.1 mg/l, respectively (EPA 2000). Due to the presence of extreme outliers, the x-axis in Figure 4 was cut off at 1 mg/l in order to show detail on the lower end.

Nitrogen in streams may indicate the presence of fertilizer, acid rain, and/or sewage discharges. Total nitrogen is comprised of nitrate (NO₃), nitrite (NO₂), ammonia (NH₄), and Total Kjeldahl Nitrogen (TKN). ProbMon results show that the dominant form of nitrogen in Mountain Ecoregions differs from other Ecoregions. The Coastal Ecoregion, which is similar to the Piedmont, was used for comparison. Nitrate (67%) is the dominant form in the Mountain Ecoregions (Figure 5). Total Kjeldahl Nitrogen (TKN) is the dominant form in the Piedmont and Coastal Ecoregions. Median total nitrogen is around 0.5 mg/l for both Piedmont and Mountain Ecoregions (Figure 5). Percentages of nitrite and ammonia were low in all three Ecoregions.

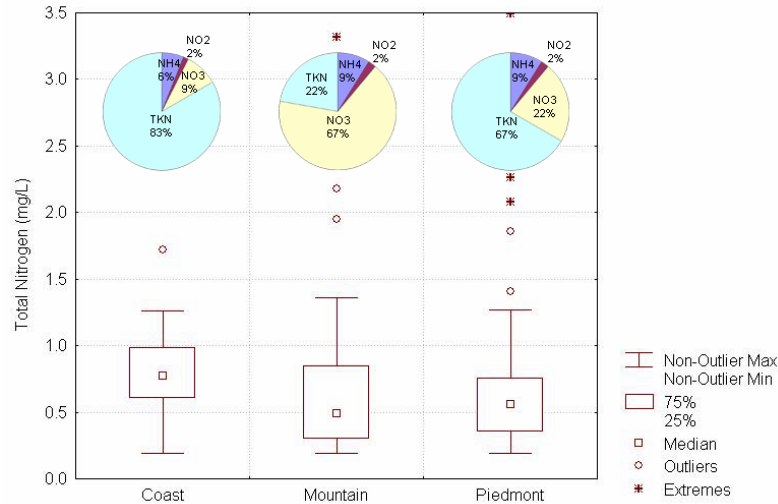


Figure 5. Total nitrogen boxplots and pie charts for the Coast, Mountain and Piedmont Ecoregions. The pie charts display the forms of nitrogen (as a percentage of total nitrogen) in each ecoregion.

Benthic Macroinvertebrate Communities

In addition to the RBP assessment method, VDEQ uses the SCI to evaluate benthic macroinvertebrate communities. To check its validity and application to ProbMon data, the SCI was used to generate CDF curves by Ecoregion.

Preliminary screening SCI values of >60 indicate a *good* benthic macroinvertebrate community, between 40 and 60 is considered *fair* and <40 is *poor*. Based on 3 years of data, 75% of stream kilometers fell into the *good* and *fair* categories. However, Mountain and Piedmont Ecoregions showed differences. Half of the Mountain Ecoregion stream kilometers received *good* SCI scores compared to 28% of Piedmont Ecoregion stream kilometers (Figure 6).

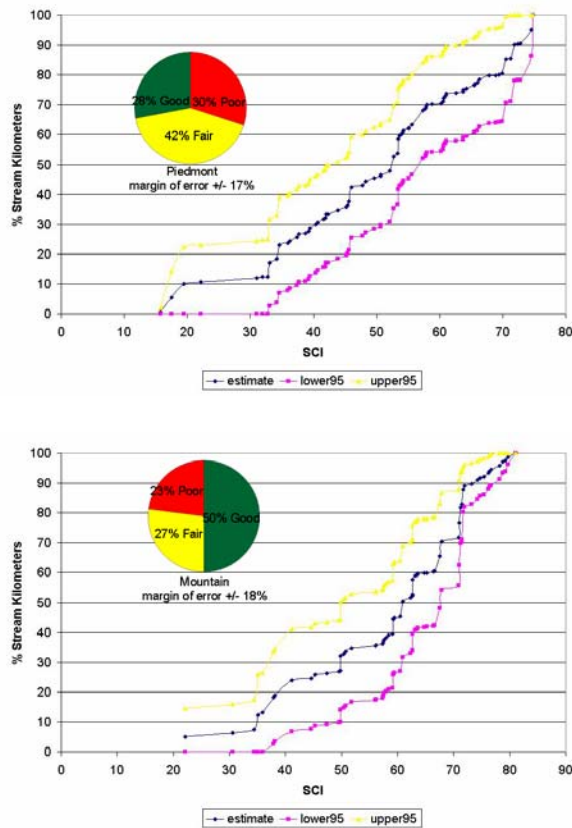


Figure 6. CDF curves and pie charts of Stream Condition Index (SCI) scores for Piedmont and Mountain Ecoregions. The 95% confidence intervals are displayed above and below CDF curves. The pie charts show the percentage of stream kilometers that have SCI scores >60 (*Good*), between 40 and 60 (*Fair*), and <40 (*Poor*).

DISCUSSION

ProbMon has introduced a practical approach to evaluating water quality in the Commonwealth. It uses a concentrated monitoring effort to address broader questions than traditional targeted monitoring. VDEQ is beginning to see trends develop with these data. As more data is collected, Ecoregional separation may become more evident. The incorporation of the SCI raises questions about its statewide application as an assessment tool. Differences among Ecoregions were noticeable in the nutrient data as well.

Fecal coliform bacteria results demonstrate that the majority of stream kilometers are meeting Water Quality Standards. *Escheria coli* will replace fecal coliform bacteria as an indicator of surface water pathogens by 2008 and it will be interesting to see if it exhibits the same patterns.

Nutrient enrichment is generally a result of runoff from non-point sources like agriculture and lawns or from point sources such as municipalities and industry. Total nitrogen trends are similar for both Mountain and Piedmont Ecoregions. When the data are presented as percentages of total nitrogen, the ecoregions are clearly different. TKN is the dominant form in the Piedmont whereas nitrate makes up more than half of the total nitrogen in the Mountains. Total phosphorus concentrations are higher in Piedmont Ecoregion streams than in Mountain Ecoregion streams.

As the dataset increases, the SCI will be refined and may show that individual metric scores (*e.g.*, Taxa Richness, EPT, *etc.*) need to be tuned to the Ecoregion. Reference streams in the Mountain Ecoregion dominated the dataset used to develop the SCI because few Piedmont Ecoregion reference sites were available. As a result, the SCI may be more accurate for the Mountain Ecoregions. Due to random station siting techniques, many ProbMon sampling locations end up in places that have little anthropogenic impact. As a result, ProbMon has helped identify new reference sites for all Ecoregions.

Future Trends

In this brief glimpse into Virginia's ProbMon program, only a fraction of the data could be presented. ProbMon includes a habitat assessment component to determine the percent of streams that are exceptional as well as degraded. One of the more elusive parameters to quantify is sediment deposition. In 2003, VDEQ began collecting quantitative physical habitat data to determine whether sediment sources are anthropogenic or natural (Rosgen 2001). A modified version of the Relative Bed Stability method is used to answer this question (Kaufmann *et. al.* 1999).

Another part of the freshwater ProbMon program involves Geographic Information System analysis of land cover upstream of ProbMon sites. The land surrounding a water body can impact water quality, thereby altering the physical habitat and biological community. VDEQ intends to create a filtering matrix of habitat, benthic macroinvertebrate, and chemical data to identify potential reference sites using land cover data. As ProbMon progresses, the quantity and variety of data collected will enable VDEQ to detect relationships among physical habitat, biological communities, land cover and water quality.

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PCB TMDL SOURCE ASSESSMENT STUDY, BLUESTONE RIVER WATERSHED

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KEYWORDS: SPMD, TMDL, watershed, PCB, sampling plan, Bluestone River

ABSTRACT

The Department of Environmental Quality (DEQ) has completed a polychlorinated biphenyl, (PCB) source assessment study for the Bluestone River Watershed in Virginia. This watershed is located within the "Coal Fields" of Virginia and West Virginia where some reports indicate over 300 mining industry related facilities existed, many of which either used or handled PCB oils. The task of source assessment was made more difficult since many operations have either closed, were located in the remote mountainous valleys and ridges of the watershed, or were operated as home-based industries. The goals of the Bluestone study were to ensure that no site is actively contributing PCBs to the watershed and to provide basis for the PCB Total Maximum Daily Load (TMDL) study required by the Clean Water Act. One of the innovative aspects of the project was the use of semi permeable membrane devices (SPMDs) to determine the contribution of PCB from the major tributaries to the Bluestone.

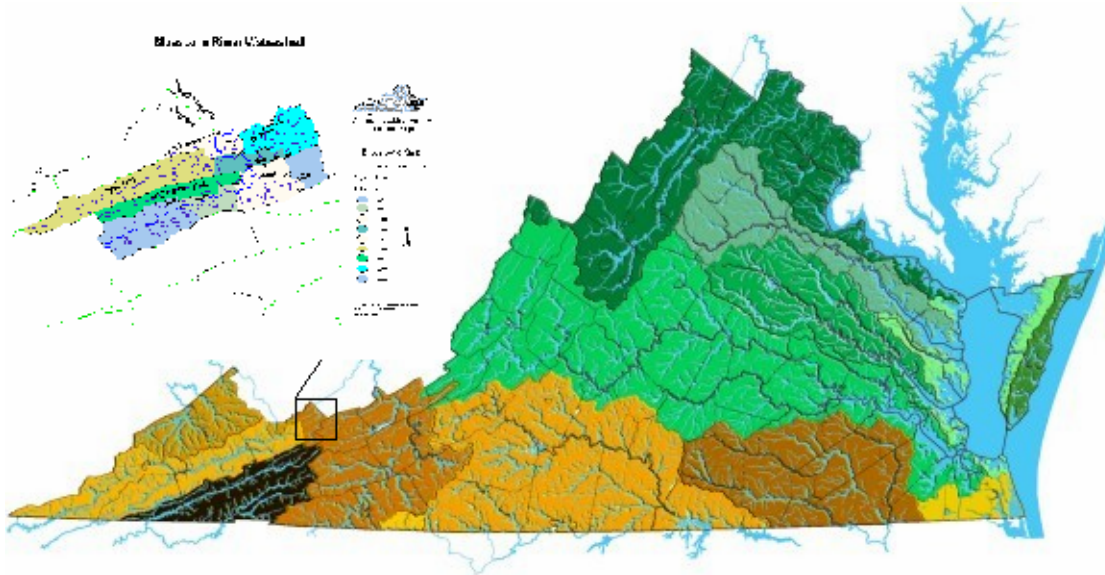
The study resulted in locating two sites that will require remedial PCB activities. Both sites are located in the State of West Virginia. USEPA and the West Virginia Department of Environmental Protection (DEP) will direct the cleanup efforts. The DEQ activities in the watershed had some other benefits such as prompting the removal of trash and metal from the stream bank, removal of barrels full of oily fluids, the repair of a broken sewer line along the stream, and the development of a new local Bluestone River conservation organization.

PROJECT DESCRIPTION

Several factors complicated the Bluestone River Watershed PCB Source Assessment Study:

- 1) Most misuse of PCBs occurred decades ago.
- 2) The watershed is dissected by the Virginia, West Virginia State Line (Figure 1). Tributaries from industrial areas in West Virginia enter Virginia and ultimately the Bluestone exits Virginia into West Virginia. Therefore, DEQ worked closely with the State of West Virginia Department of Environmental Protection (DEP) and the USEPA.
- 3) Two surface water intakes supply drinking water to the Virginia portion of the watershed. Citizens and agencies were concerned about the impacts to these potable water sources.
- 4) Karst geology moved water from the surface through underground pathways near known PCB contaminated sites.
- 5) PCB transport mechanisms and bioaccumulation interactions are different in each watershed and even within a subwatershed.

Over the past several years, the Commonwealth of Virginia has been developing an approach for evaluation of state waters for Persistent Bioaccumulative Toxins, PBTs, such as methyl mercury, Dioxin, and PCBs. The first step is a study of the extent of contamination in each state watershed by testing one or more samples of fish and sediment at very low concentration levels. The congener analytical method has been used for PCBs. When a high level of PCBs was found in the Bluestone River, in both fish and sediment, a second more intense study was implemented.



Bluestone River Watershed

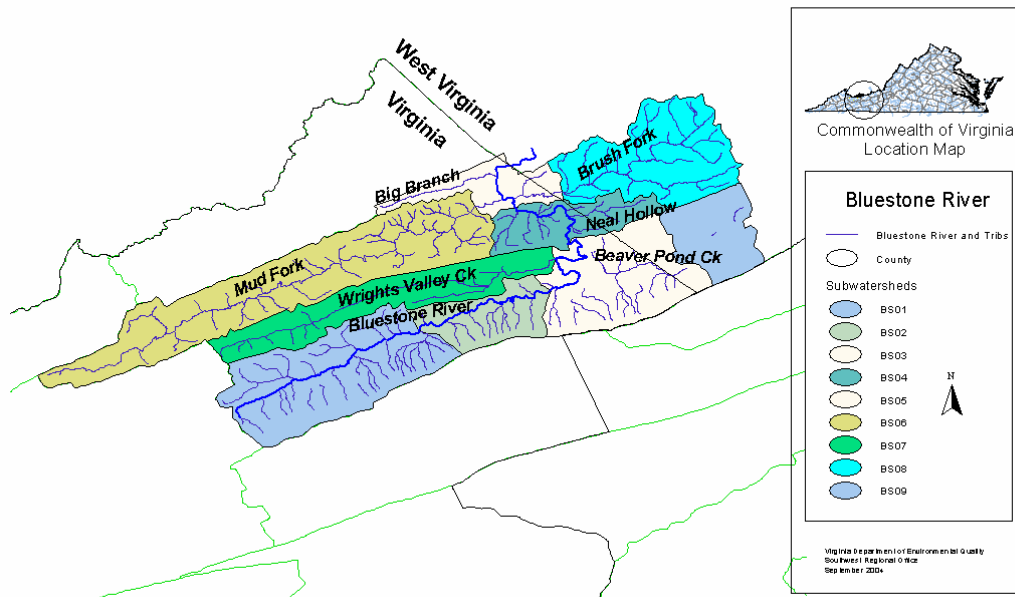


Figure 1A and B. Maps of Bluestone River Watershed; a subset of Virginia.

SOURCE ASSESSMENT EFFORTS

During the first phase, the Bluestone River was sampled in two locations for fish and sediment in August, 2000. The results came back demonstrating a significant PCB contamination level at the state line and decreased amounts in the same species at the further upstream sampling location. Various fish species were sampled and indicated different levels of uptake based upon many factors including their feeding habits, size/age, bioaccumulation and metabolic rates, fat content, habitat temperature, *etc.* The historic VDH advisory level for PCB contamination in fish tissue is 600 ($\mu\text{g}/\text{kg}$) ppb in edible fillets. The highest concentration found in the first round of PCB in fish sampling of this watershed was 2,368 ppb in a composite of 3 carp. This sample was taken south of the VA and WV state line at Yards, VA. As a result of the initial findings, the Virginia portion of the Bluestone River was posted by the Virginia Department of Health, VDH, with a fish consumption advisory for carp, for most of its fishable waters. Although the original Bluestone River fish consumption advisory was based upon 600 ppb, the VDH is in the process of dropping the level to 50 ppb of total PCB in edible fish fillets.

In response to these findings, DEQ initiated a source assessment study to locate potential sites that are contributing PCBs to the watershed. The study included the following elements:

1) Involvement of the Stakeholders from the Watershed

One of the initial steps in the evaluation of the watershed, involved enlisting the aid of the stakeholders in the watershed. These are persons who knew or could find out about the historical PCB related activities in the watershed. The watershed stakeholders were informed of the data and the PCB source assessment study with public notices and two TMDL meetings for the bacterial and benthic impairments in the Fall and Winter, 2003. Through discussions with the stakeholders, it became apparent that there was much public concern and stress over the potential health problems from drinking water intakes. The VDH Bluestone River fish consumption advisory was based upon the finding of PCBs in fish tissue above the VDH standard. However, few people are thought to consume fish from the Bluestone watershed above the recommended frequencies. Many more residents rely on the drinking water produced at the Bluefield Virginia Water Treatment Plant, BVWTP, which has the intake located within an impaired segment listed for fish consumption due to PCBs.

2) Investigation of the contamination of the Potable Water Supply

In response to public concerns, sampling was performed to evaluate the raw water and potable water at the BVWTP for presence of PCBs. VDH coordinated the low level drinking water analysis effort. This testing revealed that the PCB levels in the drinking water were not detected, compared to the maximum contaminant levels for potable water as defined by the regulations supporting the Safe Drinking Water Act.

DEQ performed support sampling efforts at the request of VDH, both at and upstream of the raw water intake on both fish tissue and sediment. Fish may range upstream of the raw water intake, but downstream travel is obstructed by a 6 foot spillway just below the intake. The carp and other fish species which were collected, were several years old and would have represented an average exposure through time and weather events.

Coordination was done with VaDGIF (Virginia Department of Game and Inland Fisheries), the DEQ Sampling and Monitoring group, and local residents, to sample fish at the intake reservoir and take sediments from strategic locations above the water supply. This follow up sampling event occurred October 2003. The fish tissue data revealed that carp had 300 ppb of PCBs. The sediment results indicate fairly low concentrations in various locations above the drinking water source, ranging from 0.32 ng/g at the BVWTP to 8.97 ng/g upstream on the mainstem. Even if fish in the reservoir were exposed to a concentration in the sub ppb (ng/g) range, evidence supports that these low levels are enough to reach PCB concentration levels in the fish above the edible fish tissue consumption advisory level of 50 ppb.

3) Research Existing Records

Research was performed on existing records of PCB related activities of state and federal agencies. These records include PCB complaints, spills, emergency responses, *etc.* Several lists were generated including one of property owners in the watershed using the Tazewell County tax map. Another list was made of current and historic businesses related to PCB oil activities, which included anecdotes of PCB misuse on private residential or public properties. Next, records of historical PCB activities were researched and reviewed from various EPA websites including the EnviroMapper, CERCLA-Superfund, and TRI sites. Additionally, research was performed on activities from VaDEQ PREP files, VaDEQs Waste Program files, WVDEP, WVDHHR, EPA On-Scene Coordinator (OSCs), Bluefield Fire Department, and Virginia Office of Emergency Services. Among the information compiled were PCB sampling schemes, data, and in some cases reports of activity, cleanups, and site closures in the watershed. Many of the regulatory and responsible authority contacts had responded personally to historic PCB related events in the Bluestone watershed.

4) Collect Stakeholder Information

During the fourth task, potential sources were identified by personal contacts and results of survey forms to gain information about historic PCB usage in the watershed. Contacts with the stakeholders in the watershed were the primary source of information leading to several contaminated sites.

The Bluestone River PCB Source Assessment survey consisted of a two-page questionnaire, which was tailored mainly toward industry and reporting of industrial activities. A second version of the survey was developed to communicate with stakeholders in the watershed who were not related directly to industrial PCB activities. Over one hundred fifty surveys were handed out or mailed out in the Bluestone River watershed in Virginia and West Virginia to industries, local historians, and residents of the watershed. Approximately 20 percent of the surveys were returned.

An important aspect of this project was the personal contact with the stakeholders to supplement the information that was provided with the forms. DEQ visited all the industries along the Virginia sections of the stream bank to obtain information and prompt completion of the forms. The personal visits to these industries was very labor intensive but was critical to obtaining the information about the PCB use in the watershed. Figure 2 indicates the sites of concern from EPA's EnviroMapper program.

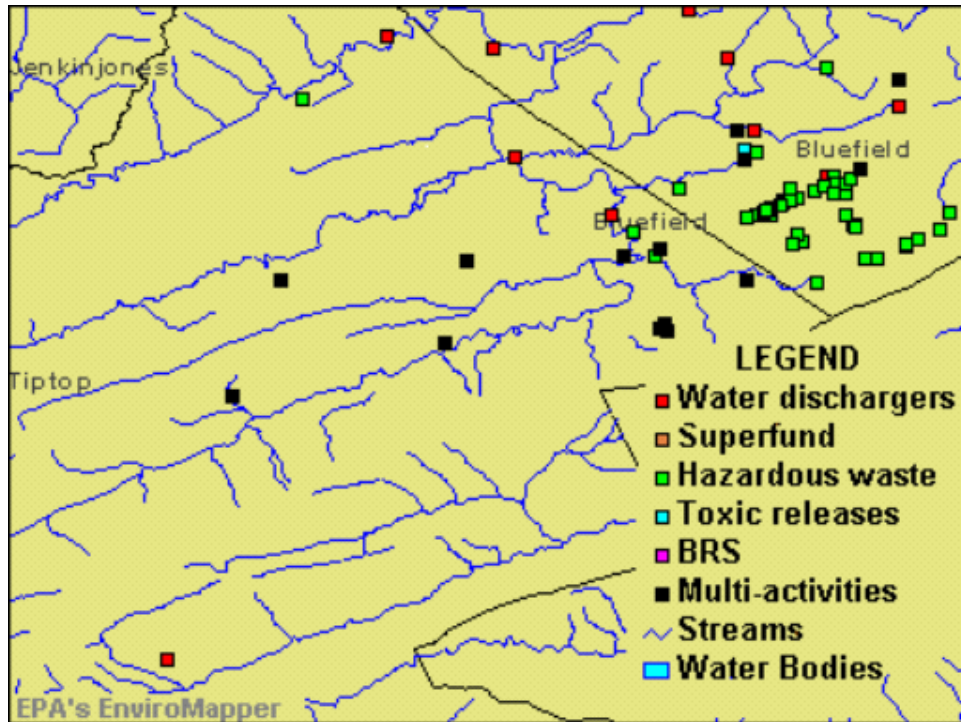


Figure 2. Bluestone River Watershed PCB sites of concern (USEPA EnviroMapper).

5) Research Underground Water Movement

The fifth task was researching the Karst influences on PCB transport in the Bluestone River watershed, including both surface and subsurface water flows. Understanding the movement of surface water as it enters sinkholes and caves and knowing where the water re-emerges on the surface became important since some of the potential and known historical PCB use sites are upgradient of sinkholes where surface flow is transported underground. It is well known that surface drainage re-emerges into some of the larger springs in the watershed: Beaverpond Resurgence Spring, Big Spring, Little Spring, and various other springs. On this portion of the study, DEQ conferred with the following geological experts for their resources and understandings of water transport: VaDMME, VDOT, WVDOT, and WVDEP mining hydrologists.

One of the sites of concern, called Hart Electric Corporation facilities, had surficial flows that drained into a sinkhole on adjacent property. The Hart Electric Corporation was known to have spilled PCB oil, buried drums of used PCB oil, and baked out the transformer coils in their facility. When it became a CERCLIS - Superfund cleanup site in the 1980s, contamination was extensive. The structure was removed and some soil was excavated and replaced. The sinkhole that received flows from this property, was dye tested to determine the flow retention and discharge location(s). The dye emerged within several hours in Beaverpond Creek. Researching the underground flows from Hart Electric was important since it was feared that PCB contamination had been transported by this underground stream to Dill Spring, which is a source of potable water for the BVWTP. During the survey of the hydrology it was discussed that the drainage from West Virginia probably does not extend down to Dill Spring because of a rise in the underground geology south of the Beaverpond Resurgence. DEQ performed sediment

sampling at Dill Spring and the results confirmed that only extremely low PCB concentrations were present.

Another area of concern was the older industrialized section along the railyards of Bluefield WV. This industrial section includes several previously investigated PCB sites. An interesting aspect of water movement is that both surficial and storm system drainage directs water into Beaverpond Creek, a tributary to the Bluestone. However, the Karst formation directly underground of the railyard, appears to drain into an adjacent watershed, the East River. PCB contamination from this industrial area or seepage from the old storm drainage system could migrate to either the Bluestone or East Rivers.

Third and fourth areas where underground water movement is important to potential PCB transport are located in the Abbs Valley and Town of Pocahontas Valley. Neither of these areas was originally considered a potential contributor to the PCB issue in VA. However, the geological research and responses to the surveys revealed underground transport to VA waters. Pollutants can be transported via these underground pathways from the deep mines of the Pocahontas formation into the Bluestone watershed. The mining operations used many pieces of equipment that contained PCBs including transformers and heavy mining hydraulic equipment. It is widely reported that mining operations disposed of spent fluids in the mines. A common routine maintenance practice for hydraulic mining machinery was to discharge these fluids from the equipment onto the mine floor. The deep mine drainage is to Laurel Fork, a tributary to the Bluestone downstream in West Virginia. Although Laurel Fork was not included in the original DEQ impaired stream segment, this information provided the basis for an expanded study in this watershed by EPA.

A second 'Big Spring' in NeMours West Virginia, drains Abb's Valley, where survey responses indicated that a home operation baked out used transformer coils and reclaimed the copper. This type operation typically generates dioxin and PCB soil and water contamination. This site is being evaluated by both EPA and the VDH since there is a potable water intake for the Town of Pocahontas within one half mile downstream.

6) Coordination with Interstate and Federal Agencies for Sediment Sampling

Additional sediment and soil sampling was performed at approximately 44 sites in the watershed after PCB survey form responses were reviewed. DEQ worked with WVDEP and USEPA to design the sampling plan and EPA contracted the actual sampling and analysis. Results indicated that one site previously cleaned up by EPA has significant contamination remaining in storm drains that flow toward the Bluestone River. This contamination may have been contributed by the industrial operation that occupied the site following the original EPA cleanup. Just downstream of the discharge of the contaminated storm drain, PCBs were not detected, providing confusing results.

Another site, which was identified through the survey responses, had both PCB and dioxin contamination around EPA response levels. Except for these sites, the sediment in the Bluestone River and at several other sites measured below detection limits, around 50 to 80 ppb of PCBs, when they were tested using the Arochlor method. One potential problem with using the PCB Arochlor method is that weathered samples yield variable results. If the ratios of the primary

quantification/identification congeners were significantly different than the original Arochlor ratios, then the sample may indeed have total PCB congener concentration above the TSCA cleanup concentration, yet not be identified as having PCBs.

7) Semi-Permeable Membrane Device Study

DEQ wished to determine the PCB loading by the various tributaries in the watershed in order to determine the total maximum daily loads (TMDLs) for the watershed. Semi-Permeable Membrane Device (SPMDs) were selected as the appropriate sampling device. The study design includes deployment of SPMDs at two times during the year, at seasonal high and low streamflows. The SPMDs are called 'virtual fish' for their ability to uptake chemicals such as PCBs and accumulate them with measurable kinetics that can be used to calculate the levels of contamination in a water body. The SPMDs uptake only the dissolved fraction which is the readily bioavailable form.

In this study, SPMDs were deployed during the high seasonal flow period in March 2003. The deployment for the low seasonal flow period is scheduled for November or December, 2004. Sampling locations were chosen to isolate loadings from different sources in the watershed, and met depth and mixing requirements for the devices.

In the Winter, 2004, eight locations were selected for SPMD deployment. Six of the original eight devices survived the observation term and were sent to the laboratory for analysis. After exterior cleaning of the SPMD tapes, the removed surficial algae and debris, were recombined with the other fractions for cleanup and extraction. The results were then background corrected and all QC samples were evaluated for loss of the deuterated compounds and presence of PCBs. All the quality control checks met specifications and were background corrected for contamination, then the sample results were deemed valid. The samples were analyzed by the congener high resolution GC/MS method according to USGS CERC and then summed for total PCBs. The corrected results were in units of nanograms of PCBs per SPMD tube. The USGS uptake rate chemical kinetic studies were used to convert the results into the approximate water column concentration of PCBs. The water column results were reported in units of picograms per liter.

Sample deployment locations were selected to represent various areas within the watershed. The first location selected was the public water supply for the Town of Bluefield, VA to help answer the health risk exposure concerns. Survey responses indicated several potential sources above the drinking water intake, if the WTP had been contaminated. The second site, Camp Joy, was selected upstream of the filtration plant uptake for two reasons: 1) A more upstream sample above the WTP would either provide a clean reference site and eliminate some potential sources above the device, or 2) help pinpoint the contamination source. The third site was located in Wright's Valley Creek, just above the confluence with the Bluestone. A foundry located on the banks of Wright's Valley Creek was reported to historically take metal parts for melting that might have contained or been coated with PCB oils. Another facility nearby, repaired mining machinery and appeared to have large quantities of oil in the ground at multiple locations. The fourth sampling site was in Beaverpond Creek, just above the confluence with the Bluestone River. The Beaverpond Creek location was critical for various reasons. Three known EPA Superfund cleanup sites, two PCB and one dioxin, were located on the drainage; as well as

stakeholders reports of additional suspicious PCB activity sites. The fifth site was the sewage treatment plant. Sludge from the BVWTP was flushed down to the STP. This could dramatically increase the amount of PCBs this site could have if the BVWTP were effectively removing PCB laden particulates from river water. Additionally, there were reports of various industrial practices, such as pouring barrels of PCB contaminated oils down the storm and sewer drains. Some sewer pipes might continue to release PCBs dumped years ago from their contaminated linings. The sixth sampling site was selected upstream of Brush Fork confluence with the Bluestone River. Brush Fork is just above a second Virginia Sewage Treatment Plant, STP, and this STP by this tributary and Mud Fork. Contamination from various transformer maintenance shops which operated historically up this valley in West Virginia would be trapped in the SPMD located in Brush Fork. Mud Fork, the seventh sampling site, provided a drinking water source to much of Bluewell, WV, however it doesn't have industrial development or much reported reputation for contamination. The eighth sample location was along the mainstem of the Bluestone River just before the river crossed into West Virginia at Yards.

8) Evaluation of Results

Two of the samples deployed in April of 2004, Camp Joy and Mud Fork, had to be redeployed because they were either removed from the stream intentionally during stream cleaning, or washed partially up onto the bank during storm events. The quality control checks were all well within specifications and all field and trip blanks were clean. The remaining sample results indicate the highest contamination level at Beaverpond Creek. This site had a water column concentration 3,700 pg/L, more than two times the surface water standard for PCBs, 1,700 pg/L (Figure 3). The second level of contamination was 1,200 to 1,300 for sites three, six, and eight. The next level of contamination was 640 pg/L measured at the STP. The lowest level of contamination was measured at the BVWTP at 230 pg/L.

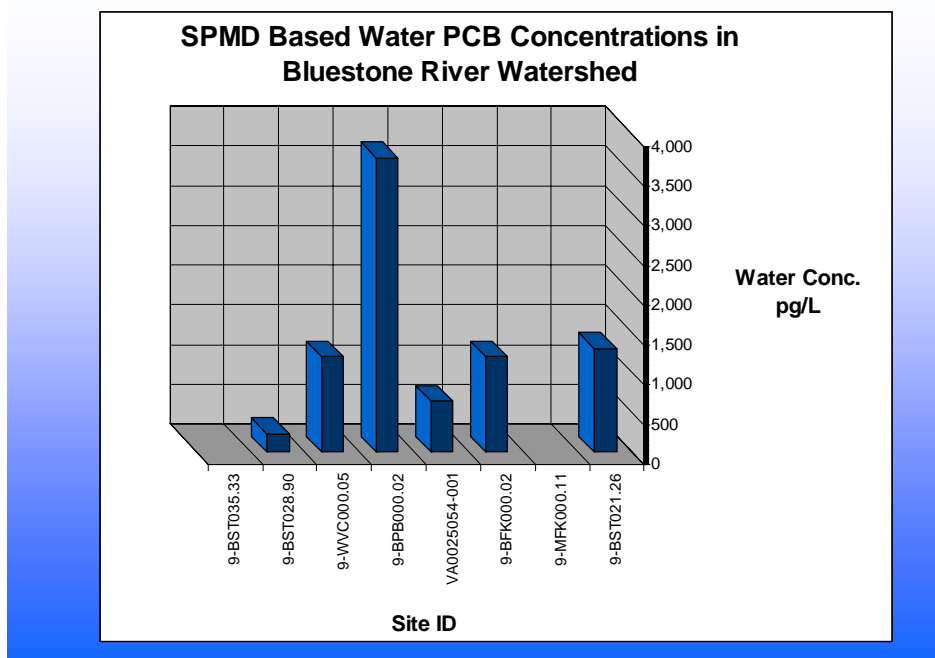


Figure 3. SPMD based water PCB concentrations in Bluestone River Watershed.

After the PCB congener results were tabulated, the data were plotted in elution order format using a line graph and compared against known unweathered manufacturer specifications for the original arochlor mixtures. The Ballschmitter system was used to assign congener identification and to help develop plots. Each result was evaluated and determined to be mostly one Arochlor or combinations of several different Arochlor mixtures. Using flow results the loading from the different tributaries were estimated from the high flow period and will be used to help develop a Total Maximum Daily Loading for PCBs in this watershed (Figure 4).

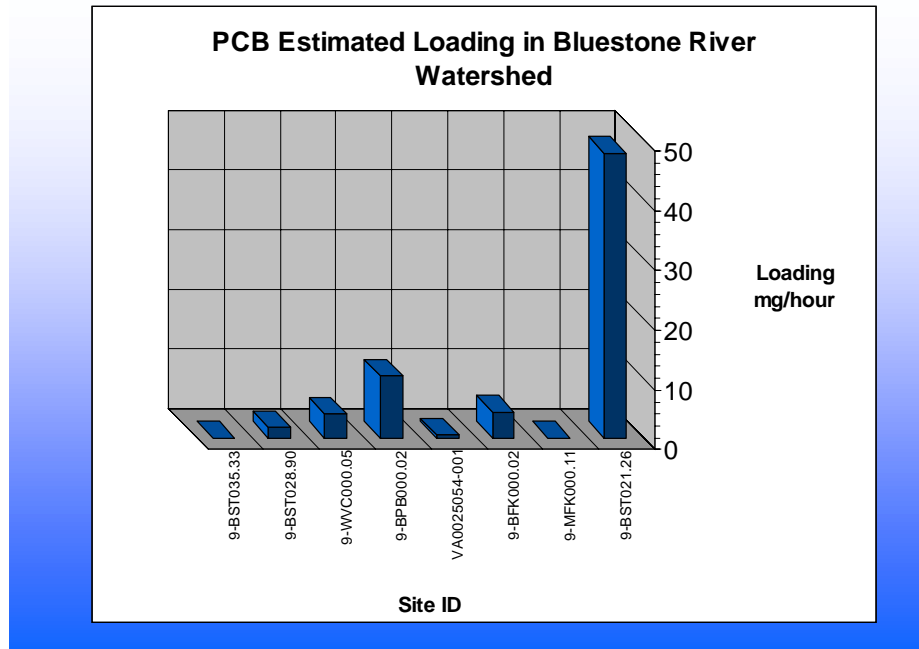


Figure 4. SPMD based PCB loading rates in Bluestone River Watershed.

SPMDs have several advantages over fish or other biological tissue analysis. They remain in one location (hopefully) during a specific sampling period. They remain for 20 to 30 days, providing an average of conditions that fish might have been exposed to in that water body during the same period. Not only does the SPMD provide a time averaged uptake that mimics the fish exposures, but it allows determination to some degree, of the particular Arochlor or mixture of Arochlors. Various mixtures of Arochlors might be present in the region of contamination, and SPMD results demonstrate better accuracy than previous methods of analysis, which may involve mobile targets such as fish or sediments. Interpretation of the Arochlor ratios depends upon what occurred to the oil mixture during metabolism and weathering. One of the problems with analyzing fish is that not only does the fish metabolize the PCB but the ratios of the congeners may change as some of the compounds become dechlorinated. Additionally, the fish contain more interfering compounds than an algae covered polyethylene tube. The connection to one location allows the contaminants to be correlated to flow and temperature conditions, which vary within a watershed and affect the chemical kinetics of uptake and loss across the membrane. Extensive kinetics studies have been done by various research organizations on most of the PCB congeners to compare the effects of temperature, flow, and other environmental conditions upon uptake.

Disadvantages of the devices include that they may be lost or damaged by powerful flows or storm events, and that converting SPMD results into 'calculated total water concentrations' depends upon scientific interpretation of theoretical partitioning. These equations estimate the equilibrium between the dissolved and particulate fractions of a chemical. The calculations provide results, based upon previous research into real world partitioning situations.

Lateral mixing depends upon the flow regime of each location and may not necessarily have been optimal during lower flow conditions. Although sites were carefully selected, lateral cross-sections of PCB concentrations in the water column were not assessed and may introduce variability into the results. Water column loading values from this study are considered estimates.

FUTURE EFFORTS AND GOALS

The original study design included sampling in lower flow conditions. Possible relocation of some SPMDs is being considered based on the contamination levels of the first round of high flow conditions. For example, if Camp Joy and Mud Fork have low concentrations, the devices could be moved to suspect hot spots on Beaverpond Creek drainage. Additionally, the Sewage Treatment Plant may not be sampled in the low flow conditions. Currently, USEPA, Virginia Department of Health, West Virginia Department of Environmental Protection, and West Virginia Department of Health and Human Resources are considering options of contributing SPMDs to further enhance the study of the watershed, simultaneous to the DEQ efforts.

The redeployment of SPMDs during low flow conditions will include one additional sample for air analysis. Historically, transport mechanisms in some watersheds include up to 90 percent of PCB transport through air.

The exercise of investigating PCBs in the Bluestone River watershed has thus far resulted in the discovery of two contaminated sites currently being evaluated by EPA for cleanup prioritization. Additionally, another site has been reported and will be sampled in the near future. The public water supply for Pocahontas, VA is being studied for potential health risks. One broken sewer line was discovered, reported, and repaired. Several barrels of oil were removed from the stream, after being discovered during SPMD deployment. The barrels may have been washed downstream during recent floods. Perhaps most importantly, questions of drinking water quality at the Town of Bluefield, Virginia, have been answered with several different sets of low-level analyses including the SPMDs. Follow up to help better pinpoint the locations of the contamination in the watershed may reveal additional cleanup sites or provide other alternatives to improving the quality of water, sediment, fish, and recreational activities in the Bluestone River Watershed.

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THE OCCURRENCE OF 17 β -ESTRADIOL IN SOIL AND SURFACE WATERS IN VIRGINIA

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KEYWORDS: 17 β -estradiol, agricultural watershed, surface waters, transport

ABSTRACT

The estrogen compound 17 β -estradiol (E2) is an identified endocrine-disruptor in aquatic organisms, even at relatively low concentrations. Its occurrence in surface waters is associated with sewage contamination and animal agriculture. Limited knowledge exists regarding its transport in the subsurface, particularly in agricultural watersheds. Soil water was collected from pan lysimeters in November 2003, and April and June 2004 at a 1.2-km² subcatchment in central Virginia, where poultry manure was applied in May 2003 only. The lysimeters, installed in a previous study, were located at depths of 15, 46 and 91 cm in a cornfield (upgradient in the watershed) and a cattle pasture (downgradient in the watershed). A synoptic survey of 34 streams in the Delmarva Peninsula, northern and central Virginia, and the Eastern Shore of Virginia was completed in June 2004 to understand the magnitude of E2 presence in the region's surface waters. The survey samples were collected from areas impacted by agricultural practices or urban development. All samples were analyzed using an enzyme-linked immunosorbent assay (ELISA). The November 2003 concentrations of E2 in soil water ranged from 0.03 to 0.45 ng/mL, where the highest levels were in the shallow soil and decreased with depth at both sites. The April 2004 results showed decreased, but existing E2 levels of 0.08 to 0.14 ng/mL. The June 2004 E2 concentrations were further reduced, but the highest concentrations were observed at the greatest depth. These results indicate that E2 persists in the subsurface even many months after introduction at the land surface and migration to soil depth may take place over the course of at least one year. Regional E2 concentrations in surface waters from the synoptic survey reached up to 0.10 ng/mL. Consistent elevated E2 levels were observed in the Delmarva Peninsula samples, demonstrating E2 occurrence is related to surrounding land use, particularly that of animal agriculture.

INTRODUCTION

The economic success of the poultry industry has raised an increasing environmental concern over its resulting waste production and management practices. In 1997, over 14 million tons of solid manure were produced by chickens alone. Presently, the most cost-effective use of poultry manure is as an agricultural fertilizer, due to its low moisture and high nutrient content. Use of poultry litter improves soil fertility and improves water retention (Sims and Wolf 1994, U.S. Senate Committee 1997, Gupta and Charles 1999). However, in addition to containing high

concentrations of nitrogen, phosphorous and bacteria, poultry litter consists of pharmaceuticals, including antibiotics and steroids.

The occurrence of natural estrogen compounds—namely, estrone (E1), 17 β -estradiol (E2) and estriol (E3)—in the environment is also due to the fecal material of mammalian and avian species. The practice of using these compounds as growth hormones in animal production further intensifies the environmental impact of the manure. These reproductive hormones, of which E2 is the most potent (Nichols *et al.* 1997, Ying *et al.* 2002), can have significant health effects, even at relatively low concentrations. Health effects linked to estrogen exposure include loss of fertility in general or disruption of reproductive organ development and shift in gender ratio in subsequent generations in particular (McLachlan and Arnold 1996, Harrison *et al.* 1997, Kramer *et al.* 1998, Routledge *et al.* 1998, Zarogian *et al.* 2001). Overexposure to these compounds has been associated with lowered sperm counts and increased risk for breast cancer in humans and linked to a number of dramatic changes in wildlife populations.

In their 1997 study, Nichols *et al.* demonstrated that considerable levels of E2 occurred in runoff from agricultural fields fertilized with poultry manure. Studies conducted in the Midwest United States reported significant concentrations in surface waters influenced by runoff from pig and poultry farms. Although much information is available regarding the identification and health risks presented by E2, little is known on its fate and transport in agricultural watersheds; in particular its environmental persistence and the natural processes that may mitigate that persistence.

Our study included a survey of the occurrence of E2 in the surface waters of the Delmarva region, including northern and central Virginia. In addition, we conducted a 13-month study of the presence of E2 in the subsurface of an agricultural watershed, in order to understand the extent of E2 persistence over time.

METHODS

Our study site is the Muddy Creek watershed located in the Shenandoah Valley, approximately 20 km northwest of Harrisonburg, VA (Figure 1). The Shenandoah Valley is one of two crop-intensive regions in Virginia where the cattle and poultry production both contribute to the application of manure to fields. The 1.2-km² subcatchment is located in the northwest portion of the main basin and is drained by the Muddy Creek Tributary. The land is partially forested and is predominantly used for agricultural purposes, such as corn, turkey and cattle production. The underlying soil is a deep and well-drained silt loam, formed as a residuum from extensively weathered limestone. The average depth to groundwater ranges from 0.5 m near a low-lying stream to 13 m at higher elevations in the watershed. Liquid cattle manure and dry poultry compost are applied to the cornfields and pastures every year in mid- to late April as fertilizer and as a method of waste disposal (Hyer *et al.* 2001).

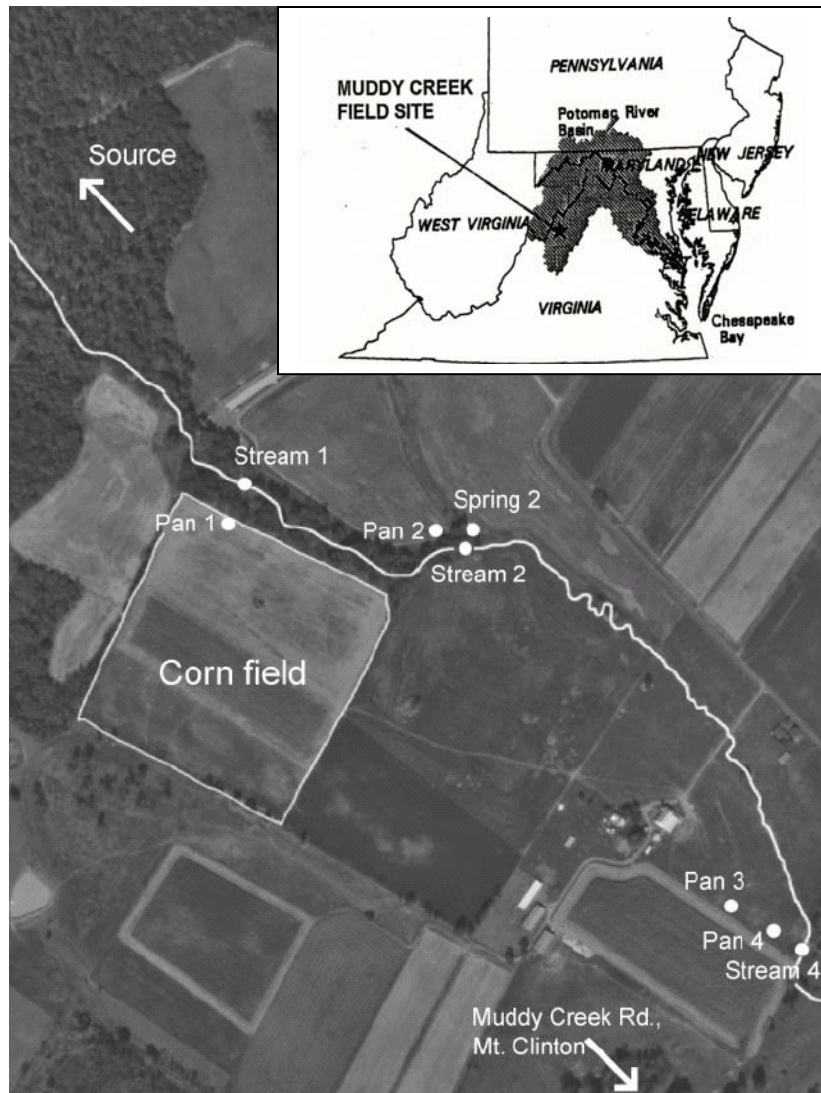


Figure 1. Muddy Creek watershed.

Subsequent to manure application to the field site in May 2003, soil-water sampling took place in November 2003, and in April and June 2004. The samples were collected from pan lysimeters installed in a previous study (Kauffman 1998). The lysimeters were installed at three depths: 15, 46 and 91 cm at two locations. Site 1 is on the edge of the cornfield and is upgradient in the watershed. The cornfield receives composted poultry litter application annually prior to planting. Site 4 is in a cattle pasture and is downgradient in the watershed. Liquid cattle manure is applied to the cattle pasture sporadically throughout the year.

Surface water sampling for the synoptic survey was completed in June 2004. The stream sites were within the Delmarva Peninsula, northern and central Virginia and the Eastern Shore of Virginia. Sites were chosen based on the surrounding land use, particularly those that would be influenced by urban development and animal agriculture.

All samples and duplicates were filtered (0.45 μm) in the field prior to preservation and were collected in acid-washed amber glass vials. The samples were analyzed for E2 using an enzyme-linked immunosorbent assay (ELISA).

RESULTS AND DISCUSSION

The November 2003 E2 concentrations in soil water ranged from 0.03 to 0.45 ng/mL, with the highest concentrations in the shallow soils at both sites, decreasing systematically with increasing depth (Figure 2). The E2 concentrations from the April 2004 sampling event also showed reduced but detectable concentrations of 0.08 to 0.14 ng/mL. Similarly, the June 2004 results showed a further decline of E2 levels, with the maximum concentrations observed at the greatest depth at both agricultural watershed sites.

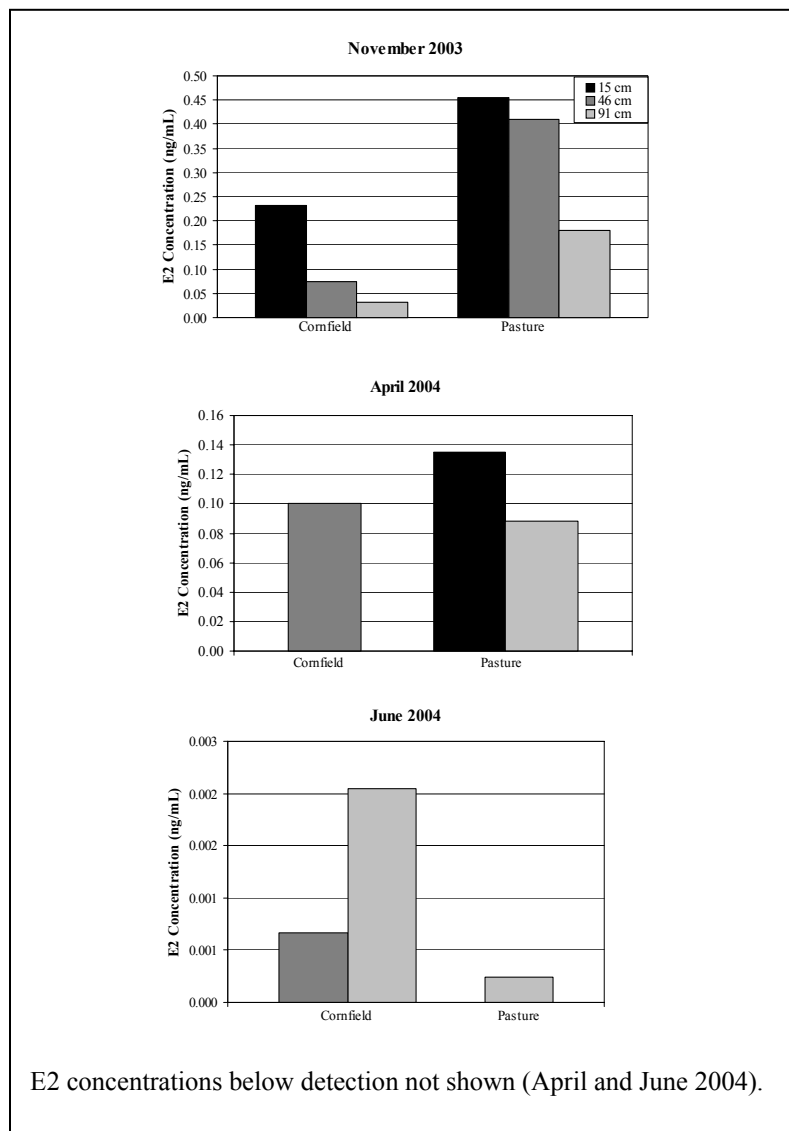


Figure 2. E2 concentrations in soil water from November 2003, April and June 2004 agricultural watershed sampling events.

The synoptic survey of surface waters from locations around Virginia revealed E2 concentrations reaching up to 0.10 ng/mL (Table 1). The higher E2 levels were predominantly found in the Delmarva Peninsula, where the maximum concentration was 0.07 ng/mL.

Table 1. Synoptic survey sampling locations and E2 concentrations.

State / Site Name	E2 Concentration (ng/mL)
DE / Herring Run Branch	0.07
DE / Butler Mill Branch	0.003
DE / Unity Branch	0.02
DE / Philips Branch	0.01
MD / Foulkner Branch	0.07
MD / Burnt Mill Branch	0.01
MD / Andrews Branch	0.02
VA / James River	0.10
VA / Appomattox River	0.01
VA / Pamunkey River	Below detection
VA / Rappahannock River	Below detection
VA / Cobb Mill Creek	0.03
VA / Nickawampus Creek	Below detection
VA / Katy Young Branch	Below detection
VA / Assawoman Creek	Below detection
VA / Taylor Creek	0.003
VA / Blacks Run	0.007
VA / Christians Creek	0.008
VA / Muddy Creek Stream 1	Below detection
VA / Muddy Creek Stream 2	Below detection
VA / Muddy Creek Stream 4	Below detection
VA / Muddy Creek Stream 5	Below detection
VA / Mt. Clinton Gaging Station	0.007
VA / Shenandoah River South Fork	0.006
VA / Biscuit Run	0.007
VA / Moorman's Creek	0.01
VA / North Fork Rivanna	0.01
VA / South Fork Rivanna	Below detection
VA / Rivanna River	0.004
VA / Moore's Creek	0.01
VA / Rivanna River (at Riverview)	Below detection
VA / Potomac River	0.002
VA / Great Falls	0.03
VA / Accotink Creek	0.04

The time-course results from soil water collected in an agricultural field indicate that E2 persists in the subsurface even many months after introduction at the land surface with applied manure. This conclusion is supported by our results of a previous study of E2 persistence over a single summer (2001) immediately following manure application in this watershed, where the highest

concentrations in the samples collected were in June but progressively decreased until the last sample in August. The rate of E2 loss motivated laboratory biodegradation experiments that resulted in rates more than adequate to completely consume E2 at the levels seen in soil water (Herman & Mills 2003). In addition, the results of the synoptic survey in June 2004 around Virginia indicate that the presence of E2 in surface waters is influenced by the surrounding land use, especially in areas of intensive animal agriculture. Among the areas sampled for the survey, the Delmarva Peninsula is the most productive in animal agriculture, particularly in poultry farming.

The behavior of E2 in the subsurface is still unknown, especially its interactions with other components in the natural environment through complexation, sorption, or abiotic transformation due to ageing. Further work will be conducted in an attempt to answer these questions.

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ORGANOARSENIC FATE AND TRANSPORT IN AGRICULTURAL WATERSHEDS

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KEY WORDS: arsenic, animal feed additives, water quality

ABSTRACT

The use of the organoarsenical roxarsone, added to poultry feed to increase weight gain, results in elevated arsenic concentrations in poultry litter. This litter is used extensively as fertilizer in agricultural regions. We are conducting field and laboratory studies to examine the fate and transport of roxarsone and its degradation products in an agricultural watershed in the Shenandoah Valley of Virginia, an area of intense poultry production. Results from previous lab experiments have shown that although roxarsone exhibits weak adsorption to Ap soils, it is rapidly biotransformed to As(V) in this soil horizon. Both roxarsone and its degradation product, As(V), adsorb strongly to Bt soils, suggesting that the Bt horizon may act as a sink for arsenic species. However, soil water data collected from lysimeters at the field site suggest that As(V) is mobile in Bt soil water. Current field studies, including electrical resistivity surveys and tracer dilution gauging, are focused on delineating potential pathways for litter-derived arsenic from fields to streams and groundwater supplies. Results of these combined studies will yield new and environmentally-relevant information on arsenic cycling within poultry-dominated agricultural watersheds.

INTRODUCTION

The organoarsenical roxarsone (3-nitro-4-hydroxyphenylarsonic acid), added to poultry feed at a concentration of 20.6-41.3 mg/kg, is used for growth stimulation, improved feed conversion, better feathering, increased egg production, and pigmentation (Anderson 1983). During the 42-day growth period, each broiler excretes about 150 mg of roxarsone (Anderson and Chamblee 2001), resulting in a total arsenic concentration of 10 to 50 mg/kg in poultry litter (Garbarino *et al.* 2003). Poultry litter, a mixture of poultry manure and bedding material, has widespread use as a fertilizer. In the Shenandoah Valley of Virginia, for example, there are approximately 364,000 tons of poultry litter produced each year, with most of the litter being applied as fertilizer on cropland (Mullins 2000). Wershaw *et al.* (1999) estimate that approximately 10^6 kg/yr of roxarsone and its degradation products are added annually to the environment worldwide from the use of poultry litter as fertilizer.

Recent work in our lab has focused on quantifying adsorption and biotransformation parameters for roxarsone in Frederick series soils, the main soil type in the Shenandoah Valley (Brown *et al.* in press). The goal of this study is to use a combination of electrical resistivity and tracer

dilution methods, along with geochemical data, to delineate pathways for litter-derived arsenic from fields to ground and surface water.

FIELD STUDY AREA

The study site is located in the Muddy Creek subcatchment, approximately 20 km northwest of Harrisonburg in Rockingham Co. The site is contained within the Chesapeake Bay watershed, in which 98% of Virginia's concentrated poultry operations are located (VADEQ 1998). The subcatchment is underlain by the Lower Ordovician-Upper Cambrian Conococheague Limestone, consisting of interbedded limestone, dolostone, and sandstone. The soils of the Muddy Creek subcatchment are of the well-drained silt loam of the Frederick series (Hockman *et al.* 1982). The vadose zone at the site is highly variable, with thicknesses ranging from 0 to 20 m below surface. The depth to groundwater is also variable, and ranges from 2 to 20 m below surface (Hyer *et al.* 2001). The study area contains four small monitoring sites (Figure 1), which are instrumented with a total of seven monitoring wells, four drive point samplers, two stream gauging stations, and a tipping-bucket precipitation gauge (Figure 1). The data presented in this study were collected from the Upper and Lower Sites (shown as Upper and Lower Well Pair on Figure 1). Three zero-tension (pan) lysimeters, which collect up to 5 L of soil water at 15, 45, and 90 cm depths, are installed at each of the four monitoring sites. Details on lysimeter construction can be found in Kauffman (1998). The study area includes two farms, which contain cornfields, cattle pastures, and a poultry barn. Dry poultry manure is applied on the cornfields at the annual rate of about 4500 kg/ha, as a fertilizer and method of waste disposal (Hyer *et al.* 2001). A study of poultry production in the larger Muddy Creek watershed documented the close to 1 million animal units, which contribute approximately 22,000 tons of poultry litter to the watershed annually (Muddy Creek TMDL Establishment Workgroup 1999).

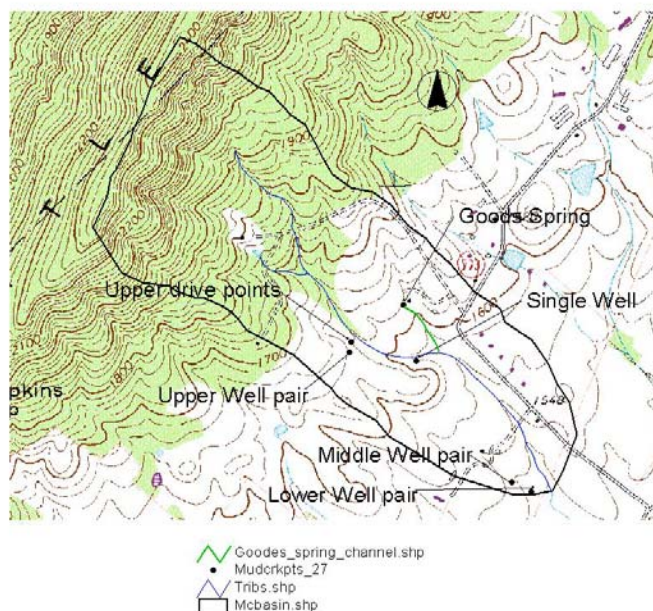


Figure 1. Topographic map showing locations of monitoring sites in the Muddy Creek sub-watershed. Courtesy of Doug Chambers USGS West Virginia. Thick solid line shows location of electrical resistivity profile shown in Figure 4. Poultry litter is applied to cornfields located west-southwest of the Upper Well pair.

METHODS

The zero-tension (pan) lysimeters at the Upper Site and Lower Site (Figure 2) were sampled in May 2003, after litter application. Samples were collected under low-flow (<10 ml/min) pumping conditions. Prior to sample collection, field measurements, including pH, temperature, specific conductance, and dissolved oxygen, were taken. Samples for chemical analysis were filtered (0.2 micron), preserved, and kept at 4°C until analysis. Samples for total arsenic analysis were preserved with HNO₃ and analyzed by GFAAS. Samples for arsenic speciation were preserved with EDTA and analyzed using HPLC separation combined with hydride generation (HG) and detection by ICP-AES, following the method of Garbarino *et al.* (2002). Samples were analyzed for DOC by combustion after preservation with HCl. Samples for Fe were preserved with HNO₃ and analyzed by ICP-AES. Samples for PO₄ were analyzed by ion chromatography.

Surface water discharge along a 100 m study reach was measured using tracer dilution gauging methods (Gordon 1992). The method we used involved continuous injection of rhodamine and NaCl at known concentrations and measurement of rhodamine concentrations and conductivity at 5 m intervals downstream from injection. Dilution gauging was conducted on August 10, 2004. Samples for rhodamine analysis for the dilution gauging were analyzed using a portable fluorimeter; conductivity was measured using a YSI Hydrolab sonde.

Electrical resistivity was used to evaluate subsurface geologic features, including the saprolite-bedrock interface, rock type, and degree of saturation, along a cross-section perpendicular to the stream (Figure 1). A Campus Geopulse earth resistivity meter connected to a multi-electrode cable with 25 electrodes was used to collect four two-dimensional resistivity profiles at the site. Dipole-dipole arrays were used with 10 m electrode spacing. This method measures a series of apparent resistivities that are converted to “true” resistivities using linear inversion modeling (Loke and Barker 1996).

RESULTS AND DISCUSSION

Results of soil water sampling show that arsenic is present in soil water in the underlying the litter-treated site, but not in the control site where litter is not applied (Figure 2). Although these samples were collected after litter application, when arsenic concentrations would tend to be highest, sampling in July, November, April reveals that arsenic is measured in soil water throughout the year. Note that arsenic is correlated with Fe, dissolved organic carbon (DOC), and phosphate (Figure 2). DOC and phosphate are litter-derived; Fe is likely complexed by the DOC, increasing its solubility in soil water. Speciation analyses conducted on soil water samples reveal that all of the arsenic is present as As(V), not roxarsone. This is significant as As(V) is considerably more toxic than roxarsone (Andreae 1983).

Measurement of discharge at specified intervals along the reach adjacent to the litter-treated field in August 2004 shows that the stream is gaining along the study reach (Figure 3). Although this demonstrates that groundwater is discharging to the stream, this method does not shed any light on the origin of the discharging groundwater. Tracer studies are being planned to delineate flow pathways from the upper site, where litter is being applied, to the stream.

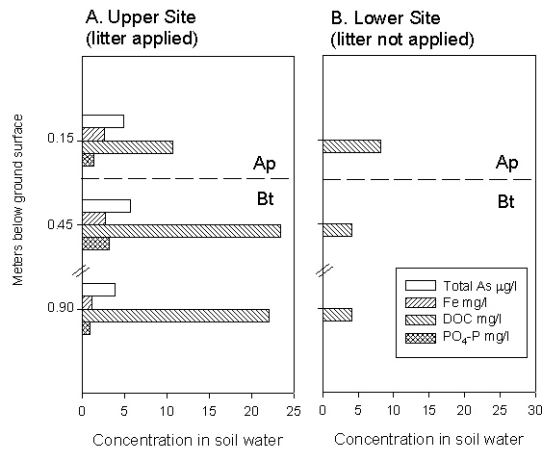


Figure 2. Concentrations of selected chemical parameters (As, Fe, DOC, PO₄) of soil water collected from lysimeters at Muddy Creek site in May 2003. The pH of soil water ranges from 5.5 to 6.0. A) Data from Upper Site, treated with poultry litter. B) Data from Lower Site, not treated with poultry litter. Note break in depth on y-axis. Delineation of Ap and Bt horizons is approximate.

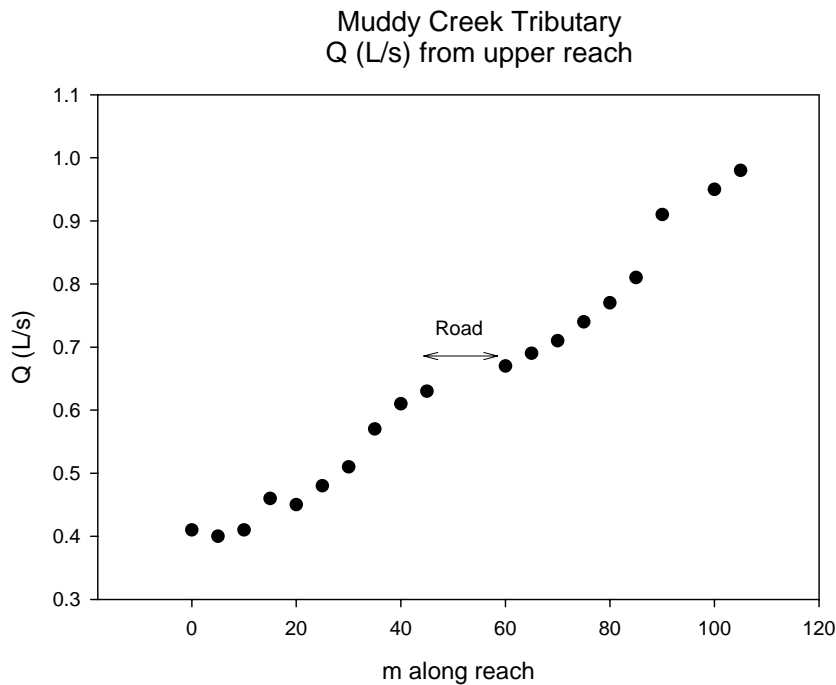


Figure 3. Discharge measured using a rhodamine tracer dilution method at 5 m intervals on Muddy Creek tributary. Starting point (0 m) is the location of the upper flume (see Figure 1). Discharge data collected by Brendan Brown on August 10, 2004.

Preliminary results from the electrical resistivity profiling reveals several important features. The profile shown in Figure 4 was conducted on August 10, 2004, perpendicular to the stream, to

delineate the nature of the soil/bedrock interface. The bedrock surface, shown by a sharp transition between blue (low) and brown (high) resistivity, is located close to 20 m below the stream, and appears to slope away from the hill where the Upper Site wells and lysimeters are located. The ER profile also shows a depression of the water table (transition between green and blue areas) along the hillslope between the upper site and the stream.

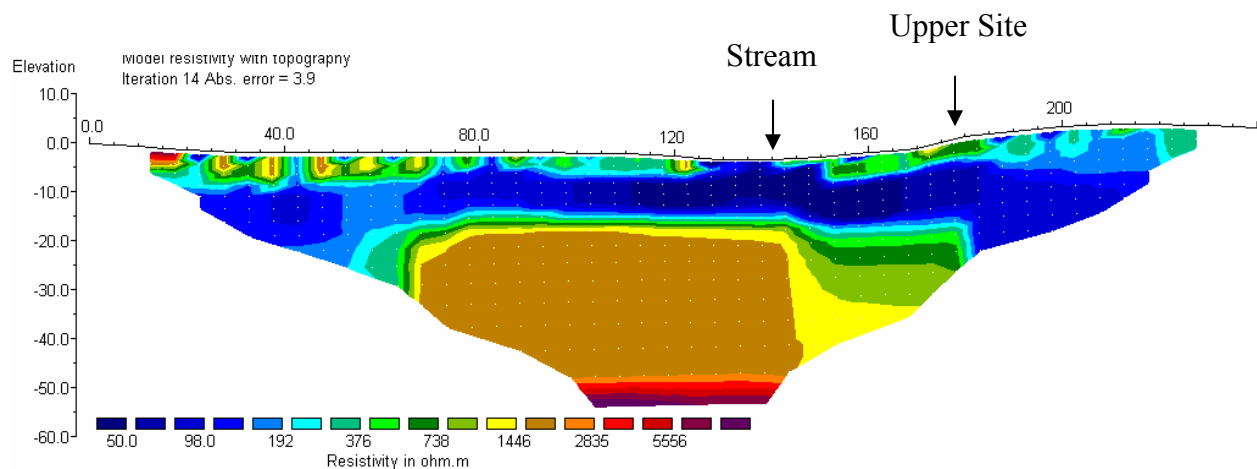


Figure 4. Electrical resistivity profile across stream (see location in Figure 1). Elevation is in meters. ER data collected by Benjamin Schwartz.

SUMMARY

A combination of stream gauging and electrical resistivity profiling is being utilized to delineate pathways for water flow and contaminant transport at an agricultural site where poultry litter has been applied as fertilizer. Soil water collected at the Upper Site, at the edge of a cornfield to which poultry litter has been applied, contains arsenic and other litter-derived solutes. Measurement of discharge in a stream down a hillslope from the Upper Site shows that the stream is gaining, but the origin of the discharging groundwater is unclear. Electrical resistivity data collected along a transect from the Upper Site and across the stream show a deep (20 m) bedrock surface and a complex water table surface. These combined data suggest that the flowpaths from the Upper Site, where poultry litter is applied, to the stream may be difficult to delineate without tracer experiments. Although the pathways have not been fully delineated, results show that the methods, in combination with water sampling and tracer experiments, will yield useful information on both flow and transport.

ACKNOWLEDGMENTS

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KINETICS OF SCORODITE DISSOLUTION AS A FUNCTION OF TEMPERATURE

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KEY WORDS: scorodite, arsenic, mineral weathering

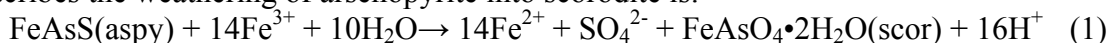
ABSTRACT

Scorodite ($\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$) is a common weathering product of arsenopyrite (FeAsS), one of the main arsenic-bearing minerals in the Earth's crust. Previous studies on scorodite have focused on the stability of scorodite (Dove and Rimstidt 1985, Zhu and Merkel 2001) and on methods for precipitating scorodite to remove arsenic from mine waste (Demopoulos *et al.* 1995). Neither of these studies investigated the dissolution kinetics, which are important for assessing the controls on arsenic release to and transport within natural waters. The purpose of this research was to examine the kinetics of scorodite dissolution as a function of temperature. The first set of experiments was designed to examine scorodite dissolution rates at temperatures representative of seasonal variation (20, 30, and 40° C). Dissolution experiments were conducted in batch reactors in a controlled temperature water bath and samples were collected over a four-hour period. The initial rate method was used to calculate the scorodite dissolution rate. Results of the temperature dependent study revealed that the dissolution rates increase as a function of temperature with rates ranging from $10^{-6.00}$ to $10^{-5.53}$ mol/(m²*sec).

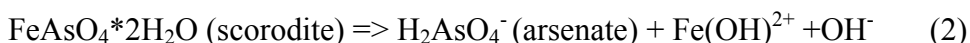
INTRODUCTION

Arsenic is a known carcinogen and toxin that is naturally occurring in areas worldwide. Arsenic is an important element because of the impact it has on human health. It causes damage to neurological and cardiovascular systems, and has also been linked to cancer (NRC 1999). Anthropogenic sources of arsenic include wood preservatives, glass production, pesticides, and animal feed additives (Welch *et al.* 2000). Natural sources of arsenic in groundwater include weathering of arsenic-bearing minerals, such as arsenopyrite, scorodite, realgar and orpiment. The release of arsenic from minerals is controlled by its dissolution rate, which is in turn controlled by mineral form, pH, redox conditions, and presence of bacteria (Welch *et al.* 2000).

Scorodite, $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$, is a stable alteration product of arsenic-rich sulfides (Welch *et al.* 2000). Scorodite has an equilibrium constant of $K_{sp} = 10^{-21.7 \pm 0.5}$ and is stable between pH ranges of 0.5 to 3 and E_h ranges of 0.4 to 1.0 mv (Dove and Rimstidt 1985). The overall reaction that describes the weathering of arsenopyrite into scorodite is:



Scorodite can dissolve congruently or incongruently. Congruent dissolution of scorodite can be described by the following reaction:



Dissolution experiments conducted by Dove and Rimstidt (1985) showed that scorodite can also dissolve incongruently via the following reaction:



Neither type of dissolution has been shown to consistently occur. In addition, little is known about the dissolution kinetics of scorodite. The objective of this research was to evaluate the kinetics of scorodite dissolution as a function of temperature. The rate of dissolution is important because it controls the release of arsenic from scorodite into the environment over time.

METHODS

Precipitation of Scorodite

Scorodite was precipitated using a modified method from Demopoulos *et al.* (1995). First, an iron-arsenic solution is made in a 500 mL volumetric flask by adding 3.647 g of ferric chloride and 4.163 g of sodium arsenate. To the solids, 53.25 mL of HCl is added, and the solution is brought to 500 mL with deionized water. This solution is then transferred to a 1500 mL beaker and heated to 95°C while being stirred. Next, 587.5 mL of 1N NaOH are added to the solution over a period of thirty minutes. The solution is stirred and kept at 95°C for five hours, and then filtered. The precipitate is washed by centrifuging 2 g of precipitate with 10 mL of DI water for 15 minutes. The supernatant is decanted and the process is repeated. The scorodite precipitate is then dried and sieved to a grain size of 106-150 μm. Scorodite precipitation was confirmed by x-ray diffraction (Figure 1). X-ray diffraction is a method that directs x-rays at the surface of a powder sample at a specific angle. The intensity in which the x-rays reflect off of the surface creates a distinct pattern for each individual mineral. The sample was also analyzed on the BET surface area analyzer, which yielded a specific surface area of 9.52 m²/g.

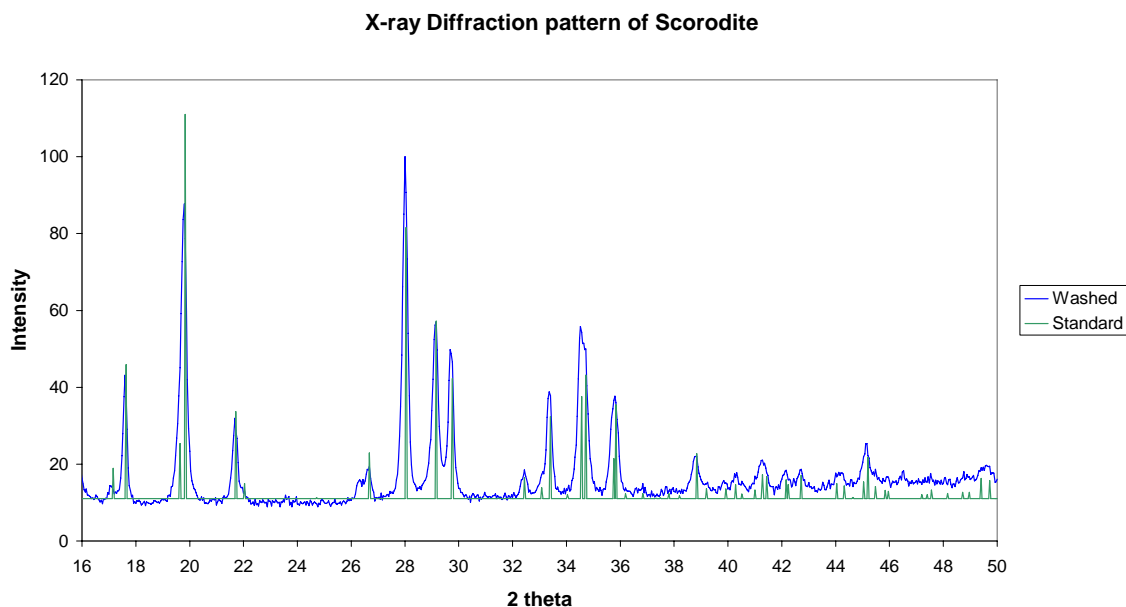


Figure 1. The x-ray diffraction pattern of scorodite.

Batch Reactor Design

The dissolution experiments were conducted using batch reactors (250 mL erlenmeyer flasks). These reactors contained 100 mL of solution made from 0.01 grams of washed scorodite and DI water. The reactors were placed into a shaker-water bath to keep the flask well mixed and at a constant temperature. Experiments were conducted at three temperatures: 20, 30, and 40°C. All experiments for each temperature were run in triplicate.

Analytical Methods

Approximately 2.75 mL of sample were collected from each reactor at predetermined times. A portion of these samples was filtered (0.45 micron) and preserved with nitric acid. The other portion was analyzed for pH. Samples were analyzed for arsenic using a graphite furnace atomic absorption spectroscopy (GFAAS). Samples were analyzed for iron using inductively-coupled plasma atomic emission spectroscopy (ICP-AES).

Determination of Rate Constants

The reaction progress variable (RPV) chosen to represent scorodite dissolution was arsenic, as it appeared to behave more conservatively than iron. The concentrations of arsenic were plotted against time and a linear equation was fit to each set of data as well to the average data. The initial rate method was used to determine the dissolution rate. The initial rate method involves measuring the rate of reaction at very short times. For this experiment, the rate was calculated over a period of ninety minutes.

RESULTS

The first experiment used to calculate scorodite dissolution was kept at a temperature of 20°C for ninety minutes. The arsenic concentrations are presented in Figure 2. The 20°C arsenic concentrations yielded a linear equation of $y = 0.015776x + 0.939679$. The change in concentration over time was $10^{-1.80}$ mol/min which yielded a dissolution rate of $10^{-6.00}$ mol/m²/sec.

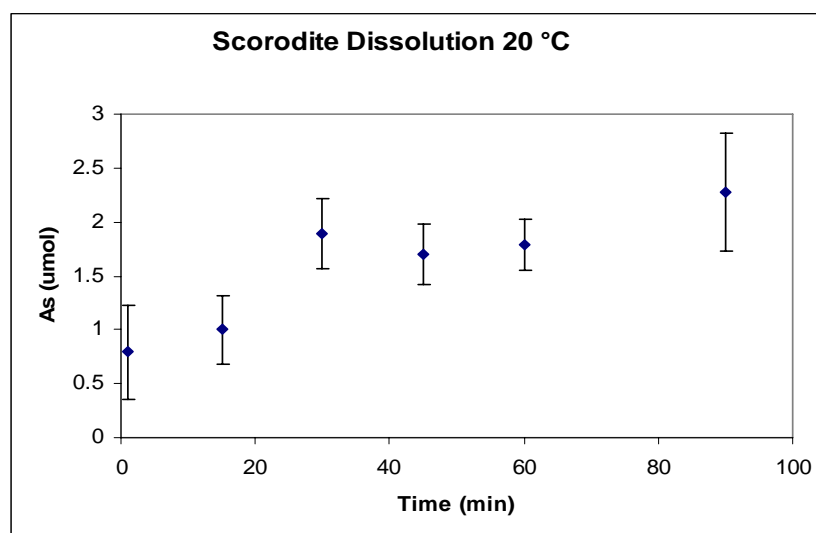


Figure 2. Average arsenic concentrations at 20°C. Error bars represent standard deviation from results of triplicate experiments.

The next experiment was conducted at 30°C (Figure 3). The arsenic concentrations of the 30°C experiments fit an equation of $y = 0.027871x + 1.162949$, thus having a change in concentration over time of $10^{-1.55}$ mol/min. The calculated dissolution rate from these data is $10^{-5.75}$ mol/m²/sec.

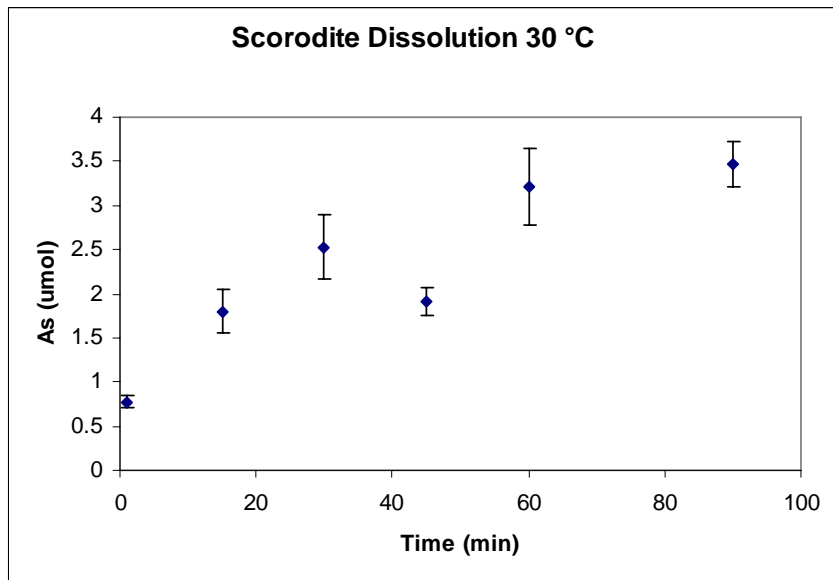


Figure 3. Average arsenic concentrations at 30°C. Error bars represent standard deviation from results of triplicate experiments.

The final experiment was conducted at 40°C (Figure 4). The arsenic concentrations vs. time of this experiment had a best fit equation of $y = 0.046845x + 2.490907$. The slope was $10^{-1.33}$ mol/min, resulting in a dissolution rate of $10^{-5.53}$ mol/m²/sec.

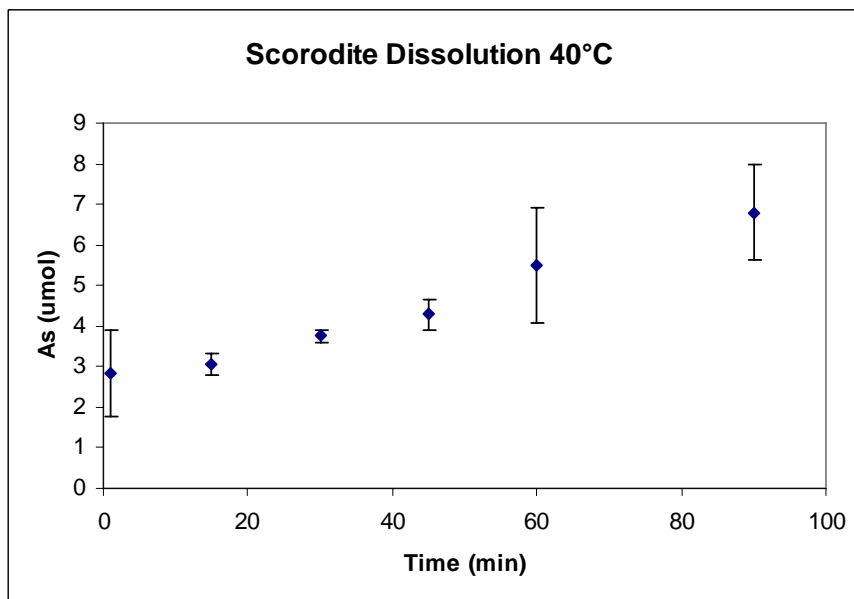


Figure 4. Average arsenic concentrations at 40°C. Error bars represent standard deviation from results of triplicate experiments.

Figure 5 presents the average arsenic concentration for each trial. It is evident from the data that the release of arsenic to solution increases at a faster rate over time at higher temperatures.

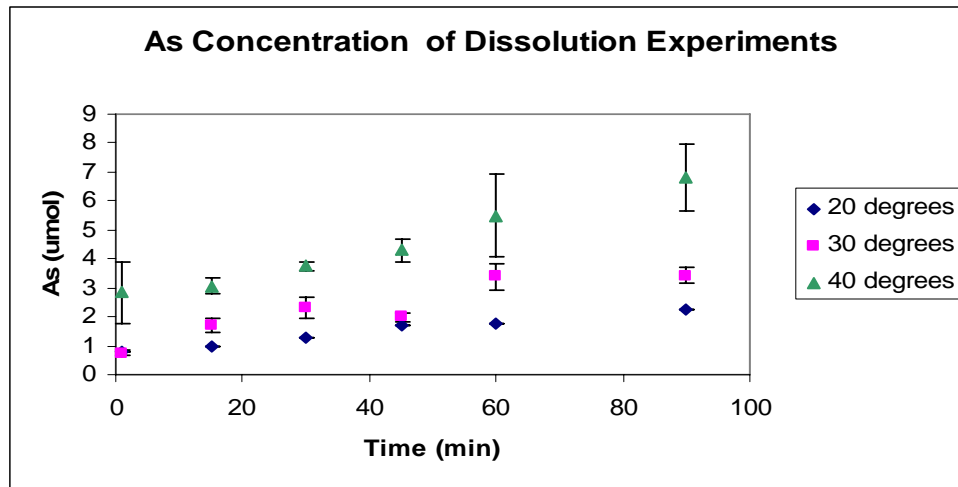


Figure 5. Graph of average arsenic concentrations at the three temperatures. Error bars represent standard deviation from results of triplicate experiments.

The dissolution rates for each individual trial as well as the averages are displayed in Table 1.

Table 1. Dissolution rates and pH conditions for each trial over a range of temperatures.

Temperature (°C)	Trial	Dissolution rate (mol/m ² /sec)	Initial pH	Final pH
20	AA	10 ^{-5.92}	4.70	4.48
	BB	10 ^{-6.26}	4.38	4.44
	CC	10 ^{-5.91}	4.33	4.41
	Average	10^{-6.00}	4.47	4.44
30	DD	10 ^{-5.73}	4.75	4.54
	EE	10 ^{-5.72}	4.91	4.45
	FF	10 ^{-5.80}	5.50	5.06
	Average	10^{-5.75}	5.05	4.68
40	H	10 ^{-5.57}	5.23	5.41
	I	10 ^{-5.59}	4.78	4.97
	J	10 ^{-5.53}	4.58	6.10
	Average	10^{-5.53}	4.86	5.49

Note: The average rate was calculated from the regression of the average concentrations of arsenic at each sampling time.

DISCUSSION

Dissolution Kinetics

The average dissolution rate of scorodite increased as a function of temperature (Figure 6). The 20°C experiments yielded an average dissolution rate of $10^{-6.00}$ mol/m²/sec. The 30°C experiments produced an average dissolution rate of $10^{-5.75}$ mol/m²/sec. The 40°C experiments generate an average dissolution rate of $10^{-5.53}$ mol/m²/sec. The dissolution rate increases 10^{-8} mol/m²/sec/°C.

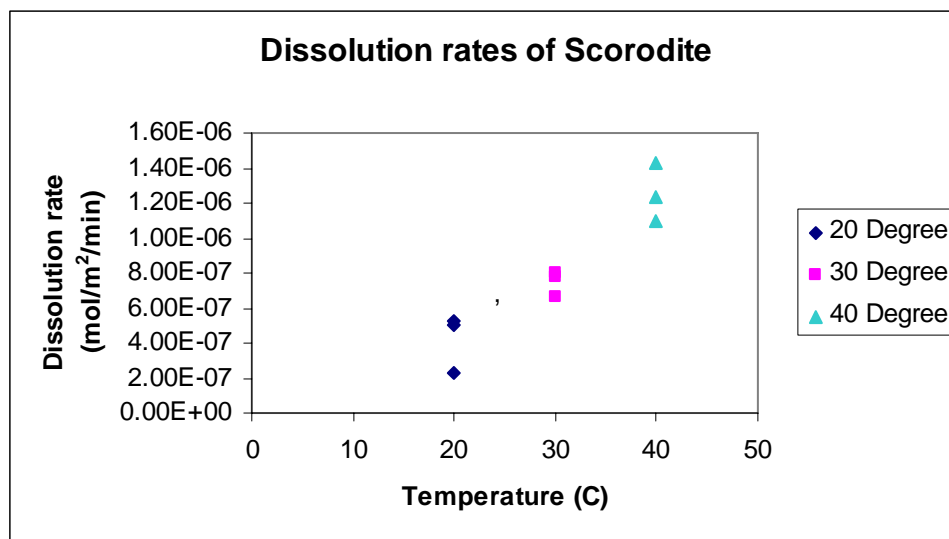


Figure 6. Scorodite dissolution rate as a function of temperature.

CONCLUSIONS

As temperature fluctuates in natural environments, it can affect the rate of mineral weathering. The data collected from this study show that there is a positive, linear correlation between the dissolution rate of scorodite and temperature. At room temperature, the scorodite dissolution rate is $10^{-5.84}$ mol/m²/sec. This rate is several orders of magnitude greater than the oxidation rates of other arsenic-bearing minerals, including realgar (Lengke and Tempel 2003), orpiment (Lengke and Tempel 2002), and arsenopyrite (Walker *et al.* 2004). These differences in rates indicate that where scorodite is present, its dissolution may play a significant role on arsenic release to natural waters.

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CHANGES IN SURFACE CONCENTRATIONS AND ASSOCIATIONS OF Pb(II) ON PLANAR γ -Al₂O₃ EMPLACED IN NATURAL SEDIMENTS

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KEY WORDS: oxide, metal, sorption, lead, sediments

ABSTRACT

The objective of this work is to develop planar oxides as a field deployable tool that can be used to quantitatively understand metal ion sorption behavior in natural sediments. Planar oxides are thin coatings approximately 20 nm in thickness prepared on an underlying metal substrate. Previous work in our lab has compared the reactivity of planar oxides to the traditionally used bulk oxides. Specifically, uptake of Pb(II) on the planar γ -Al₂O₃ was compared to uptake on bulk γ -Al₂O₃. Results have shown that Pb(II) uptake behavior was comparable on the two materials, and that the local atomic binding environment of Pb(II) on each material is similar. This work expands on these results by emplacing both unreacted and Pb(II) loaded planar γ -Al₂O₃ in natural sediments, allowing them to react, and characterizing the changes in metal sorption complexes on the alumina surface. Mesocosms using sediments collected from the Baltimore Harbor were used. Planar aluminas were prepared with Pb(II) coverages ranging of 0.0, 0.3 and 2.2 $\mu\text{mol Pb/m}^2$ γ -Al₂O₃. ToF-SIMS was used to identify the identities and quantities of elements on the γ -Al₂O₃ surface prior to emplacement in the sediments. The planars were allowed to react for 4 time periods: 1, 4, 7 and 14 days. Upon removal, the planars were characterized again to quantify changes in Pb(II) concentrations in oxic, anoxic and boundary layer sediments, and to identify other complexes having an effect on the resultant Pb(II) complex. Results from this study indicate the potential for using planar oxides as a field deployable tool for metal ion sorption studies. This, combined with the ability to use the same materials in controlled laboratory studies, will lead to a better understanding of factors controlling metal ion behavior in natural environments.

INTRODUCTION

It is well known that the fate and transport of heavy metals in aquatic systems is largely coupled to reactions that occur at the sediment-water interface. These reactions are highly dependent on the nature and abundance of reactive mineral phases. In oxic sediments, layers of Fe, Mn and Al, along with organic matter are thought to dominate metal ion surface complexation in natural

systems (Hem 1977, Warren and Zimmerman 1994, Bertsch and Seaman 1999, Dong *et al.* 2001). In anoxic sediments, the partitioning of metals is strongly influenced by the presence of sulfide phases (Simpson *et al.* 2004). Much of the knowledge regarding metal sorption behavior has been derived from experiments on pure phase model compounds. These studies have provided a large base of knowledge describing the boundary conditions of sorbate/mineral surface interactions and have been successful in identifying surface functional groups involved in complexation reactions (Bertsch and Seaman 1999). However, the heterogeneous nature of sediments as well as the complexity of the natural environment has made it difficult to interpret these results in relation to observations made at the field scale. The complexity of natural soils and sediments and the large spatial scales over which these processes occur makes understanding contaminant behavior in natural environments extremely challenging.

This work utilizes a novel approach to understanding metal ion sorption processes in natural sediments through the use of planar oxides. Planar oxides consist of thin oxide coatings approximately 20 nm in thickness prepared on an underlying metal substrate. They can be prepared through high-pressure high-temperature steam oxidation of a metal foil or through discrete particle attachment. Because these oxides are formed on a support, they are more similar to natural reactive phases which are often present as coatings on mineral surfaces. Also, because of their morphology, planar oxides can be used not only in laboratory studies, but emplaced directly into natural sediments where they can react *in situ*. The planars can then be retrieved and the surface complexes formed on them under natural conditions can be characterized and compared to surface complexes formed on identical materials under controlled laboratory conditions. Thus, the use of planar oxides is a step forward in the challenge of bridging the gap between laboratory and field studies.

The γ -Al₂O₃/Pb(II) system has been extensively studied by both spectroscopic and wet-chemical techniques (Chisholm-Brause *et al.* 1990 and references therein, Bargar *et al.* 1997). Previous work in our laboratory has compared Pb(II) uptake on planar γ -Al₂O₃ to that on bulk γ -Al₂O₃ under controlled conditions (Conrad *et al.* 2002). The results of these studies have shown that Pb(II) uptake on the planar oxides was similar to that seen on the bulk materials, thus connecting the planar system to the large body of knowledge describing Pb(II) uptake onto the bulk materials. The purpose of this study is to build upon those results by emplacing the planar γ -Al₂O₃ into natural sediments. The objective of this study is test the ability to use planar oxides as a tool to quantitatively describe metal ion sorption/desorption processes in natural sediments, thereby linking laboratory and field studies. Surface complexes formed on the planar γ -Al₂O₃ surface will be characterized to understand changes in Pb(II) complexation behavior as a function emplacement time and redox characteristics of the sediments. Environmentally relevant complexing agents such as Cl, Fe and S will also be measured, and their effect on Pb(II) concentrations will be assessed.

METHODS

Sediments used for this study were collected from Baltimore Harbor, Maryland. Sediments mixed and placed into two 25-gallon Nalgene containers resulting in mud beds approximately 12 cm in depth. Unfiltered water from the York River was pumped through the tanks at a rate of 3 L/hr (temperature=28°C, pH=7.45 ± 0.05, salinity=18.0 psu). Sediments were allowed to

equilibrate for a period of two weeks prior to the experiment. During this time, a visible oxic layer that was approximately 2-3 mm thick formed on the surface sediments in both tanks.

Planar $\gamma\text{-Al}_2\text{O}_3$ was prepared by a high-pressure high-temperature steam oxidation reaction. Briefly, high purity aluminum foil was exposed to steam, rapidly heated to 550°C and pressurized to 45 psi resulting in a $\gamma\text{-Al}_2\text{O}_3$ surface coating approximately 20 nm thick supported by the metal substrate. The initial conditions of the experiment consisted of three sets of planar $\gamma\text{-Al}_2\text{O}_3$. One set of planars was unreacted with Pb(II), with the remaining two sets loaded with 0.3 $\mu\text{mol Pb/m}^2$ (Low [Pb(II)]) and 2.2 $\mu\text{mol Pb/m}^2$ (High [Pb(II)]) (see Conrad *et al.* 2002 for details). The three conditions were chosen to represent the following scenarios: a clean particle, a particle with a low contaminant load and a particle with a higher contaminant load being deposited in the sediments. At lower sorption densities, metal surface complexes are often bound to surface sites with very high affinities making the metal more strongly attached to the surface. At higher sorption densities, these high affinity sites may be unavailable, causing metals to bind less strongly to the sediment surface. Each set of planar $\gamma\text{-Al}_2\text{O}_3$ was emplaced in the sediments with a notch cut into the bottom edge to indicate the orientation of the sample. The planars were approximately 6 cm in length by 1.5 cm in width. Approximately 2 cm of the planar was visible above the sediment surface. The oxic anoxic boundary occurred at 2-3 mm depth in the sediments. The remaining 4 cm of the planar was in anoxic sediments. Each set of planars was allowed to react for 4 time periods, 1, 4, 7 and 14 days and was run in duplicate. Each treatment will have an area that was exposed to oxic sediments, anoxic sediments and a boundary layer where the redox conditions of the sediments are rapidly changing.

Table 1. ToF-SIMS Pb/Al ratios of planar $\gamma\text{-Al}_2\text{O}_3$ with three Pb(II) loadings prior to emplacement.

Sample	Estimated Pb(II) Loading ($\mu\text{mol Pb/m}^2 \gamma\text{-Al}_2\text{O}_3$)	ToF-SIMS Pb/Al
Unreacted	0.0	0.000
Low [Pb]	0.3	0.023
High [Pb]	2.2	0.046

Changes in Pb(II) concentrations, as well as the presence of other complexing agents, on the planar surfaces under each condition were analyzed using a TFS-2100 TRIFT II Time of Flight Secondary Ion Mass Spectrometer. Samples were analyzed by rastering a 22 KeV Au^+ ion gun over an 80 μm x 80 μm area. Data were normalized to Al intensities to minimize instrumental variation. The Pb/Al ratios for the starting materials are listed in Table 1. Time of Flight Secondary Ion Mass Spectrometry (ToF-SIMS) analyzes only the first monolayers of a sample and provides detection limits on light element oxides better than 1 ppm of the surface. The specific peaks analyzed are listed in Table 2. Due to the ability to know the exact location of measurement on the sample with ToF-SIMS, a vertical measurement of the gradient of change from the oxic portion of the sample to the anoxic portion was obtained on each set of samples.

Table 2. Analytical peaks used for ToF-SIMS analysis of emplaced planar γ -Al₂O₃.

Element	ToF-SIMS Peak (Mass)
Al	27
Cl	35
Fe	56
Pb	208
S	32

RESULTS AND DISCUSSION

Pb(II) Surface Complexation

Changes in surface Pb(II) concentrations as a function of initial Pb loading and redox conditions were analyzed for each reaction time period (Figure 1).

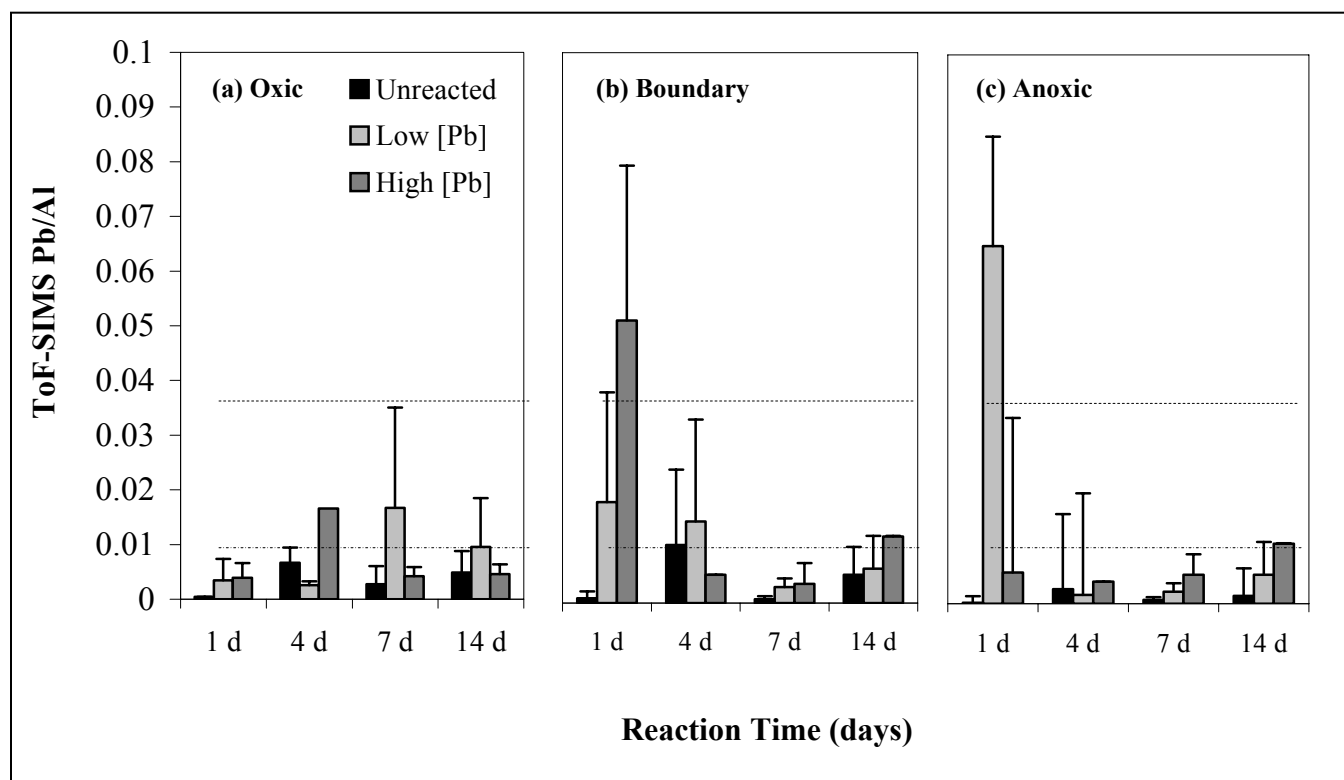


Figure 1. ToF-SIMS Pb/Al ratios of planar γ -Al₂O₃ with varying Pb loadings emplaced for 1, 4, 7 and 14 days in (a) oxidic sediments, (b) at the boundary layer between oxidic and anoxic sediments and (c) anoxic sediments. Error bars indicate the positive standard deviation of duplicate measurements. The two dashed lines indicate the Pb/Al ratios for the Low and High [Pb] samples prior to emplacement.

Unreacted Planar γ -Al₂O₃

In oxic sediments, small amounts of Pb(II) were measured on the unreacted planar surface after 4 days of emplacement. There was little change in the Pb/Al ratios between the 4, 7 and 14 day reaction times, with the ratio for the 7 day emplacement being slightly lower than the 4 and 14-day samples. In contrast, the Pb/Al ratio for the unreacted planar emplaced for 1 day is significantly lower than that for the other three reaction times. In the boundary layer between the oxic and anoxic sediments, the highest Pb/Al was again seen after 4 days. Pb/Al ratios are lower for the 1 and 7 day samples, with the ratio for the 14 day emplacement sample being more comparable to the 4 day ratio. Unreacted planar γ -Al₂O₃ emplaced in anoxic sediments had low Pb/Al ratios compared to the other redox zones, and were not significantly different than the non-emplaced unreacted planars.

Low Pb(II) Loading

In almost every case, Pb(II) concentrations decreased as a result of emplacement relative to the starting concentration. The only exception to this is the 1 day emplacement in anoxic sediments. Planar γ -Al₂O₃ with low Pb(II) loadings showed little variation with time in oxic sediments. The 7 day sample had the highest Pb/Al ratio, but because of the large deviation between replicates was not significantly different from the other samples. The 1 and 4 day emplacement samples had lower Pb/Al ratios than the 14 day sample, and the 4 day sample had a Pb/Al lower than the 4 day unreacted planar γ -Al₂O₃. In the boundary layer, planar γ -Al₂O₃ with low Pb(II) loadings emplaced for 1 and 4 days had higher Pb/Al ratios than those emplaced for longer periods of time. However, these differences may not be significant as the standard deviations for these samples are quite large. All of the planars with low Pb(II) loadings in the anoxic sediments show similar Pb/Al ratios with the exception of the 1 day sample. The Pb/Al ratio for this sample is an order of magnitude higher than the ratios of the other samples

High Pb(II) Loading

None of the planars with high Pb(II) loadings had Pb/Al ratios significantly higher than the starting materials after emplacement. In the oxic and anoxic sediments, the Pb/Al ratios were reduced significantly between 1 and 4 days of emplacement. After this initial decrease, they seemed to reach an equilibrium as they showed little variation throughout the remainder of the experiment. More variation was observed in the boundary layer than in the oxic or anoxic sediments, with the Pb/Al ratios being slightly reduced after 1 day, reaching a minimum between 4 and 7 days, and then rising again after 14 days.

Overall, Pb(II) was sorbed to the unreacted planar γ -Al₂O₃ in small amounts in the oxic and boundary layer. Little to no Pb(II) was present on unreacted planars emplaced in anoxic sediments. This is likely due to the difference in speciation of Pb(II) in these two environments. In the oxic and boundary layers, the planars are more exposed to the overlying waters where Pb(II) is likely present as an aqueous species and not yet strongly bound to mineral or particle surfaces. This is in contrast to the anoxic sediments where Pb(II) has already been interacting with the sediments and other reactive phases. Nevertheless, sufficient interaction of Pb(II) with the planar γ -Al₂O₃ occurred in all cases so that ToF-SIMS was able to detect Pb(II) concentrations on the unreacted surfaces. Since metal-sorption is a relatively fast process, it is not surprising that there is no obvious variation with time.

Planar γ -Al₂O₃ loaded with Pb(II) prior to emplacement showed different trends than the unreacted planars. Pb/Al ratios rapidly decreased upon emplacement in oxic and anoxic sediments. In the anoxic sediments, this can likely be attributed to binding of the Pb(II) by more reactive sulfide phases present in the sediments. In oxic sediments, Pb(II) may have been exposed to more reactive phases such as Fe oxides present in the surface sediments and redistribution may have occurred. Organic matter has also been shown to inhibit Pb(II) sorption to the planar surface when it is present in the aqueous phase (Conrad *et al.* 2002). In the boundary layer, the processes causing the decreases seen in the oxic layer seem to be inhibited, as it takes longer than 4 days for the Pb/Al ratios to reach their minimum.

Pb(II) and Environmental Complexants

Pb(II) is known to strongly bind to a variety of phases in both oxic and anoxic sediments, specifically Fe oxides in oxic environments and sulfides in anoxic environments. In addition, when Pb(II) is present in the water column, it can strongly interact with other dissolved ions, namely Cl and OH. To try to understand what processes may have affected the change in Pb/Al ratios during emplacement of the planar γ -Al₂O₃, Fe, S and Cl were also measured by ToF-SIMS, and the relationships between Pb(II) and these complexants were analyzed. Data were normalized to Al to minimize matrix and instrumental variation. Regression analyses were performed at a 95% confidence interval, and the results are shown in Table 3.

Table 3. Regression analysis R² values for Pb with three environmentally relevant complexants (Fe, S and Cl) for emplaced planar γ -Al₂O₃ in oxic, anoxic and boundary conditions. All analyses were performed on Al normalized data. All data were tested at a 95% confidence interval ($p \leq 0.05$). Significant correlations are marked with a *.

Sample	Redox	Fe (56)	S (32)	Cl (35)
Unreacted Planar γ -Al ₂ O ₃	Oxic	19.5	0.0	0.1
	Boundary	1.1	13.4	9.1
	Anoxic	14.1	97.4*	89.1*
Low Pb Loading	Oxic	13.2	11.9	84.6*
	Boundary	81.8*	80.6*	28.3
	Anoxic	36.5	4.2	4.0
High Pb Loading	Oxic	55.5	71.5*	84.7*
	Boundary	22.7	88.2*	92.8*
	Anoxic	10.1	14.9	0.6

Unreacted Planar γ -Al₂O₃

No significant correlations were observed between Pb(II) and the other species in the oxic or boundary layers. Pb(II) was found to be significantly correlated with both S and Cl in the anoxic sediments ($R^2 = 97.4$ and 89.1 , respectively)

Low Pb(II) Loading

In the oxic layer, the only significant correlation found was between Pb(II) and Cl ($R^2 = 84.6$). Pb(II) was also found to be related to Fe and S in the boundary layer ($R^2 = 81.8$ and 80.6 , respectively). No significant correlations were observed in the anoxic sediments.

High Pb(II) Loading

The samples loaded with high [Pb] had the highest number of significant relationships. In both the oxic and boundary layer, Pb(II) was correlated to S and Cl ($R^2 = 71.5$ and 84.7 , respectively and $R^2 = 88.2$ and 92.8 , respectively). No significant correlations were observed in the anoxic sediments.

Of the three complexants measured, Pb(II) is most commonly associated with S and Cl. These compounds are often more mobile in the environment as they are not attached to mineral surfaces as Fe oxides often are. Thus, the higher number of correlations with these species may be due to the ability of S and Cl to freely interact with the Pb(II) on the planar surface. The relationship between Pb(II) and S and Cl is strongest in the anoxic layer for the unreacted planar $\gamma\text{-Al}_2\text{O}_3$. Since these surfaces are clean upon emplacement, the association of Pb(II) with these species is likely due to a co-adsorption of these phases onto the planar $\gamma\text{-Al}_2\text{O}_3$. For the planars loaded with Pb(II), the correlations with S and Cl occur in the oxic and boundary layers rather than in the anoxic sediments. Fe is only found to be correlated to Pb(II) concentrations once, at the boundary layer on the low [Pb] samples. A rust colored layer located at the boundary region was frequently visible on the planar oxides upon removal from the sediments, and Fe/Al ratios were often highest in this region suggesting a concentration of Fe at the boundary layer.

CONCLUSIONS

This study investigated the use of planar $\gamma\text{-Al}_2\text{O}_3$ as a tool for understanding metal-ion sorption processes in natural systems by emplacing them in natural sediments for varying amounts of time. Using our techniques, we were able to observe increases in Pb(II) surface concentrations on emplaced unreacted planar $\gamma\text{-Al}_2\text{O}_3$ over time. While sorption processes under controlled laboratory conditions are generally complete within short periods of time (<24 hrs), the maximum Pb(II) concentrations measured on the unreacted planar $\gamma\text{-Al}_2\text{O}_3$ occurred after 4 days of emplacement in both the oxic and boundary layers. For planars loading with Pb(II) prior to emplacement, Pb/Al ratios almost always rapidly decreased after being introduced into the sediments. This may suggest that the Pb(II) on the planar surface is not tightly bound and is being taken up by more reactive phases present in the sediments (*e.g.*, Fe oxides and organic matter). Attempts to understand factors controlling Pb(II) surface complexation under the various redox conditions revealed that S and Cl had the largest influence of the complexants measured, but in different layers for planars with different Pb(II) loadings. This suggests that different processes are controlling Pb(II) associations for unreacted planars and those with Pb(II) already present prior to emplacement.

While this study was successful in quantifying changes in Pb(II) using planar $\gamma\text{-Al}_2\text{O}_3$, there are some limitations in using this technique. The highest amounts of Pb(II) were detected in the oxic and boundary layers. In anoxic sediments, Pb(II) levels were very low regardless of loading prior to emplacement. Thus, the use of planar oxides may be limited to oxic sediments in regions of lower contamination due to analytical and instrumental detection limit boundaries. Also, relationships between Pb(II) and other complexants were difficult to establish. This could be due to the fact that other reactive phases such as Fe oxides and organic matter out-compete the planar $\gamma\text{-Al}_2\text{O}_3$ for Pb(II). Future work should seek to establish lower limits of detectable contaminant concentrations, as well as explore possible modifications to the planar $\gamma\text{-Al}_2\text{O}_3$

surface to increase their affinity for metals making them more competitive with natural reactive phases already present in the sediments.

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CATALYTIC WET AIR OXIDATION AS A TREATMENT SOLUTION FOR ALCOHOL INDUSTRY WASTEWATERS

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ABSTRACT

Wastewaters produced during ethanol production in alcohol fermentation processes contain high concentrations of suspended solids and oxygen demanding compounds. Before being disposed of, or recycled, these wastewaters should be treated in order to meet regulatory discharge standards. Currently, concentration/incineration, and anaerobic digestion are the two main treatment systems widely practiced for alcohol producing industries. However, as standards become more stringent, the need for advanced treatment systems becomes more necessary, as conventional systems sometimes are not enough to handle waste loads from such industrial discharges. Catalytic wet air oxidation (CWAO) may be a suitable treatment method for wastewaters generated during alcohol fermentation processes. Unlike other treatment processes, catalytic wet air oxidation converts contaminants mainly to carbon dioxide and water as end products, leaving smaller amounts of residue (in charred form) for further disposal and treatment. Recent studies suggest that with the use of a suitable catalyst, more than 90% of total organic carbon (TOC) content of wastewater can be removed under mild reaction conditions and short periods of time. This paper reviews CWAO as a solution for the treatment of alcoholic beverage industry wastewaters, with its advantages and limitations.

INTRODUCTION

Wastewaters produced by many industries contain high concentrations of suspended solids, oxygen demanding compounds, and nutrients that have to be treated before being discharged into the municipal sewer system or the receiving water body, to meet regulatory requirements. The composition of the wastewater, and the concentrations of various pollutants contained within the wastewater usually determine the type of treatment that should be applied. Wastewaters produced during ethanol production for alcoholic beverages (namely the brewery and distillery wastewaters) are good examples of high-strength industrial wastewaters.

Ethanol for use in alcoholic beverages is produced by a process called fermentation. It is a process where certain types of yeast are employed to metabolize sugar in the absence of oxygen,

producing ethyl alcohol as a by-product. Beer and wine are the two alcoholic beverages that are produced this way. Since yeast organisms can only withstand about 14% alcohol, alcoholic beverages that contain higher percentages of alcohol are produced by a process called distillation, where alcohol from the brewing process is concentrated. Thus, alcoholic beverages produced by ethanol production through fermentation can be grouped into three major categories: Beer, wine, and distilled beverages.

The brewing of beer has two stages, malting of barley and brewing the beer from this malt. Brewery wastes are composed mainly of liquor pressed from the wet grain, liqueur from yeast recovery, and wash water from the various departments. During production of wines, liquid wastes originate mainly from three sources — spillage, cleanup water, and cooling water (Nemerow 1978). In ethanol production by distillation, molasses from sugar manufacture is fermented by yeast after a suitable dilution. The fermented solution contains about 6-12% ethyl alcohol, which is recovered by distillation. The effluent remaining after alcohol recovery, which is referred to as spent wash or stillage, is dark brown in color and about 6-15 times the volume of the alcohol produced. The spent wash has a very high organic content (COD=60-200 kg/m³) and is very complex in nature (Mishra 1995). Table 1 lists the composition of distillery stillage for different types of distilled alcoholic beverages.

Table 1. Composition of distillery stillage (Nemerow 1978).

Parameter	Spirit Type	Bourbon type	Molasses	Apple Brandy
pH	4.1	4.2	4.5	3.8
Total solids, ppm	47,345	37,388	71,053	18,866
Suspended solids, ppm	24,800	17,900	40	50
BOD, ppm	34,100	26,000	28,700	21,000
Total volatile solids, ppm	43,300	34,226	55,608	16,948

Currently, for alcoholic beverage producing industries, anaerobic digestion with methane recovery seems to be the commonly practiced treatment system. Concentration/incineration also used to be a viable method since the waste is high in organic content. However, concentration/incineration is becoming an outdated treatment system, because of its potential air pollution risks. Anaerobic digestion is an efficient system, yet it is a slow process and results in a lot of sludge that has to be further treated and disposed of. Table 2 summarizes treatment techniques tried on fermentation wastes. According to recent research, catalytic wet air oxidation (CWAO) may have a potential as an efficient alternative for the treatment of wastewaters produced by the alcohol fermentation industries. Unlike other treatment processes, CWAO converts contaminants mostly to carbon dioxide and water as end products, leaving minimal residues (in charred form) for further treatment.

Wet air oxidation is an advanced oxidation system, and may be defined as the oxidation of organic compounds in an aqueous solution under high pressures and temperatures. Typical conditions for wet oxidation range from 180°C and 2 MPa to 315°C and 15 MPa. Residence times may vary from 15 to 120 minutes (Luck 1999). Employing a catalyst during the wet oxidation reactions reduces the severity of the mentioned reaction conditions. The energy

requirement for catalytic oxidation is a lot lower than non-catalytic wet air oxidation reactions, and the removal efficiencies are a lot higher. Catalysts used for oxidation reactions can be divided into three classes: metals, metal oxides, and metal salts and their complexes. A variety of solid catalysts, including metal oxides of Cu, Mn, Co, Cr, V, Ti, Bi, and Zn, as well as noble metals (Ru, Pt, and Pd), have been tested as active components of catalysts in the oxidation of water pollutants (Matatov-Meytal 1998).

Table 2. Treatment efficiencies for fermentation wastes (Nemerow 1978).

Treatment process	Average BOD reduction, %
Electrodialysis	28
Chemical treatment	10
Anaerobic digestion	83
Activated sludge	30
Trickling filter	72

CWAO AND ALCOHOLIC BEVERAGE WASTEWATERS

In the 1990s, WAO started to gain popularity as a viable method for treating refractory organic material containing wastewaters. Among the work done, some studies are treatability studies, concentrating on whether the organic compound of interest can be oxidized efficiently, while others discuss the kinetics and the reaction mechanism. Some studies concentrate on partial oxidation of compounds, trying to break them down into compounds that can be beneficially used. In all cases, several reaction conditions (temperature, pressure, catalyst loading, *etc.*) are examined, and the results evaluated.

The CWAO of alcohol distillery wastewaters is yet an area to be explored. Belkacemi *et al.* (Belkacemi 1999) investigated the CWO of high strength alcohol-distillery liquors. The authors state that to their knowledge, they are the only ones to have studied the CWAO of alcohol distillery liquors. Three heterogeneous catalysts were used to evaluate the CWO reaction efficiency. These were a 1%w/w Pt over γ -alumina catalyst, manganese/cerium composite oxide, and a Cu (II)/NaY zeolite. The experiments were run in a batch reactor with a volume of 300 ml. For each run, 100 ml of distillery waste was fed into the reactor. The zero time for the reactor was taken as the time the catalyst was introduced to the system, after the oxygen. The catalyst used was 5g/liter of liquor. Temperatures of 180-250 °C and pressures of 0.5-2.5 MPa were studied. Only one sample was drawn from each reaction, generating only one data point at given reaction conditions. All samples were analyzed for protein, sugar, total Kjeldahl nitrogen, total organic carbon, and chemical oxygen demand. The Cu (II)/NaY catalyst was the most efficient catalyst under all temperatures, however, some leaching of Cu was detected. The lowest reaction rate was observed with the Pt/Al₂O₃ catalyst.

Currently in Virginia, there are numerous wineries and breweries, some with very large production capacities. It may not be feasible for small capacity facilities to practice catalytic wet air oxidation for wastewater treatment. However, with an integrated treatment system, this problem could be overcome. Ideally, this system would be a system where there is one central CWAO reactor that will process all the winery/brewery wastewaters in a certain area. Every

facility would have to pay a certain fee to be able to discharge and get their wastewater treated. A cost/benefit analysis could prove whether such a system would really be a solution to alcoholic beverage production wastewaters.

Another innovative idea might be to try to convert the ethanol in wastewater to certain aldehydes and ketones, in other words, to a product that can be used by other industries. It is a known fact that through catalytic oxidation, alcohols can be converted to aldehydes, ketones, or certain carboxylic acids. Such a concept, if can be successfully done, would benefit both the environment and the alcohol brewery/distillery industries tremendously.

Both the fermentation process for the production of alcoholic beverages and CWAO are batch processes, which might make CWAO a suitable treatment alternative for wastewaters produced by this industry. The reactors could be programmed to work only when there is wastewater coming in. Another advantage of the CWAO system is that it is a compact unit compared to traditional treatment techniques, such as anaerobic treatment. There is also appreciable energy produced during the catalytic oxidation reaction. Distilleries require considerable energy for the separation of alcohol. If the energy that is produced during the treatment process could be integrated into the energy requirement for the distillation process, it would lead to significant savings in the product cost.

As can be seen in Table 1, wastewaters from the alcoholic beverage industry contain very high concentrations of suspended solids. This may pose a problem with the CWAO process. A separate filtration unit before the reactor could be employed to remove the solids in wastewater. In this case, the solids that are collected would have to be handled and disposed of separately. Another option is to treat the wastewater as it is, with the solids. In this case, the solids would supposedly get incinerated and turn to charred form in solution during the reaction. When the catalyst is being filtered out at the end of the reaction, the solids in charred form could also be filtered out. This might cause problems if the catalyst is to be recycled and reused, as the recovery of the catalyst would be harder. Whether the high solids content would impede the performance of the treatment system can be investigated through experimental trials.

The CWAO of wastewaters from alcoholic beverage production is not an area that has been extensively researched. The process has shown to be successful for many different types of wastewaters in the past. Whether it would work for alcoholic beverage wastewaters is something yet to be examined. Cost-benefit analyses compared to other treatment techniques could determine whether this type of treatment is feasible. Table 3 summarizes the pros and cons that are discussed for the treatment of alcoholic beverage wastewaters.

Table 3. Pros and cons for CWAO of alcoholic beverage wastewaters.

Pros	Cons
Faster than conventional techniques	Can prove to be costly
Has high treatment efficiency	Suitable catalysts that do not become deactivated should be chosen
Compact size	Solids might hinder the process, therefore additional units, such as a filtration unit might be needed
Can handle high concentrations of organics	
Energy produced during the reaction may be used for the distillation process	
Could be employed to produce a usable product from wastewater	

CONCLUSION

CWAO is a lot faster than conventional techniques such as biological treatment, and leaves smaller amounts of residues that have to be further disposed of. It is also an efficient process for the removal of toxic organic compounds that are not amenable to biological treatment. According to recent studies, CWAO may have a potential for effectively treating alcoholic beverage wastewaters.

There yet remains a lot of research to be done on the CWAO of alcoholic beverage production wastewaters. Scientific research should be conducted to determine whether the proposed treatment system can efficiently treat these types of wastewaters. Different catalysts and reaction conditions should be studied to find the optimum operating conditions. Kinetics and reaction mechanism studies should be conducted to be able to design efficient reactors. More importantly, feasibility studies have to be conducted to be able to decide whether such a treatment system would prove to be more economical and efficient if used by industries. If the system does prove to be feasible and beneficial, it would be a significant progress for the treatment of alcoholic beverage wastewaters.

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THERMAL HABITAT ASSESSMENT OF ALTERNATIVE FLOW SCENARIOS IN A TAILWATER FISHERY

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The Smith River tailwater (Henry County, VA) offers a self-sustaining brown trout fishery managed for trophy trout (≥ 406 mm), however trophy sized fish are rare. Slow growth and small size are likely caused by any one or a combination of limited food resources, physical habitat, and thermal habitat. To evaluate the potential for thermal habitat improvement, temperature changes resulting from alternative flows released from the hydropeaking Philpott dam were assessed with a one-dimensional hydrodynamic model coupled with a water temperature model. Simulated temperatures from each flow scenario were assessed every 2 river kilometers over a 24 kilometer river section below the dam for occurrence of optimal growth temperatures, as well as compliance with Virginia Department of Environmental Quality hourly temperature change and daily maximum temperature standards. The occurrence of optimal growth temperatures was increased up to 11.8% over existing conditions by releasing water in the morning, decreasing the duration of release, and not increasing baseflow. Occurrence of hourly temperature changes greater than 2°C was reduced from 4% to 0-1.2% by non-peaking releases, increasing baseflow, releasing in the morning, and decreasing the duration of release. Maximum temperature occurrence greater than 21°C decreased from 1.3% to 0-0.1% by releasing every day of the week to prevent elevated temperatures on non-generation days, increasing baseflow, increasing duration of release, and releasing in the morning rather than evening. Despite conflicting adjustments to best improve all thermal criteria concurrently, a 7-day/week, morning, one hour release regime was determined to improve all criteria compared to existing conditions.

MINIMIZING WATER QUALITY IMPACTS WHEN REPAIRING BRIDGE AND CULVERT SCOUR DAMAGE WITH GROUT

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KEY WORDS: tremie concrete, underwater grout placement, bridge scour, high pH

ABSTRACT

The Virginia Department of Transportation (VDOT) has routinely used what is commonly referred to as *tremie concrete* (concrete or grout placed underwater by way of pumping through a metal tremie pipe) to repair bridge substructure and scour damage. Because the effects of tremie concrete on water quality have been a concern for some environmental agencies, a number of VDOT projects have been put on hold until the problems with tremie concrete were better documented and/or until VDOT developed a better method of repairing bridge scour. The Virginia Department of Environmental Quality requested that all in-stream scour repairs, with few exceptions, be conducted “in the dry.”

The purpose of this study was to determine a way to allow VDOT to remain in compliance with current state and federal water quality standards and regulations while rehabilitating structures with significant scour using concrete placed underwater. The study included the monitoring of 31 sites in the field and a laboratory component to compare the effects of various placement methods on various water quality parameters.

Results showed that the primary water quality parameter affected by the placement of grout underwater is pH. Such placement can cause pH values to exceed 11 under particular flow conditions. However, in-stream pH values can be kept below the state water quality level of 9.0 through the use of a combination of placement techniques and/or an anti-washout admixture. The techniques required are very site specific but depend primarily on stream flow volume and grout pumping rates.

INTRODUCTION

In the past, the Virginia Department of Transportation (VDOT) has routinely used tremie concrete for bridge repair projects designed to repair substructure and scour damage. The term *tremie concrete* refers to the pipe used to transfer concrete underwater, in this case, to fill voids near piers and/or abutments. The tremie typically consists of a vertical steel pipe, the lower end of which is designed to remain immersed in the concrete or grout (a mixture of cementitious material and water, with or without aggregate, proportioned to produce a pourable consistency) (American Concrete Institute 2000) that is being pumped into the void so that a minimum

amount of material comes in contact with the surrounding water (Gerwick and Holland 1986, Malisch 1986, Khayat *et al.* 1993).

Because the effects of tremie concrete on water quality have been a concern for some of these environmental agencies, namely the Virginia Department of Environmental Quality (DEQ) and the U.S. Army Corps of Engineers (Corps), the acquisition of these permits has become a problem. The concerns of the regulatory agencies stem from several incidents of fish kills and associated high pH levels in streams where tremie concrete was being used. Although spikes in pH levels were recorded at several sites, it was not known if other water quality parameters were also affected by the placement of concrete and therefore contributed to water quality degradation. As a consequence, the agencies put a number of VDOT projects on hold until the problems with tremie concrete were better documented and/or until VDOT developed a better method of repairing bridge scour. DEQ requested that all in-stream scour repairs, with few exceptions, be conducted “in the dry.” Performing this type of work in the dry requires installing cofferdams around the perimeter of the pour area followed by pumping out the water inside the dammed area. Once the area is dewatered, the concrete is placed. After the concrete sets, the cofferdams are removed.

Based on several cost estimates for scour rehabilitation for several VDOT structures, the cost of performing the work in the dry is just over double that of performing the same work in the wet (*i.e.*, pumping grout underwater). The increased costs are due to the installation of the cofferdams (and in some cases, the construction of causeways into the stream or river to provide access for the equipment needed to install the cofferdams) and the construction of dewatering basins. It is also safe to assume that performing the work in the dry it has the potential for negative environmental impacts as well because the in-stream construction and dewatering activities can result in increased turbidity, streambed disturbance, and altered flow characteristics. Installing cofferdams (and the associated construction of causeways) also requires significantly more time than does pumping grout underwater, potentially resulting in increased lane closure time. In many instances, particularly for smaller structures requiring rehabilitation, installing cofferdams is all but impossible because of the lack of overhead clearance.

METHODS

Field Monitoring

In-field monitoring was conducted at 31 sites in three VDOT districts during the course of the study: the 11 sites in the Fredericksburg District were repaired using tremie concrete, and the 19 sites in the Staunton District and the 1 site in the Salem District were repaired using grout bags. Approximately 900 yd³ of concrete were placed at the sites. Five of the sites yielded no water quality data because all concrete work was done in the dry because of abnormally low water levels.

Parameters Measured

Water quality parameters measured during the monitoring included pH, dissolved oxygen, conductivity, temperature, and alkalinity. These parameters were selected based on input from DEQ. In previous instances where concrete was placed and fish kills occurred, in-stream pH

values were high, but it was not known if this was the only factor contributing to the stress aquatic biota were experiencing or if other factors such as dissolved oxygen levels were also affected by the chemical constituents of the concrete. Temperature was a concern because the curing of large quantities of concrete can generate significant heat.

Site Instrumentation and Setup

At each site, water quality was monitored at three evenly spaced stations along transects located 10, 20, and 30 m upstream of the pour area; at the pour location; and at 10, 20, 30, 40, 50, and 60 m downstream of the pour area. A Horiba U-10 water analyzer containing multiple probes was physically carried to each monitoring location to measure each water quality parameter. In addition to the manual measurements, a series of CSI-M11-L pH probes connected to a Fisher Scientific CR-10 data logger were placed in the center station of each of the downstream transects to measure pH. Figure 1 is a sketch of the typical site setup.

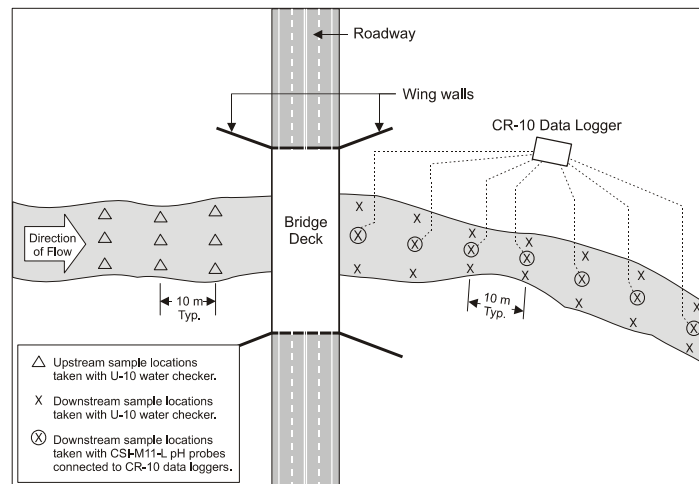


Figure 1. Instrument setup for monitoring typical structure undergoing scour rehabilitation.

Baseline Analysis

Baseline conditions of the five water quality parameters were measured for each site. In most cases, baseline readings were taken within 1 hour of the start of concrete placement to minimize any changes that could take place because of precipitation, temperature, and diurnal fluctuations caused by plant photosynthesis. Measurements of pH, dissolved oxygen, temperature, and conductivity were taken with the Horiba U-10 water analyzer at each monitoring station as shown in Figure 1. Single water samples were collected at the pour locations using 250-mL polyethylene bottles. The samples were analyzed for alkalinity at a commercial lab in Charlottesville, Virginia, after each pour was completed. Stream flow velocities and water depths were measured near the center of each transect using a Global Water FP-201 stream velocity probe. All sampling equipment was calibrated in accordance with the manufacturer recommendations prior to establishing the baseline water quality for each site.

Monitoring During and After the Pour

Following the initiation of concrete pumping, water quality parameters were initially measured manually at three locations along each transect. It soon became apparent that the greatest water quality impacts were occurring immediately downstream of the pour location. Subsequently, for the Staunton and Salem districts, readings were taken at each of the three monitoring stations at the 10-m downstream transect every 15 minutes and hourly readings were taken for each monitoring station further downstream. For sites in the Fredericksburg District, water quality data were measured manually at least once every hour and pH was recorded every 15 minutes using the data logger system. Monitoring of water quality was continued following the completion of pumping at each site until the water quality parameters returned to baseline values. Benthic and habitat surveys were conducted 3 to 5 days after the completion of the concrete placement.

Alteration of Placement Techniques

At the sites monitored during this project, grout was placed in the voids around or under the structures by one of two methods: pumping through a tremie pipe, or pumping into grout bags. The methods are described here.

Tremie Pipe

At all 11 sites monitored in the Fredericksburg District, grout was placed around or under the structure undergoing scour rehabilitation. This was accomplished by pumping the grout from a grout pump or pump truck located on top of the structure through a series of hoses and into a tremie pipe. The steel tremie pipe extended down beneath the surface of the water, through a geotextile fabric, under the structure, and into the void being filled. The geotextile fabric served as a boundary to prevent the grout from coming out of the void while at the same time allowing the water being displaced by the grout to exit the void. Once the void was full, the grout was allowed to set and the tremie pipe was cut off below the surface of the water.

Grout Bags

All 19 sites in the Staunton District and the single site in the Salem District were repaired by pumping grout into grout bags. Most of the bags used for this project were purchased from Fabriform, but grout bags, which come in different sizes and weights, are available from a number of manufacturers. The sizes used for this project varied from site to site and even within a given site but typically ranged from 3 ft x 5 ft x 1 ft to 5 ft x 12 ft x 2 ft. The different sizes accommodated varying volumes of concrete to be placed depending on the dimensions of the scoured area being filled.

Turbidity Curtain

At several sites in the Fredericksburg District and nearly all sites in the Staunton District, turbidity curtains were installed just outside the pour area. Turbidity curtains are made of a thick plastic and suspended by way of a series of foam booms sewn in the top edge of the plastic. A metal chain weights down the bottom edge of the curtain. When placed in water, the curtain forms a barrier that although not impermeable, inhibits the flow of water from one side of it to the other.

Anti-washout Admixture

An anti-washout admixture (AWA) designed to improve the performance (*i.e.*, strength) of concrete placed underwater by decreasing the percentage of fines and cement paste that are washed out prior to setting was used on the Salem District site on the Roanoke River. The AWA used was Rheomac UW450m, and it was added at the manufacturer's recommended rate of 80 oz/yd³ of concrete. Because there were concerns that the increase in the viscosity caused by its addition would negatively affect the ability to pump the grout, a water-reducing admixture, Rheomac 3030, was also added at a rate of 60 oz/yd³. Normal pumping procedures were followed with the grout containing the AWA by pumping directly into grout bags.

Laboratory Analysis

To quantify further the water quality effects of the different placement techniques used in the field, several combinations of grout bag materials and admixtures were analyzed in a laboratory setting. All tests were run under controlled conditions in the concrete and geotechnical laboratories at VTRC.

Experimental Design

The laboratory analysis was designed to determine the *relative* differences between the placement techniques that were, or could easily be, implemented in the field. Because stream flow conditions, baseline water quality parameters, and quantities of concrete used in the field varied tremendously, these tests were not set up to replicate water flow conditions that might be experienced when concrete was actually placed underwater at a field site. As a consequence, the pH values obtained in the laboratory analysis would not be the same as those expected in the field. Instead, the laboratory values provided a statistically valid means by which to compare different placement scenarios that would be used in the field.

Cylinders constructed of polyvinyl chloride (PVC) pipe with an inside diameter of 38 mm and a length of 120 mm were covered on one end with filter fabric or one of two types of grout bag material. These were stretched tight over the end of the cylinders and held in place with plastic wire ties. Grout was mixed following a typical VDOT mix design and placed in the cylinders to a depth of 90 mm. The cylinders were then suspended in 1000-mL glass beakers containing 500 mL of tap water for 90 minutes. A Fisher Scientific Accumet pH Meter 925 was used to measure the pH of each sample before and after insertion of the cylinders containing grout.

Materials Analyzed

Two grout bag materials were tested in the laboratory tests: Fabriform PJ 400 and Fabriform Ballistic. The Fabriform PJ 400 grout bag was the same as that used at sites monitored in the field. The Fabriform Ballistic is a tighter weave fabric and was not used in the field. A standard geotextile fabric similar to that used when concrete was placed by way of a tremie pipe was also tested. All three materials were tested with the standard grout mix, and this same mix containing the anti-washout admixture Rheomac UW 450. Controls (the three material types and cylinders containing no grout) were also run for each combination.

Assessment of Factors Contributing to Water Quality Degradation

Following the completion of the field monitoring and laboratory component of the study, an analysis of all the factors contributing to the degradation of the water quality at sites undergoing scour countermeasures was conducted by looking at the relationships between the site variables measured and the grout placement techniques both in the field and laboratory. Based on these findings, specific methods of reducing the chances of adverse impacts on water quality were identified.

RESULTS AND DISCUSSION

Field Survey

Numerous parameters were measured at close time intervals for extended periods of time at multiple locations for each site, resulting in the collection of a tremendous amount of data for the 31 sites undergoing scour rehabilitation. Consequently, only a subset of these data that represents the most important findings is reported here. Results of the benthic surveys are not included in this report but are available in a separate document from the author.

Tremie Concrete Placement

One of the first findings derived from the field monitoring was that of the four primary water quality parameters measured, only pH changed significantly during the concrete placement process. Temperature and dissolved oxygen had a diurnal fluctuation but changed only minimally during the placement process. The values never exceeded the ranges specified in VAR 680-21-01.5. Conductivity values did not appear to follow any specific pattern and varied as much or more from site to site as they did as a result of concrete placement.

No Turbidity Curtain Used

Summary data showing some of the maximum values for the five sites where tremie concrete was placed without using a turbidity curtain are shown in Table 1.

Table 1. Summary pH data for tremie sites without turbidity curtains.

Property	Value
Maximum pH	10.9
Maximum increase in pH	4.1
Maximum pH @ 50 m downstream	10.9
Maximum time required for pH to return to baseline (h)	4.6
Number of sites pH exceeded 9.0 threshold	2 of 5

Significant increases in pH were recorded for most of these sites. The greatest changes were recorded within 10 m downstream of the pour area. The pH increases typically occurred within several minutes of the commencement of pumping, but this varied from site to site. Two of the sites had pH values above the 9.0 state water quality standard. Once the pH of the water became elevated following the initiation of pumping, it declined only slightly at the downstream monitoring stations, indicating that dilution or dispersion of the high pH plume did not occur to any significant extent. With respect to time, the decline of the pH readings varied from 1.5 to 4.6 hours following the completion of the concrete placement. Those sites where the greatest

quantity of concrete was placed did not have the highest total rise in pH, but the time it took the pH to return to baseline conditions was greater.

Turbidity Curtain Used

Summary data for the six sites where tremie concrete was placed using a turbidity curtain are shown in Table 2.

Table 2. Summary pH data for tremie sites with turbidity curtains.

Property	Value
Maximum pH (outside curtain)	9.0
Maximum pH (inside curtain)	11.1
Maximum increase in pH (outside curtain)	1.8
Maximum pH @ 50 m downstream	9.0
Maximum time required for pH to return to baseline (h)	32
Number of sites pH exceeded 9.0 threshold (outside curtain)	0 of 6

Elevated, but not excessively high, pH values were recorded for most of the sites where tremie concrete was placed and separated from the remainder of the stream by a turbidity curtain. With few exceptions, the pH inside the curtained areas quickly rose above 10.0. This sudden jump in pH can be attributed to the lack of water coming into the curtained area, preventing the dilution of the higher pH concrete. This lack of dilution was evident in the bright gray color of the water contained in this area. pH values outside the curtain were significantly lower and never exceeded the 9.0 threshold for any of the six sites.

The elevated pH values both inside and outside the curtained area returned to baseline values more slowly than at the sites without the curtain. The pH values inside the curtain took longer to return to normal than those outside the curtain. Unless water was deliberately allowed inside through opening the upstream side, pH values inside the curtain remained elevated for at least 12 hours and in some cases more than 30 hours following completion of the pour. Because the curtains were not watertight, the high pH water inside slowly “bled off,” contributing to the slightly elevated values (though still lower than sites where no curtain was used and below the 9.0 water quality threshold) outside the curtain for extended periods. As the higher pH water leaked out of the curtained area, it was replaced with water flowing in from upstream. Over a period of hours, the system reached equilibrium, with the pH being the same inside and outside the curtain.

Grout Bag Placement

No Turbidity Curtain Used

At only two sites were grout bags placed in areas that were not contained by turbidity curtains. Both sites had very high pH levels. One of these sites had grout placed essentially in a standing pool of water. As the grout bags were filled, the pH climbed quickly to just over 11 in the pool. These values did not decline for the 23-hour period for which the site was monitored.

The second site had very low flow. Though efforts were made to prevent concrete from entering this flow by diverting the water around the area where grout bags were being placed, severe

spikes in pH occurred during pumping. However, the levels quickly dissipated when pumping was complete for each of the three trucks.

Turbidity Curtain Used

Data were collected at 12 sites where concrete was pumped into grout bags and the area was cordoned off by turbidity curtains. Table 3 is a summary of the maximum pH at these sites.

Table 3. Summary pH data for grout bag sites with turbidity curtains.

Property	Value
Maximum pH (outside curtain)	9.8
Maximum pH (inside curtain)	11.5
Maximum increase in pH (outside curtain)	2.6
Maximum pH @ 50 m downstream	8.3
Maximum time required for pH to return to baseline (h)	9
Number of sites pH exceeded 9.0 threshold (outside curtain)	2 of 12

Ten of the sites with grout bags in conjunction with a turbidity curtain had in-stream pH levels below 9.0. As was the case when the turbidity curtains were used with the tremie concrete placement, pH values inside the curtain exceeded 10 most of the time during the concrete placement and returned to baseline values only after an extended period.

Placement with Use of Anti-washout Admixture

Over a 7-day period, 170 yd³ of grout containing an AWA was placed in grout bags at a single structure in the Roanoke River. Summary results for pH levels at this site are shown in Table 4. At no time during the pumping, even when pumping occurred in areas of slack water, did pH readings go above 8.9.

Table 4. Summary pH data for grout bag sites with anti-washout admixture.

Property	Value
Maximum pH	8.9
Maximum increase in pH	1.7
Maximum pH @ 50 m downstream	8.3
Maximum time required for pH to return to baseline (h)	2.5
Number of sites pH exceeded 9.0 threshold	0 of 1

Laboratory Analysis

The mean pH for each of the grout placement techniques tested in the laboratory is shown in Figure 2. The standard deviation associated with each of these was extremely small (the largest was 0.096). Of the 78 samples analyzed, those contained by filter fabric were associated with the highest pH values in the surrounding water. The grout samples with the AWA and contained by both types of grout bags were associated with the lowest pH values in the surrounding water. Again, the actual pH values are not indicative of what would be obtained in the field because of the small volume of water available for dilution. In most field situations, the water to grout ratio would be much greater, resulting in lower overall pH values.

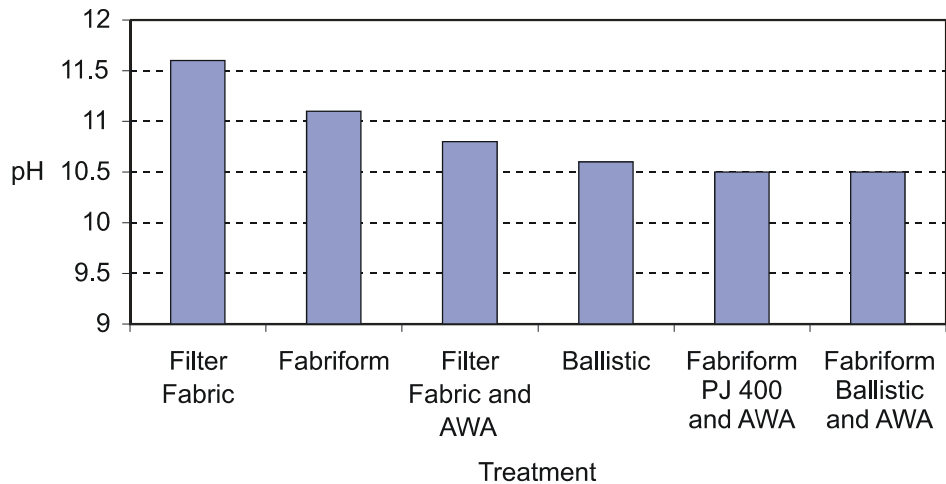


Figure 2. Mean pH values for each of six placement techniques measured in laboratory.

Assessment of Factors Contributing to Water Quality Degradation

Rises in pH represented, by far, the greatest changes in all of the water quality parameters measured. The rise in pH caused by the hydration of the cementitious materials in the grout results in the high concentrations of hydroxyl ions (OH⁻) going into solution in the water surrounding the concrete. As a consequence, the factors contributing to or limiting the negative effects of grout on the stream water quality are related to the reduction of the relative amount of free hydroxyl ions in the water surrounding the pour area.

Stream Flow

Stream flow is a function of the velocity of the water moving through the pour area and the size of the stream (or volume of water for a given area in the stream). From data collected in the field, it appears that total stream flow is one of the most critical factors influencing the rise in pH when concrete is placed. The Mossy Creek site in the Staunton District is a Blue Ribbon Trout Stream. Because of this designation, there was great concern that rising pH levels would be detrimental to the fish known to be present at the site and immediately downstream. Because of the high volume of water moving through the site, pH values rose only 0.3 pH unit above baseline values. Even values inside the turbidity curtain at this site stayed well below 9.0 because of the strong current of water moving into and out of the curtained area. Most of the sites in the Fredericksburg District with pH values that remained below 9.0 had large volumes of water contributing to the dilution of the hydroxyl ion concentration. At the other extreme, one of the Freemason Run sites in Augusta County had no visible flow at the time of concrete placement, only pooled water immediately under the bridge structure. Within 30 minutes after pumping was started, pH values exceeding 11.0 were recorded. These values did not decline in 24 hours. This indicates that once the buffering capacity of the water is exceeded, the only way of bringing the pH back near baseline values is dilution by the introduction of additional water to the site.

Although stream flow volume is relatively easy to calculate for a stream, it is difficult to measure precisely the volume of water that will be immediately available for dilution at a given pumping location within the stream. Pumping typically occurs at several locations along one or more abutments or at a number of pier foundations for a structure. In many cases, each location has a

different water depth and velocity. This results in different dilution factors for the same structure.

Grout Pumping Rate

The rate at which grout was placed also had a direct affect on the rise in the pH of the surrounding water. Similar to the stream flow variable, the rate at which grout was pumped underwater directly influenced the concentration of free hydroxyl ions in the surrounding water column. Most of the sites with pH values in excess of 9.0 (outside the turbidity curtain if one was used) were associated with pumping rates that exceeded 13 yd³/hr. Some of these sites had as few as 6 yd³ of grout placed for the entire job, but the *rate* at which the grout was placed exceeded 13 yd³/hr.

When pumping rates are combined with stream flow volumes, the data become even more meaningful. At rates above 10 to 12 yd³/hr, it takes a large volume of water moving through the site to dilute adequately the free hydroxyl ions coming from the grout mix. Five of the six sites with in-stream pH values above the 9.0 standard had a water (stream flow volumes) to concrete (grout pumping volumes) ratio of 40:1 or less. The single site with a higher ratio had an unexpected release of high pH water from inside the turbidity curtain when one end of the curtain came untied.

Turbidity Curtains

Turbidity curtains helped keep in-stream pH values in check by simply containing the high pH water. Although containment resulted in relatively stable in-stream pH values because of the relatively small volumes of water inside the curtained area, pH values for this contained water were extremely high at most sites, resulting in a number of fish being killed. Once it became obvious that the turbidity curtain was trapping fish, efforts were made to install the curtain as close as possible to the bridge abutment and then move the curtain outward to its final location (no more than 1 m beyond the perimeter of the area where grout placement was to occur). This outward dragging motion helped prevent the inadvertent trapping of fish inside the curtained area, where the pour was to take place. Also, by keeping the curtain as close as possible to the outer edges of the grout bags, the volume of water within the curtained area was kept to a minimum.

The success or failure of the turbidity curtains to contain the high pH water depended upon how well the bottom of the curtain followed the bottom of the streambed and how well the ends were secured. On smooth, fine-grained bottoms, good contact was maintained. In cases where the bottom contained large rocks, riprap, or brush, adequate contact with the bottom was not possible, and therefore containment was not as good.

Grout Bags

Both field and laboratory data indicate that grout bags are beneficial in reducing the pH levels resulting from the placement of grout underwater. These same data, however, indicate that the use of the standard Fabriform PJ 400 grout bags alone is not sufficient to ensure that pH levels remain below 9.0 in all stream flow and/or pumping situations.

Research conducted by Construction Techniques, Inc. (1995), indicates that an estimated 0.5% of the total cement paste pumped into Fabriform PJ 400 grout bags will escape through the fabric. Based on this finding, and the average cement content for typical grouts, it was recommended that the rate of concrete placement (cubic yards/hour) not exceed the rate of water flow (cubic yards/minute) through the site to ensure that the rise in pH in the surrounding water did not exceed 1. This same ratio of 60 parts water to 1 part concrete is said to apply to grout placed in stagnant water (Construction Techniques, Inc. 1995). Although no specific tests were done to verify these findings, based on field data that were collected as a part of this study, it is safe to assume that with a 60:1 water to concrete ratio, the use of grout bags would keep in-stream pH values below 9.0 for streams with a baseline pH of 8.0 or less.

The data obtained from the laboratory component of this study show that the Fabriform Ballistic grout bag is significantly tighter than the Fabriform PJ 400 grout bag material, resulting in the escape of less cementitious material into the water column. Although the decreased porosity of the Fabriform Ballistic material would appear to be advantageous in most pumping conditions, this particular bag requires venting by way of a small vent pipe when being filled in the field (as a result of the weave being so tight). For most situations, the need to vent would prevent placing the tops of these bags underwater, as grout would then be allowed to escape from the vent pipe directly into the water.

Anti-Washout Admixtures

The AWA tested helped keep in-stream pH values in check by simply keeping the free hydroxyl ions from escaping the cement paste and entering the water column. The use of AWA alone resulted in a significant drop in pH in the surrounding water in comparison with the plain grout. When the AWA was combined with either grout bag material tested, an even greater reduction in pH was obtained. From the different combinations tested, it appears that placing grout containing an AWA in a grout bag separated from the stream by a turbidity curtain would result in the lowest in-stream pH values.

CONCLUSIONS

- pH is the primary water quality parameter negatively affected by the placement of grout underwater for the rehabilitation of scour. Other parameters such as dissolved oxygen, conductivity, and temperature do not appear to change significantly.
- The placement of grout underwater can result in pH values well in excess of the DEQ-specified limit of 9.0 and in some cases approach 12. The elevated pH levels will typically return to baseline values in 4 to 6 hours after grout placement is complete. The absolute rise in pH and the duration of the rise are dependent on a combination of stream flow volume and grout placement rates.
- The rise in pH is not directly correlated with the volume of grout placed at a structure.
- When grout is placed underwater, in-stream pH values can be kept below the state water quality level of 9.0 by using a combination of placement techniques and/or an AWA. The

techniques required are very site specific but depend primarily on stream flow volume and grout pumping rates.

- Significant savings can be realized by placing grout underwater at structures requiring scour countermeasures as compared with doing the repair in dry conditions.

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HYDROLOGIC CHARACTERIZATION OF SINKHOLES IN AGRICULTURAL SETTINGS: IMPLICATIONS FOR BEST MANAGEMENT PRACTICES

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ABSTRACT

Soils in karst terranes have very high agricultural productivity and often support intensive hayland/pasture-based systems as well as row crop production systems. Application of animal manures and biosolids to these agricultural systems can have an adverse impact on the water quality in the underlying karst aquifers. However, there is little information available on how to manage surface application of fertilizers so that contamination of karst groundwater is minimized.

The main objective of this study is to characterize the hydrology and contamination potential of sinkholes in an agricultural setting located at Kentland Farm, McCoy VA. The first task involves physical characterization of the epikarst in sinkholes, which has been conducted using Electrical Resistivity Tomography (ERT). This geophysical tool allows subsurface features such as pinnacles, mud filled fractures, saturated zones, solutional channels, and the bedrock-soil interface to be modeled. Collection of these data will allow better placement and installation of hydrological instrumentation designed to sample groundwater and agricultural contaminants moving through highly heterogeneous sinkhole environments. In this research, the primary source of contamination is application of poultry litter, biosolids and commercial fertilizer treatments.

Characterization of sinkholes will provide scientific data to support development of reasonable Best Management Practices (BMPs) by identifying and better understanding the processes that control flow and transport in sinkhole environments. Farmers, regulators and educators can utilize these BMPs to minimize adverse effects of agricultural practices on karstic groundwater systems.

INTRODUCTION

Karst is a geologic landform formed by the dissolution of soluble bedrocks such as limestone or dolomite. In regions underlain by folded sedimentary rocks, these carbonates are commonly found in valley bottoms because they tend to weather faster than the surrounding non-carbonates. In these valleys, bedrock weathering produces soils suitable for a variety of agricultural uses. Large rivers leave nutrient-rich sediment deposits in many of these broad valley floors, further increasing their agricultural productivity. The Shenandoah Valley and other valleys in the Valley and Ridge geologic province are good examples of karst regions that are used extensively for agriculture. Other agricultural regions in karst are found in Florida, the Ozark Plateau, Kentucky, Illinois, Indiana, and portions of Texas. Due to the soluble nature of carbonate bedrock, karst terranes commonly contain features such as springs, caves, blind valleys, dry streams, collapse features, and sinkholes. Sinkholes, which are most often caused by preferential dissolution along structural weaknesses in the underlying bedrock, form as the soluble bedrock is slowly removed from below and surface materials move downward, creating a depression.

Sinkholes can provide a direct hydrologic connection between the surface and an underlying karst aquifer. As a sinkhole develops, it creates a surface basin that is internally drained through enlarged fractures and other solutional features in the epikarst. Precipitation on land surfaces within the drainage basin may rapidly enter a karst aquifer via these preferential flow paths. As a result, contaminants present on land surfaces and in soils can be quickly transported to the karst aquifer. This rapid transport provides little opportunity for natural degradation or attenuation of contaminants before they reach potential groundwater sources. Once they have entered a karst aquifer, contaminants can be rapidly transported via discrete conduits to receptors such as drinking water wells.

Karst is widely recognized as being susceptible to contamination, and it is well known that sinkholes provide a “short circuit” for contaminants from the surface to reach an aquifer system. However, few, if any, scientifically based BMPs (Best Management Practices) exist to provide guidance to farmers or regulators wishing to minimize the impacts that various land uses have on water quality in karst aquifers. Some of the existing BMPs are based on surface water rather than on karst systems, while others do not provide any specific guidelines for managing nutrients in karst. Critical to the development of sound BMPs is a clear understanding of processes controlling flow and transport in sinkholes. This is especially true in agricultural sinkholes where the sink floor is used for cropping or grazing and there is no obvious connection, such as an open throat or cave entrance, between the surface and the subsurface or the aquifer itself.

SITE DESCRIPTION

The research site is located on the Virginia Tech Kentland Experimental Farm near Blacksburg, VA in the Valley and Ridge geologic province. This farm provides access to several hundred acres of well-developed karst landscape containing dozens of sinkholes currently being used for a variety of agricultural practices. The site is underlain by the Cambrian-aged Elbrook formation (Henika and Schultz 1991). This primarily dolomitic bedrock is mantled by saprolite and river terrace deposits of variable thickness and age.

METHODS

The first step in evaluating hydrology and contaminant transport in sinkholes is physical characterization of the surface and subsurface. Surface characterization is based upon parameters such as sinkhole diameter and depth, the presence of any well-defined surface drainage ways, steepness of slope, and the drainage area captured by the sink. To aid in this characterization, a detailed topographic survey of the six sinkholes has been completed. This survey provided elevation data more accurate than what is available on standard USGS topographic maps.

Electrical Resistivity Tomography (ERT), a geophysical tool, has been used in the first stage of subsurface characterization. ERT uses a linear array of electrodes to generate electrical fields in the soils and bedrock below an ERT transect. Increasing or decreasing horizontal electrode spacing provides greater or lesser depth penetration and a corresponding decrease or increase in model resolution. By measuring spatial variations in the electrical field, the electrical properties of subsurface materials can then be modeled and interpreted. ERT is a proven method for identifying a variety of subsurface features including the bedrock-soil interface (Roth *et al.* 2002, Roth and Nyquist 2003, Zhou *et al.* 1999). Heterogeneities in the bedrock surface (such as mud filled fissures, bedrock pinnacles and solutional channels), and in the soil overburden (saprolite vs. terrace deposits and dry vs. wet soils), can also be resolved. Understanding the nature of these features is important since it is assumed that sinkhole heterogeneities strongly influence subsurface flow and transport. For this research, electrode spacings were used that maximized resolution of this interface.

RESULTS AND DISCUSSION

Nearly 100 ERT transects have been completed across six sinkholes at Kentland Farm. Transects are oriented so that most are perpendicular or parallel to visible surface features such as swales and other topographic features (Figure 1). It is assumed that major features in the bedrock surface follow surface topography.

Following data collection, ERT transects (Figure 2) were analyzed to obtain the location and elevation of bedrock surfaces and saturated zones within the soil horizon. Processing the ERT data and creating a 3-D model of the bedrock-soil interface from 2-D transects is challenging, but a complex data processing technique has been developed to produce data sets that can easily be used to model this surface with SURFER. ERT data are modeled using Geotomo Software's RES2DINV inversion software. Images showing model resistivity values for each transect are then saved in raster format. The raster transect images are scaled and imported into a 3-D AutoCAD 2000 drawing where each bedrock-soil interface is visually interpreted and drawn as a 3-D line. After all sinkhole transects have been processed in this manner, the resulting drawing is saved and is imported into Microsoft EXCEL where a filtering and editing process produces X, Y, Z data points representing the bedrock surface. These data are then combined with the surface topography to create a 3-D model of a sinkhole in SURFER 8 (Figures 3, 4, and 5). This model can then be used to aid in choosing instrumentation locations that have a high probability of intersecting the subsurface flow. Instrumentation will consist of wells, multi-level samplers, pan lysimeters and an overland flow collector installed in a transect from the sink bottom to the

rim. The objective of the instrumentation is to sample groundwater and contaminants originating in or near a sinkhole.

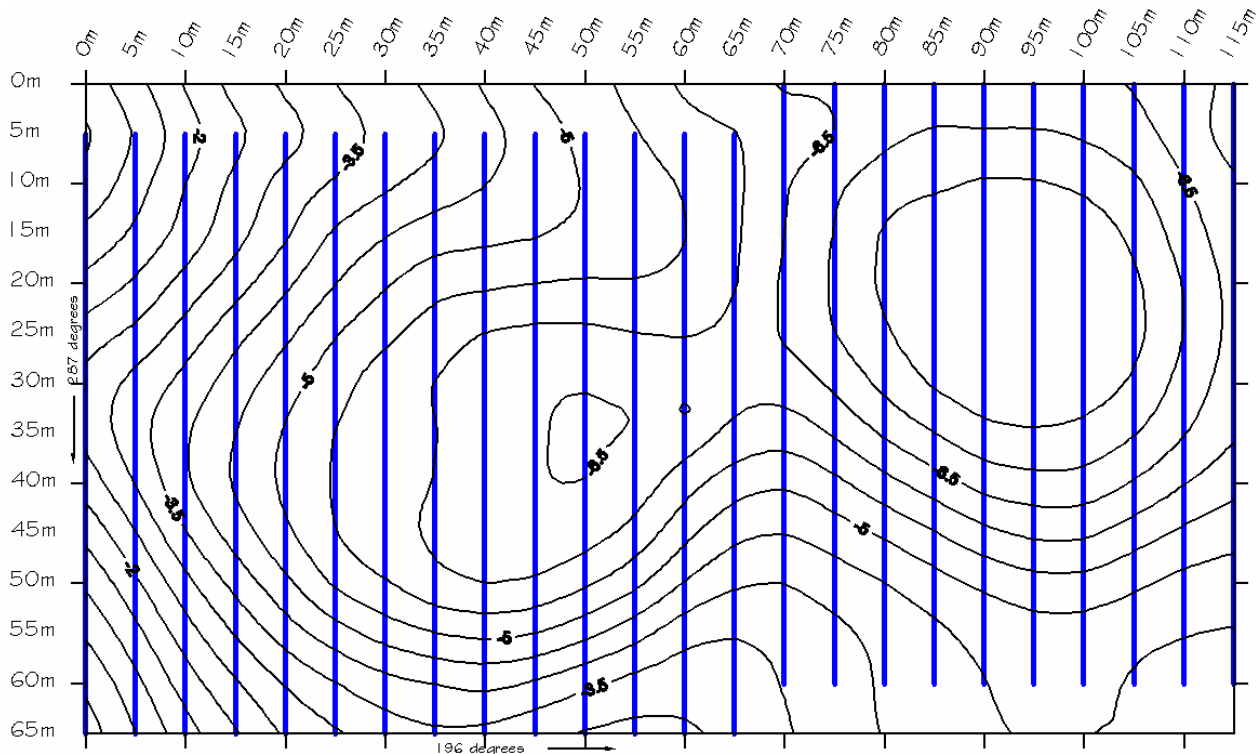


Figure 1. Topographic map showing two sinkholes. Bold lines represent ERT transects. Scale is in meters. Contour intervals are 0.5 meters.

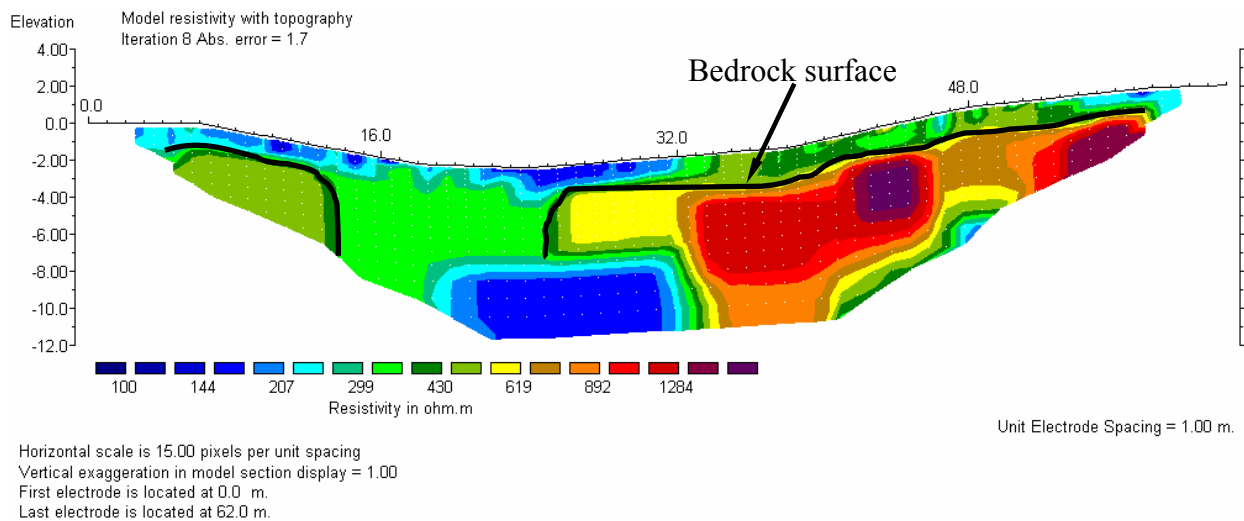


Figure 2. Sample ERT transect across a sinkhole. The interpreted bedrock-soil interface is shown in bold lines. Low resistivity areas likely represent saturated zones or wet soils. Vertical and horizontal scales are in meters.

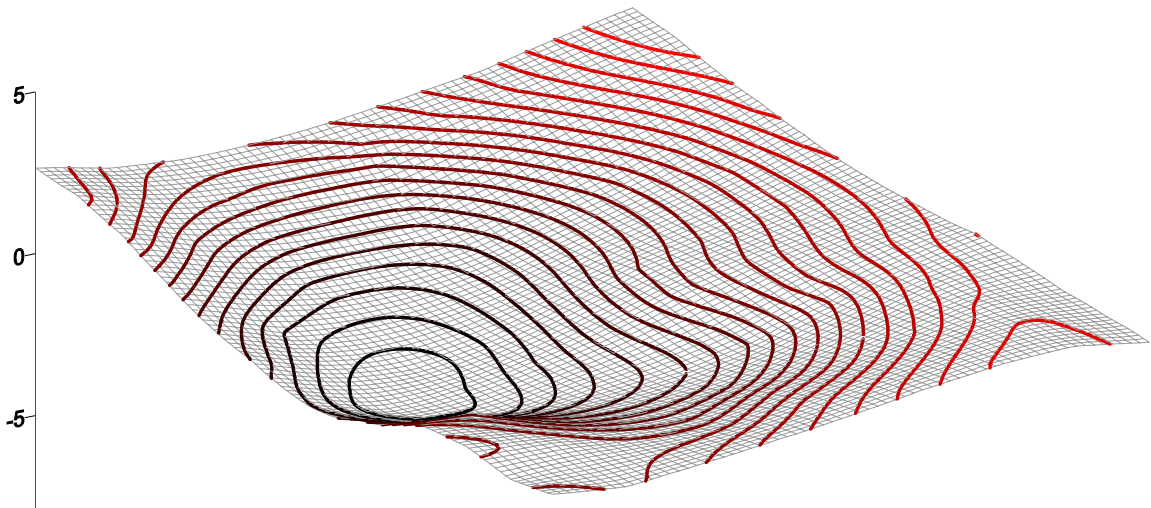


Figure 3. 3-D model of surface topography over a sinkhole. View angle is 20° above horizontal. Scale is in meters.

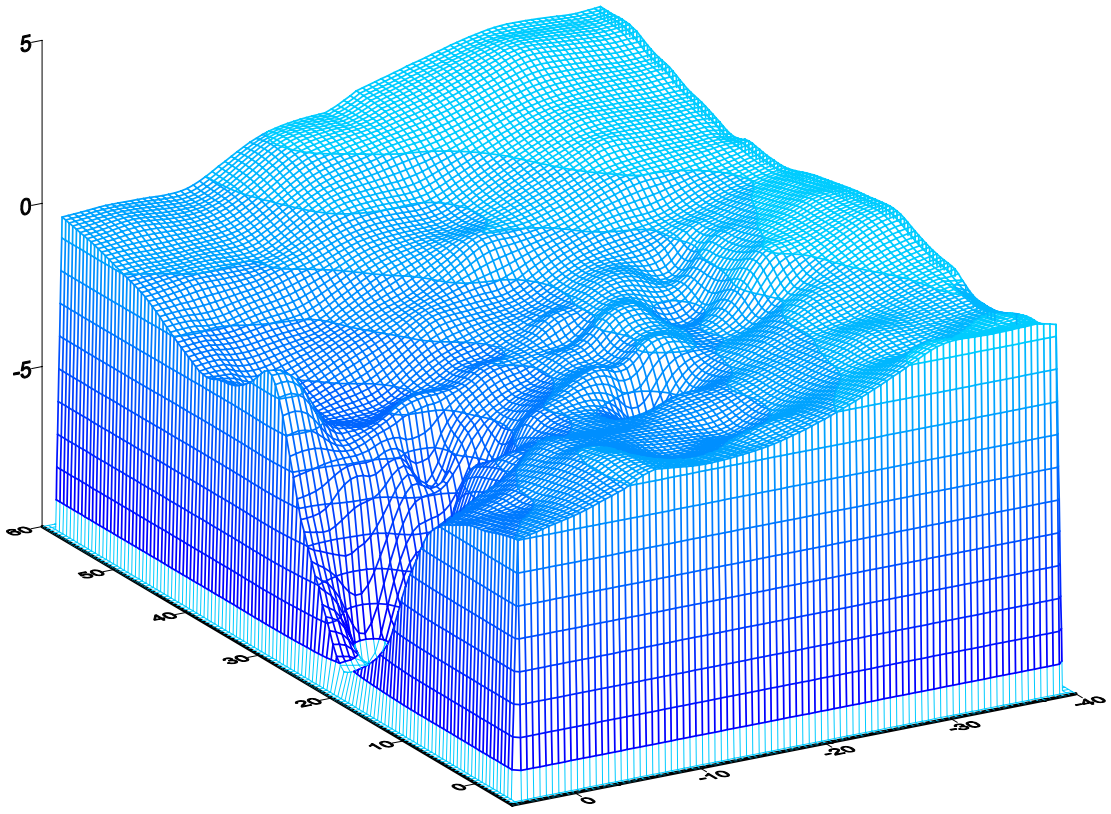


Figure 4. 3-D model of bedrock surface topography in a sinkhole. View angle is 20° above horizontal. Scale is in meters.

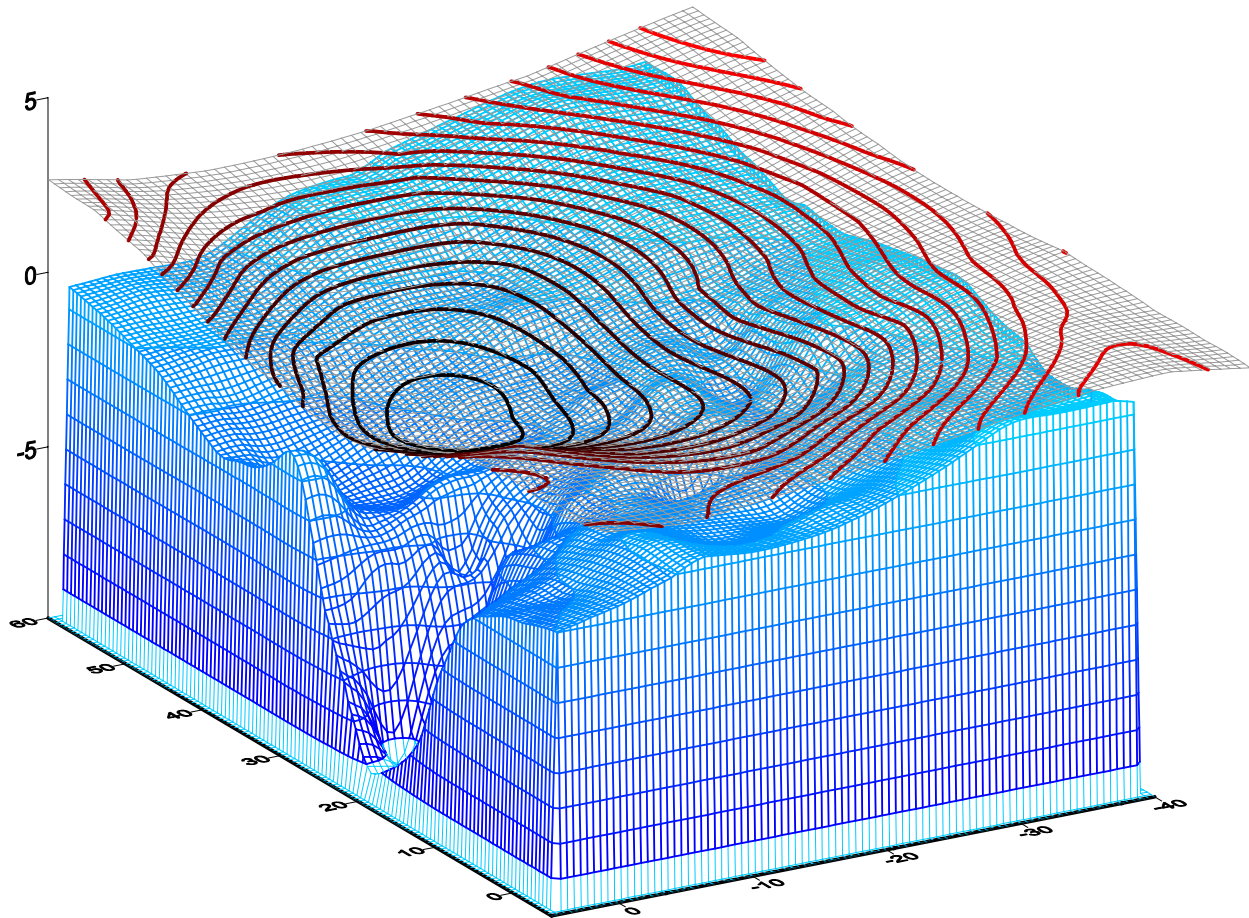


Figure 5. A 3-D model showing part of a sinkhole with surface topography overlying a bedrock surface obtained from ERT transects. Note sinkhole throat is clearly visible on lower left side. ‘Space’ between upper and lower surfaces represents the soil interval. Vertical exaggeration is 3X. Scale is in meters. View angle is 20° above horizontal.

CONCLUSIONS

Preliminary results from ERT transects show that the method provides good resolution of subsurface features such as the bedrock-soil interface. Models show that bedrock surfaces roughly mirror surface topography, though with greater vertical relief and more irregularities. ERT is also useful for resolving features such as the sinkhole throat and large bedrock channels. Resolving these subsurface features is important, as the heterogeneities will play a critical role in controlling flow and transport rates within individual sinkholes. For example, thicker soils in sinkhole throats may serve to attenuate contaminants carried to sink bottoms by overland flow during flood events. Variations in soil composition will also have an effect on how certain contaminants move through the epikarst and into an aquifer. Although understanding soil thickness can be a critical component in developing a hydrologic model, small-scale heterogeneities commonly found in the epikarst, such as soil pipes, may negate the effects thicker soils normally have on retarding contaminants. ERT is not capable of resolving small

features such as soil pipes, though they are often controlled by larger scale variations in the bedrock surface, which can be resolved using ERT.

This research will provide an improved understanding of processes governing hydrology and contaminant transport in and through sinkholes. These data will support development of models and BMPs that will be used to improve management and protection of valuable groundwater resources in karst regions.

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RECLAIMING GROUNDWATER CONTAMINATED BY VOCS AND PERCHLORATE

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ABSTRACT

A pump and treat (P&T) system was built in the late 1980s at an industrial facility in northern Virginia to extract and treat chlorinated VOC-contaminated groundwater from a deep bedrock aquifer. The system extracted at rates of 60 to 80 gpm for over 12 years. Extracted groundwater was pumped through a forty-foot air stripper tower and two 8,000-pound GAC vessels to remove approximately 99.9% of the VOCs. Following treatment, an estimated 75% of the extracted groundwater was discharged to surface water under a VPDES permit, whereas the remainder was used onsite for facility processes. Detection of perchlorate (a component of rocket propellant) in the deep aquifer in 2001 prompted deactivation of the P&T system, as the system was not designed to treat perchlorate. As a result, the facility fully converted to the local county water system for process needs. After examination of the existing P&T system, it was determined that with minor modifications, the system could be converted to an *in situ* recirculating anaerobic bioremediation (RAB) treatment capable of remediating both VOCs and perchlorate while maintaining hydraulic control of both plumes. The converted system consists of an extraction well, existing GAC vessels, a substrate amendment system, and a former extraction well converted to an injection well. After eight months of pilot-scale operation of the converted system at approximately 30 gpm, perchlorate concentrations in the extraction well were reduced by approximately 71%. As compared to typical P&T systems, which waste a large proportion of the extracted groundwater to surface water, the facility's RAB system replaces 100% of the extracted groundwater back to aquifer storage. As a result, the system presents threefold benefits in preserving precious groundwater and economic resources while reducing groundwater VOCs to concentrations that are protective of human health and the environment and perchlorate to below current method detection limits.

INTRODUCTION

In the late 1980s, a pump and treat (P&T) system was built at an industrial facility in northern Virginia to extract and treat groundwater from a deep fractured bedrock aquifer impacted by chlorinated volatile organic compounds (VOCs) such as tetrachloroethene (PCE) and trichloroethene (TCE). The P&T system consisted of two extraction wells operating between 60 to 80 gallons per minute (gpm), a forty-foot air stripper tower, and two 8,000-pound granular activated carbon (GAC) vessels. The stripper tower and GAC vessels removed approximately

99.9% of the VOCs from the system influent. An estimated 25% of the treated groundwater was used on the facility for non-potable process water and was ultimately discharged to the local sanitary sewer. Like most P&T systems, however, the vast majority of the extracted and treated groundwater was discharged to surface water under a Virginia Pollutant Discharge Elimination System (VPDES) permit. In 2001, ongoing environmental investigations at the facility detected the presence of perchlorate in the deep bedrock aquifer. Consequently, the P&T system was deactivated, as it had not been designed to treat perchlorate-impacted water. As a result of the system deactivation, the facility fully converted to the local public water supply for process needs.

Perchlorate is an inorganic ion of ammonium perchlorate (AP; NH_4ClO_4), the primary solid rocket propellant oxidizer used in the propulsion systems of military missile and rocket systems and NASA space systems. AP is relatively soluble in water, and when released into the environment dissociates into the ammonium cation (NH_4^+) and the perchlorate (ClO_4^-) anion. Once in solution, the perchlorate ion is readily flushed via infiltration through the vadose zone into groundwater aquifers. AP is a vigorous oxidizer at elevated temperatures; however, the perchlorate ion is considered a weak oxidizer at ambient environmental conditions. As a result, the perchlorate ion does not degrade rapidly and may persist in the environment despite the presence of numerous and plentiful natural electron donors. Naturally-occurring bacteria have been identified that can facilitate the biodegradation of perchlorate (Coates *et al.* 1999, Kastner *et al.* 2001) under anaerobic conditions to chloride, water, and carbon dioxide (Cox *et al.* 2000). Likewise, a great deal of research (*e.g.*, Major *et al.* 1995, Acree *et al.* 1997, Graves *et al.* 1997) has shown that naturally-occurring bacteria can degrade chlorinated VOCs under anaerobic conditions. The process of introducing suitable electron donor substrates such as acetate, vegetable oil, and methanol to stimulate these naturally-occurring bacteria and is termed *in situ* anaerobic bioremediation.

Northern Virginia, like much of the Mid-Atlantic U.S. region, has experienced significant drought conditions (DEQ 2002) along with an increase in groundwater demand due to population expansion. Consequently, both private and local, state, and federal governmental bodies have recently formed planning committees and task forces to study and address the effects of drought on public water supplies in areas with rapidly-increasing populations such as northern Virginia. As drought conditions continue and demands for public water sources increase, remedial system designs at polluted sites are expected to come under increased scrutiny for their ability not only to improve water quality, but also to preserve the quantity of water resources.

The deep bedrock aquifer at the facility consists of intensely-fractured, contact-metamorphosed sedimentary rocks including shales and siltstones. Previous pump testing and downhole geophysical logging of the deep aquifer indicated that the fracture frequency is sufficiently high to depths between 300 and 400 feet below ground surface (bgs) that the aquifer behaves as an equivalent porous medium (EPM) at the facility scale. Along the northern third of the facility, the aquifer subcrops and is under unconfined conditions. Through the southern two-thirds of the facility, however, the aquifer is overlain by a dense, sparsely-fractured diabase confining unit and is under confined conditions. In general, groundwater flows in a convergent pattern from the northern to the southern section of the facility under natural, non-pumping conditions (Figure 1).

The deep groundwater aquifer has been chemically characterized via multiple facility investigations and periodic groundwater sampling events. Historical and recent sampling data have shown perchlorate and VOC plumes centered in the north-central and central areas of the facility (*i.e.*, the treatment zone). A facility-wide sampling event conducted in May 2003 indicated perchlorate concentrations in the treatment zone ranging from 11.7 µg/L to 8,293 µg/L. The deep groundwater perchlorate plume extends approximately 2,250 feet in a direction parallel to the natural hydraulic gradient and to a maximum width of approximately 1,750 feet in the cross-gradient direction. During a sampling event conducted in November 2001, total VOC concentrations in the area historically treated for VOCs ranged from 1.0 µg/L to 327.9 µg/L. The smaller VOC plume lies within the horizontal extents of the perchlorate plume. Dissolve oxygen (DO) concentrations in the deep groundwater were also measured during recent sampling events, and ranged from a facility-wide average of 1.19 mg/L in November 2002 to 0.22 mg/L in May 2003. These relatively low DO concentrations are typical of deep semi-confined to confined aquifers (Langmuir 1997), and are a positive indication that the aquifer is naturally pre-conditioned for anaerobic bioremediation.

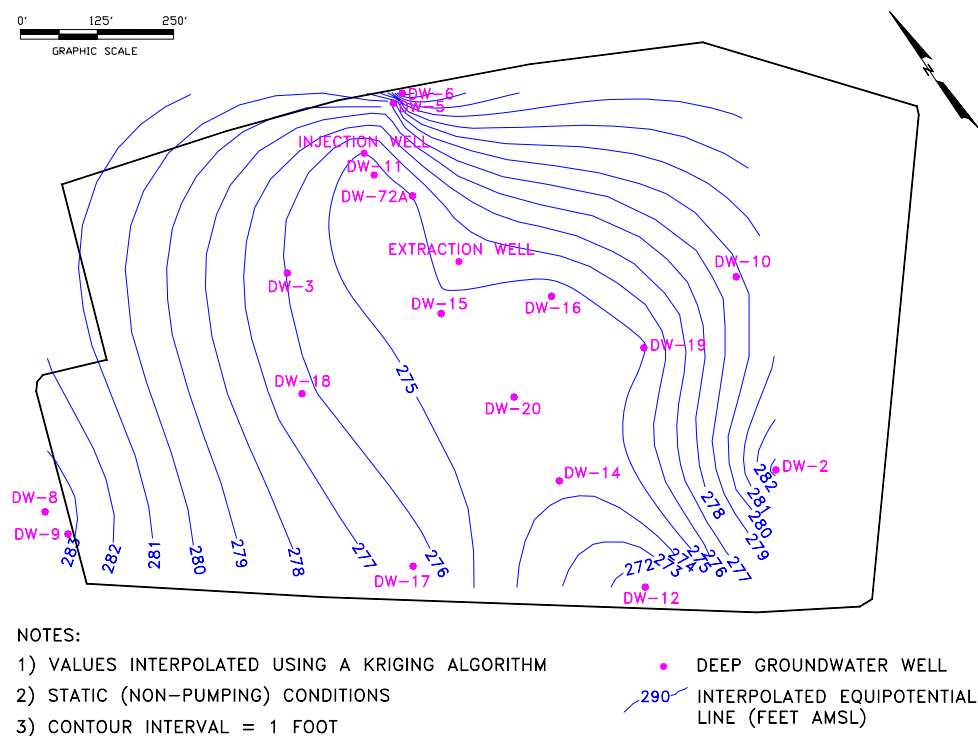


Figure 1. Potentiometric surface map prior to pilot system operation.

A pilot test was conducted from October 2002 to May 2003 to evaluate the potential effectiveness of the proposed RAB system. The initial configuration for the RAB system consisted of the existing deep groundwater extraction well, air stripping tower, and sediment filtration system along with a substrate amendment system and an existing deep groundwater extraction well converted to an injection well. An inflatable packer system was later installed in the injection well to facilitate pressurized injections (Figure 2). The existing GAC adsorption

system was not used because of historical mineral fouling issues in the P&T system. During the pilot test, extracted deep groundwater entered the air stripping tower, passed through the sediment filters, and was subsequently amended with electron donor substrate prior to reinjection into the deep aquifer via the injection well located upgradient of the extraction well.

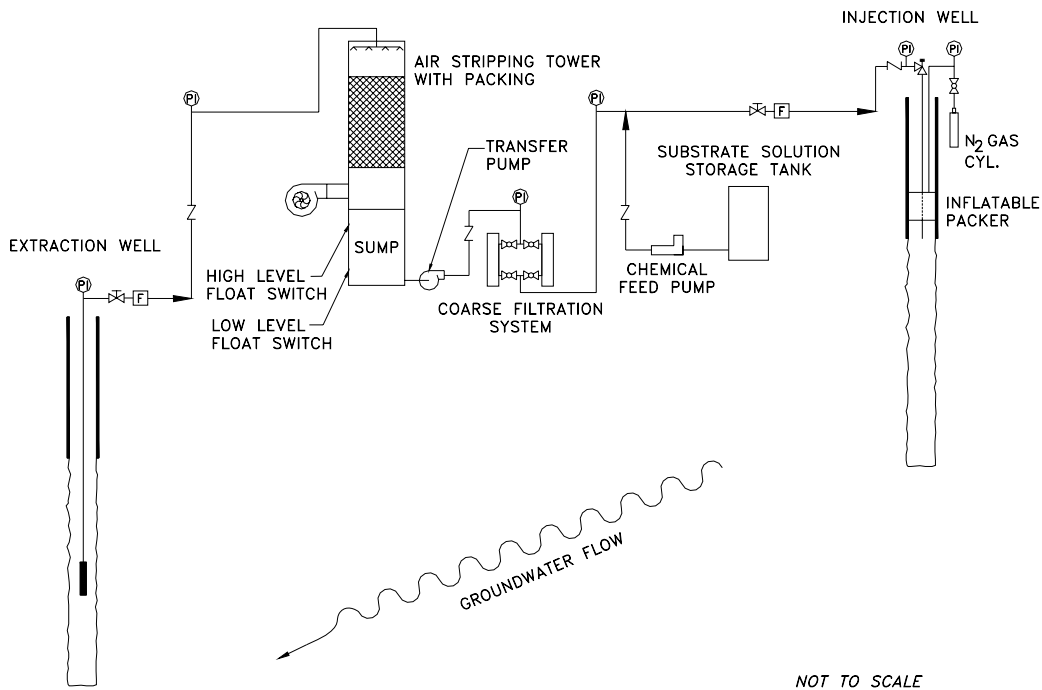


Figure 2. Simplified pilot system diagram.

MATERIALS AND METHODS

The pilot RAB system consisted of one extraction well and one injection well. These wells were selected based on their location within the core area of VOC and perchlorate impact and their historically-proven hydraulic connection to maintain hydraulic control of the VOC plume. An existing submersible pump installed in the extraction well was initially used to continuously pump groundwater through the treatment system. Extracted groundwater entered the existing air stripping tower to remove VOCs prior to reinjection at the injection well. A collection sump in the stripping tower equipped with low- and high-level switches controlled the cyclic operation of a 10-Hp centrifugal transfer pump. After the stripping tower was filled to a specified level, the transfer pump was activated, and the treated groundwater was pumped through the sediment filters. Following filtration, the treated groundwater was amended with electron donor substrate. A polyethylene tube fed by a chemical metering pump was connected to the transfer pump discharge piping. The metering pump was supplied by a 1,100-gallon plastic mixing tank containing 25% (by weight) calcium magnesium acetate (CMA) solution. Sodium iodide tracer was added to the CMA solution to create a 100 mg/L iodide concentration in the mixing tank. Following substrate addition, the treated and amended groundwater was reinjected into the deep

aquifer via the existing piping system and the converted injection well. The inflatable packer system installed at the injection well enabled the reinjection of groundwater at elevated pressures. A standard pressure relief valve rated for 125 pounds per square inch (psi) was installed at the injection well to ensure that potentially harmful backpressures were not maintained during injection cycles.

The initial extraction rate was set at 20 gpm during October 2002, and was increased to 30 gpm during January 2003. To maintain the system water balance, the extraction and injection rates were adjusted such that the injection rate was approximately twice the extraction rate. Existing flow meters were used to monitor the extraction and injection flow rates. After the extraction rate was increased during January 2003, the performance of the existing submersible pump was observed to steadily decline, likely due to the age of the pump. Prior to shutdown, the extraction rate was approximately 25 gpm. Extraction rates quickly decreased to less than 5 gpm during the first two weeks of May 2003, and as a result, the pilot system was deactivated in mid-May 2003. Backpressures at the injection well were observed to range from 30 to 68 psi during injection cycles and dissipated to background levels within two minutes after cessation of an injection cycle.

The RAB pilot test was conducted from October 2002 to May 2003, with approximately six weeks of downtime for system repairs and upgrades. At the conclusion of the pilot test in May 2003, approximately 3,132,000 gallons of water had been extracted, treated for VOCs, and reinjected. The substrate amendment system was operated for approximately two weeks in October 2002 and one week during November 2002, but was discontinued due to both mineral and bio-fouling in the injection well. In addition, during the modification of the existing P&T system, a significant amount of mineral scaling was observed to have accumulated inside the air stripping tower, transfer piping, and other system components since the late 1980s. This scaling was removed to the extent possible using mechanical methods. A file search and discussions with facility personnel revealed that the groundwater at the facility is naturally hard. Use of an electron donor such as CMA, therefore, would be expected to contribute to mineral scaling. As a result, geochemical sampling and modeling were used to evaluate the source(s) of the scaling and to evaluate the effects of the proposed electron donor substrate amendments on the process water.

Hydraulic monitoring of the RAB system was initiated just before system activation in October 2002. Monitoring of the system during startup periods (*e.g.*, beginning of pilot test, following system repairs and/or upgrades) occurred on an hourly to daily basis for a period of one to three days. Subsequent routine monitoring occurred on a weekly schedule. Routine hydraulic monitoring of the system included: 1) recording of flow rates for the extraction and injection wells using existing flow meters; 2) recording of backpressure at the injection well during injection cycles using existing pressure gauges; and 3) water level gauging at deep wells in and around the groundwater treatment zone using an electronic water level meter. In addition, a pressure transducer programmed to record water levels every 15 minutes was installed in a deep well located downgradient of the extraction well during November 2002.

The air stripping tower influent and effluent were sampled periodically during pilot system operation to monitor any changes in groundwater quality and to evaluate iodide tracer

movement. Samples were analyzed for perchlorate via EPA Method 314.0, for chlorate, acetate, and iodide via EPA Method 300.0B, and for VOCs via EPA Method 8260B.

RESULTS AND DISCUSSION

The results of the geochemical sampling and analysis and geochemical modeling of the RAB system indicated that the natural groundwater extracted from the deep aquifer, prior to air stripping, exhibits low potential for mineral scaling. However, vigorous aeration via the air stripping tower and subsequent CMA amendment would supersaturate the reinjection water with respect to numerous calcium-magnesium-carbonate and calcium/ferric-oxide minerals and could lead to an estimated 10 pounds per day of solids precipitation. Based on the information generated by the pilot test, the full scale application would require modification to the water treatment system and utilization of a different substrate to minimize/remove scaling issues.

Hydraulic head measurements collected during the pilot study showed significant drawdown in the core area of the facility surrounding the extraction well. The observed drawdown is especially significant considering the 14 inches of precipitation recorded during the pilot study period at the National Oceanic and Atmospheric Administration (NOAA) weather station at nearby Washington-Dulles International Airport. The RAB system was shown to be capable of producing a hydraulic capture zone encompassing the majority of the perchlorate and VOCs plumes at an extraction rate of only 30 gpm, as shown by the potentiometric surface map during pilot test operation (Figure 3). Comparing this to the 60 to 80 gpm extraction rates maintained during the operation of the historical P&T system, the new RAB system was able to provide hydraulic control of most of the perchlorate and VOC-impacted groundwater while pumping 50 to 63% less groundwater from the deep aquifer which was subsequently reinjected for no net loss to the aquifer. Furthermore, groundwater flow modeling of the deep aquifer performed at the end of the pilot study indicated that operation of the RAB at an extraction rate of 40 gpm would create a hydraulic capture zone large enough to control the entire VOC and perchlorate plumes.

Water samples collected during the pilot study also showed positive results. Influent samples exhibited perchlorate concentrations ranging from 6.24 mg/L in October 2002 to 1.84 mg/L in May 2003. This represents approximately 71% reduction in perchlorate concentrations at the extraction well after only eight months of system operation. The iodide tracer was not detected in the influent, possibly due to the low concentration of iodide (*i.e.*, 100 mg/L) in the CMA solution and/or the brief duration of the CMA/iodide tracer solution injection. However, acetate was detected in the influent sample collected during November 2002 at a concentration of 1.02 mg/L. This detection indicated that acetate had traveled from the injection well to the extraction well and therefore confirmed the hydraulic connection of these wells. Furthermore, the presence of acetate indicated that a substrate capable of stimulating the indigenous anaerobic microbial population was distributed within the targeted deep groundwater treatment zone.

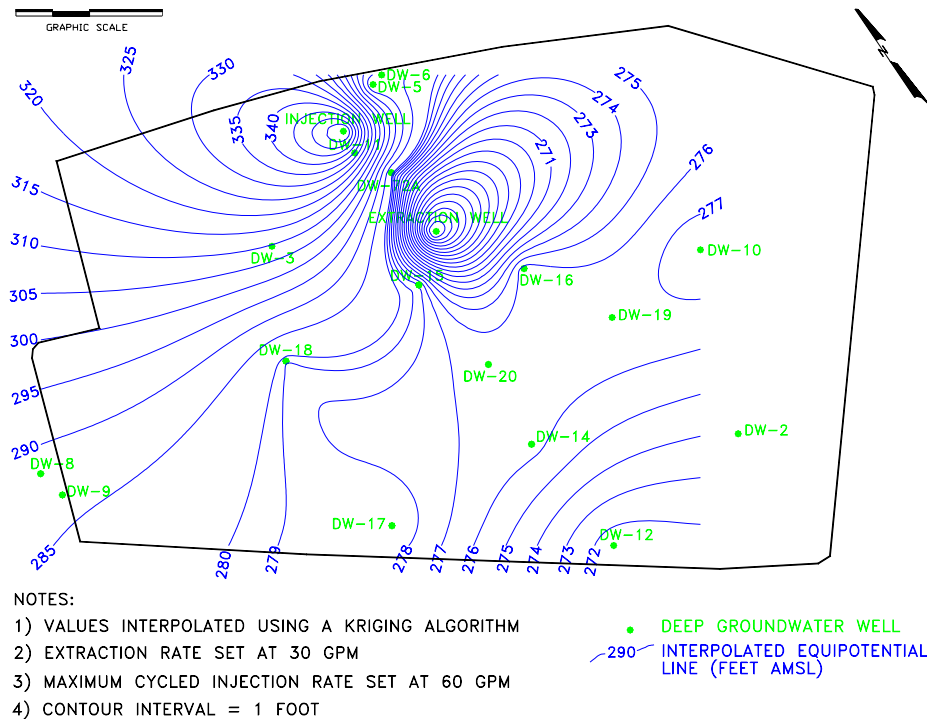


Figure 3. Potentiometric surface map during pilot system operation.

CONCLUSIONS AND FUTURE STUDIES

Based on the results of the pilot test, a modified RAB system design has been developed. This modified system will incorporate the existing extraction and injection wells used during the pilot test and will allow for system expansion to include additional extraction and injection wells. The modified RAB system consists of the existing extraction well, sediment filters, GAC adsorption system, the substrate delivery system, and the existing injection well fitted with an inflatable well-packer. To avoid aeration and consequent mineral scaling, the modified system will utilize the GAC adsorption system in lieu of the air-stripping tower for VOCs treatment. The modified system will utilize a more suitable water-soluble methanol substrate to replace CMA to further avoid the mineral scaling problem. Additionally, the centrifugal pump will be replaced with a more robust progressive-cavity pump capable of overcoming long-term backpressure to the injection well(s).

Overall, the results of the deep groundwater pilot test demonstrated the effectiveness of the converted P&T system to an *in situ* RAB technology. The benefits of this system design approach are threefold. First, by retro-fitting existing system equipment to a new technology and by reinjecting treated groundwater into the source aquifer, the system conserves economic resources associated with system construction and surface water discharge permitting and monitoring costs, respectively. Second, the system protects human health and the environment by reducing contaminant concentrations and by hydraulically controlling plume migration and therefore preventing possible impact to sensitive receptors. Finally, in contrast with typical P&T systems, which normally discharge all or most treated groundwater to surface water or

sanitary/storm sewer systems, the RAB system replaces 100% of the treated groundwater back to aquifer storage and therefore conserves the increasingly precious groundwater resource.

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**GEOCHEMICAL AND HYDROLOGIC INVESTIGATIONS OF GROUND WATER
AND SURFACE WATER INTERACTIONS AT MILLBROOK QUARRY, PRINCE
WILLIAM COUNTY, VIRGINIA – IMPLICATIONS FOR ZEBRA MUSSEL CONTROL**

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Dissolved major ion compositions and stable isotope ratios together with measurements of water balance parameters indicate a hydrologic connection between water that is impounded in Millbrook Quarry, an inactive stone quarry located in northern Virginia, and stream water in Broad Run. Under normal conditions, there is no surface water inflow or discharge from the quarry and the connection to Broad Run is via subsurface seepage. The rate of ground water exchange between the quarry and Broad Run is estimated to be 1.4 cfs (cubic feet per second), which represents approximately 3% of the average surface flow in this stream. Geochemical and stream flow measurements provide evidence that seepage from the quarry undergoes limited mixing with regional ground water before emerging down-gradient in sections of Broad Run. This is consistent with the geologic setting of the quarry, where south-trending geologic structural controls are likely to exert a strong influence on both ground water flow and stream channel morphology. The quarry is of concern to state environmental protection authorities following the discovery in August 2002 of zebra mussels that have colonized the water-filled pit. Broad Run is an important source of water supply to Lake Manassas and Occoquan Reservoir, which provides drinking water to over 1 million people in northern Virginia. Methods to control the spread of and/or eradicate the invasive mussels are currently being evaluated, and each of these alternatives may require further assessment of the potential impacts to ground water and surface water resources. The relatively direct ground water connection between the quarry and Broad Run also raises the possibility of a potential transport pathway for zebra mussel larvae to enter the stream environment.