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Proceedings

2007 Virginia / West Virginia Water Research Symposium

**“Connecting Management to
Aquatic Communities”**

November 26-30, 2007
Blacksburg, VA

Sponsored by



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Virginia Water Resources Research Center

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Session I-A

Protecting and Restoring Regional Waters

- Water Quality in Headwater Streams: A Test of Best Management Practices
- Reductions in Stormwater Quantity and Pollutant Loads Due to Bioretention and Structural Soil Practices
- A Stream Flow Restoration Project for the Potomac Headwaters

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WATER QUALITY IN HEADWATER STREAMS: A TEST OF BEST MANAGEMENT PRACTICES

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KEY WORDS: stormwater wet retention ponds, water quality, retention performance, macroinvertebrate community.

ABSTRACT

Urban stormwater runoff alters stream hydrology and carries many types of pollutants that affect both the diversity and abundance of aquatic organisms. Stormwater best management practices (BMPs) were developed to temporarily retain and treat urban runoff before entering nearby streams. Stormwater retention ponds are the most common type of BMP, with nearly 500 ponds throughout James City County, VA. Stormwater management, however, has outpaced the

science: the level of water quality improvement associated with retention ponds is for the most part unknown in this region. Our studies determining water and pollutant budgets for retention ponds during storms have documented 1) underperformance with respect to water retention regulations, 2) positive pollutant removal efficiencies for phosphate and nitrate+nitrite, and 3) negative pollutant removal efficiencies for total particulate phosphorus and ammonium contributing to fewer EPT taxa in the macroinvertebrate communities downstream. Retention pond performance can be improved with specific attention to both structural and functional design criteria.

INTRODUCTION

In recent years, the Lower Peninsula of Virginia has experienced rapid growth and development that show few signs of slowing down soon. While growth generally is considered good for the economy, the associated environmental impacts must also be considered. Increased development of land for commercial, industrial, and residential purposes means that more land is covered by impervious surfaces like roads, sidewalks, parking lots, and roofs. These impervious surfaces add to the quantity of stormwater runoff generated from rainfall that cannot infiltrate into the soil. Normally, rainwater slowly percolates through soil to the groundwater that over time discharges to a natural water course like a headwater stream. With development, storm events create pulses of stormwater runoff traveling overland to streams and rivers.

In addition to the increased quantity of water, there is also a change in the quality of stormwater runoff. Typically, in natural ecosystems soil microbes or plants remove particulate and dissolved constituents as water percolates slowly into the ground. However, in an urban watershed infiltration is blocked, and stormwater runoff occurs too rapidly for these biological processes to occur. Stormwater runoff from developing watersheds is therefore managed by approximating the natural runoff characteristics of the pre-developed site, primarily by controlling the timing and quality of stormwater discharge. In constructed stormwater retention ponds (a form of Best Management Practice, or BMP), water quality from developed sites improves when pollutants physically settle out of the water column before runoff discharges to a stream. Stormwater retention also allows for biological uptake of nutrients and other pollutants by the plants and the microbes in the pond. Hence, the longer the retention time in the BMP, then the more time particulates have to settle out of solution and nutrients have to be taken up.

Design requirements for stormwater retention ponds are based on the inference that retention time abates the quantity and improves the quality of runoff, but this has not been tested fully. The local regulations in James City County require that stormwater retention detain the 1-year, 24-hour storm event for 24 hours. The 1-year, 24-hour storm event in James City County amounts to 2.8 inches of rain in a 24-hour period. Therefore, a BMP should be able to accept the runoff generated by 2.8 inches of rainfall and retain it for at least 24 hours. Recent research at the College of William & Mary, however, suggests that BMPs often do not meet this requirement (Hancock and Popkin 2005). If retention time is inadequate, then significant water quality improvements are unlikely to be achieved. Further, because many of these ponds are constructed immediately adjacent to headwater streams, then water quality in those streams may be compromised.

Water quality encompasses many parameters that characterize the types and amounts of materials suspended or dissolved in the water, the levels of any of which may impact aquatic habitat downstream. High levels of suspended sediment, for example, can block light penetration and smother organisms living in the water. Elevated nutrient concentrations can stimulate nuisance algal blooms. The macroinvertebrate community in streams is usually diverse and abundant when water quality is unimpaired, but many species are intolerant of poor water quality (Lemly 1982, Barbour et al. 1999). Benthic community structure downstream of stormwater retention ponds has not been compared with streams not receiving discharge from a BMP.

The objectives of our research are to answer the questions: 1) Does the observed retention time of a stormwater retention pond match the engineered design? 2) Do stormwater retention ponds significantly improve the water quality of stormwater runoff? 3) Does discharge from BMPs affect downstream aquatic macroinvertebrate community structure? For this research, the ultimate goal is to determine the influence of stormwater retention ponds on the quantity of water retained, the quality of water released, and the impacts of that discharge on macroinvertebrate communities in headwater streams. Establishing the connection between stormwater retention ponds and their downstream impacts will help to better gauge the success of these best management practices.

METHODS

Three stormwater retention ponds have been selected for study in James City County, Virginia. These BMPs and their associated single family housing developments are located at the headwaters of first-order streams. The site names are Kensington Woods, Pointe at Jamestown, and Longhill Grove. Three reference headwater streams surrounded by forest and without retention ponds or associated development have been chosen for comparison. These reference streams are Pogonia stream in the College Woods of William & Mary, an unnamed tributary in York River State Park, and an unnamed tributary to Waller Mill Reservoir in Williamsburg, Virginia. For one full year the water quality of stormwater runoff entering and discharging from the three retention ponds is being collected and analyzed. The macroinvertebrate community is being sampled a total of three times in all six streams (spring 2007, fall 2007, and spring 2008).

To address the first objective regarding water quantity, monitoring stations were established to collect data on all storm flows into and out of the three BMPs. A continuous record of pond inflow, outflow, and storage is being obtained by using automated data loggers at each retention pond. A tipping bucket rain gauge collects rainfall and a pressure transducer with a staff gauge records the elevation of water in the ponds every five minutes. Elevation measurements are then converted to storage volumes from the data in the as-built engineering plans for each retention pond. Pond inflow is derived from the measured pond outflow and the change in storage volume ($Q_{in} = \delta V / \delta t + Q_{out}$). The pond elevation data combined with the discharge to elevation curves produces inflow-outflow hydrographs that reveal actual retention times in the ponds. Comparisons between seasonal storms and varying storm intensities are also possible.

To address the second objective regarding water quality, water samples are being collected at the inflow and outflow points of each pond during storm events. Automated ISCO[®] water samplers

are programmed to collect the water samples every 15-60 minutes, depending on the expected duration of the storm. The samplers collect up to 24 500-ml water samples during storm events. Immediately following the storm events the water samples are retrieved and placed on ice prior to analyses. The water quality parameters that are being measured are pH, conductivity, specific conductivity, total suspended solids, total particulate phosphorus, dissolved inorganic phosphorus and nitrogen, and fecal coliform bacteria.

All the water samples are treated identically. pH, conductivity, and specific conductivity are measured using calibrated water quality meters or probes. Total suspended solids are derived by weighing the filtered particulates, and nutrient concentrations in the water samples are determined using standard colorimetric analysis. First, each sample is passed through a pre-ashed, pre-weighed, 0.45- μm glass fiber filter to remove suspended sediment. The filter is then dried in a 60°C oven and re-weighed for determination of total suspended sediment, then ashed at 450°C for three hours and resuspended in 1N HCl for determination of total particulate phosphorus (Chambers and Fourqurean 1991). The filtrate is analyzed for dissolved inorganic phosphate, nitrate+nitrite, and ammonium using standard methods (Parsons et al. 1984). Fecal coliform bacteria counts are quantified using Coliscan[®] plates scored for the number of bacteria colonies after 48 h incubation.

To address the third objective regarding downstream impacts, the macroinvertebrate abundance and diversity in three reference streams is being compared to the macroinvertebrate abundance and diversity in the three streams receiving runoff from BMPs. Each stream is being sampled a total of three times (spring 2007, fall 2007, and spring 2008), and all the macroinvertebrate organisms found are being collected, preserved, and identified in the lab. A standard macroinvertebrate sampling protocol designed for the coastal plain is being used to sample all streams (Maxted et al. 2000). All organisms are being grouped into their respective family and counted. The following metrics established in the literature are being used to compare macroinvertebrate diversity between streams: taxa richness, EPT index, % Ephemeroptera, the modified Hilsenhoff index, and % clingers (Hilsenhoff 1987, Barbour et al. 1999, Maxted et al. 2000). Expected ranges for macroinvertebrate diversity in non-tidal coastal streams have also been established by the Virginia Department of Environmental Quality (VDEQ 2003). These expected ranges provide an additional reference comparison for area streams along with the actual reference stream counts.

RESULTS

The preliminary results presented here are for one of the retention pond sites, Kensington Woods. The site has 10.84 acres of drainage area with 4.25 acres of impervious surface expected at buildout. Figure 1 displays the centroid lag time between the inflow curve and outflow curve of the pond for storms during the past year. This centroid lag time is the time period between the midpoint of the inflow and outflow hydrographs and effectively refers to the retention time of stormwater runoff in the pond.

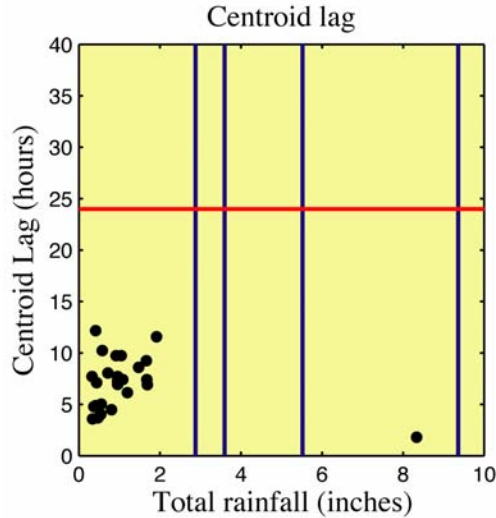


Figure 1. The centroid lag time between the inflow and outflow for storm events at Kensington Woods.

All but one storm event was less than the 1-year, 24-hour storm event. However, none of the storm events achieved the regulation time of 24 hours indicated by the red line in Figure 1.

Pollutant removal efficiencies in the Kensington Woods retention pond were variable. Figure 2 shows the cumulative mass of phosphate, total phosphorus, nitrate+nitrite, and ammonium in the inflow and outflow for a selected storm event at Kensington Woods.

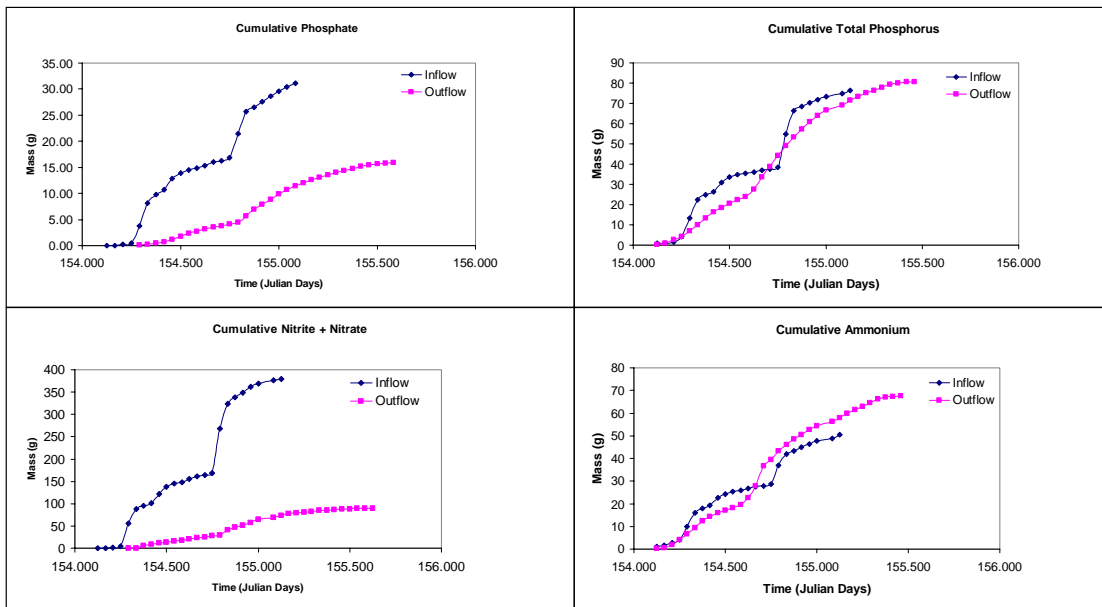


Figure 2. Cumulative mass for phosphate, total phosphorus, nitrate+nitrite, and ammonium in the inflow and outflow from a storm event on June 3, 2006 at Kensington Woods.

The panel for total phosphorus and ammonium show that the cumulative mass in the pond outflow is greater than the inflow indicating that the pond was a net exporter of these pollutants. Conversely, the panels for phosphate and nitrate+nitrite show that the retention removed a substantial portion of these pollutants from the stormwater runoff.

The macroinvertebrate collections made in the spring of 2007 did not show significant differences in taxa richness ($P=0.10$). Figure 3 shows the macroinvertebrate taxa richness at pond sites versus reference sites.

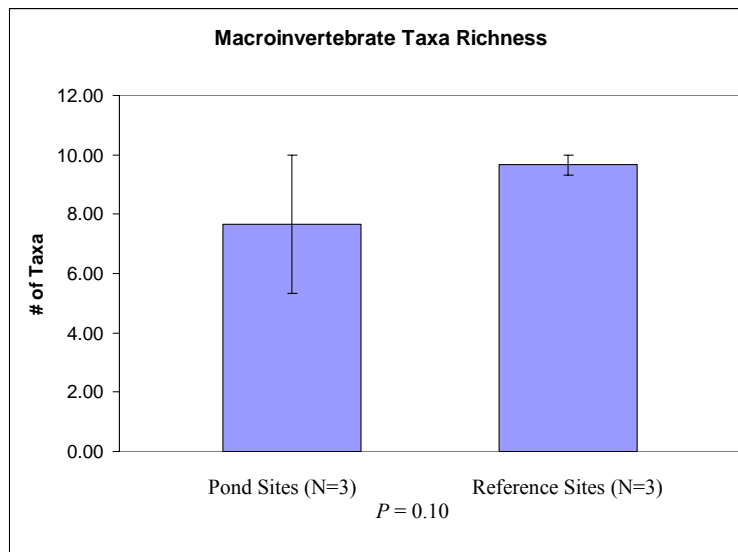


Figure 3. Macroinvertebrate taxa richness in headwater streams at pond sites versus reference sites.

However, the number of EPT taxa did significantly differ from pond sites (0.333 ± 0.577) versus reference sites (2.67 ± 1.15) $P=0.035$. Additional fall and spring collections will be made to supplement these data and additional analysis using the metrics discussed in the methods are being applied. Storm events are also still being monitored at the sites for retention performance and pollutant removal.

DISCUSSION

The preliminary results presented for the Kensington Woods site demonstrate the underperformance of stormwater retention ponds that is consistent among the monitored sites in James City County. The initial hypothesis that stormwater retention ponds are not performing as intended is evident starting with the poor retention performance that does not approach the required retention time of 24 hours for the 1-year 24-hour storm event. Preliminary measurements of the retention pond determined that the Kensington Woods pond is slightly undersized, which could partly explain the pond's shortfall (*unpublished data* Bonnette & Hancock, 2007). However, the consistent underperformance for even small storm events that do not approach the design storm point to a more fundamental design flaw. The actual inflows and outflows do not reflect what is predicted in the engineered plans for the pond. The higher than predicted inflow suggests that runoff coefficients should be adjusted upward to match observed field conditions. These findings are even more disturbing considering that the Kensington

Woods development is approximately only half complete. When the development is complete even more runoff will overwhelm the retention pond.

The deficiency of the Kensington Woods retention pond with regards to retention performance must also have an effect on the ability of the pond to remove pollutants. The principal concept in stormwater quality improvement is that slowing the speed of the runoff and giving it time to infiltrate or undergo biological reactions will remove pollutants. Therefore, if significant water retention is not occurring, then the ability for water quality improvement to occur must also be reduced. The results displayed in Figure 2 indicate that approximately half of the cumulative mass of phosphate was removed from the runoff by the pond. Much of this phosphate was likely bound to sediment deposited in the pond. Nitrate+nitrite was removed in even greater quantities which is likely due to it being a limiting nutrient in most aquatic environments. However, total particulate phosphorus, which includes organic forms, had a higher cumulative mass exiting the pond than what entered. A possible explanation for this could be organic matter in the form of algae or other microorganisms flourish in the pond due to excess nutrients provided by runoff that are then flushed out during a storm event. Similarly, ammonium exits the pond in greater quantities than it enters suggesting that ammonification or the breakdown of organic matter by microorganisms is occurring. Clearly biological processes are occurring in the pond to produce the observed nutrient transformations, but they may not be occurring at the rates or in the directions that would necessarily improve water quality.

Finally, the diversity of macroinvertebrate communities observed in streams has been shown in numerous studies to correlate to the amount of urbanization (Shaver et al. 1995, Roy et al. 2003). The response of these organisms to increased stormwater runoff and poor water quality may tell us the most about the impacts of retention ponds. On the surface, it seems that overall taxa diversity is not significantly affected in streams receiving retention pond runoff compared to reference streams. However, the number of EPT taxa suggests otherwise. Pending further analysis of the macroinvertebrate organisms collected and examining other commonly employed metrics may help to elucidate the impacts.

If we want to continue to use stormwater retention ponds as a best management practice, then we must evaluate BMP effectiveness and identify deficiencies that can be improved. The principal concern associated with permitting new development is managing the quantity of stormwater, but there are water quality issues and stream community impacts, too. Together, all of these processes are poorly understood with regard to stormwater runoff and retention ponds. The measured performance of stormwater retention ponds can guide future policy on managing runoff and protecting our water resources. Starting locally at the headwaters of streams where these retention ponds are commonly located makes the most practical sense as we work towards regional improvement of the Chesapeake Bay. The 2006 annual report on the state of the Chesapeake Bay indicates that its health is still degraded, and urban runoff of sediment and nutrients is one of the main areas where pollution reduction is still needed (Chesapeake Bay Program 2006). Since many stormwater BMPs are located at the headwaters of streams, this is where we must begin to address our water quality issues.

Overall, the significance of this research is that it will provide much needed information to concerned citizens, planners, government leaders, and the development community on the

effectiveness of stormwater retention ponds as a best management practice. Linking detailed retention pond performance data to the benthic aquatic community will address the poorly understood impacts of best management practices. Our goal is provide detailed recommendations on effective BMP design to improve future stormwater management.

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REDUCTIONS IN STORMWATER QUANTITY AND POLLUTANT LOADS DUE TO BIORETENTION AND STRUCTURAL SOIL PRACTICES

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ABSTRACT

Increases in impervious surfaces associated with urbanization change stream hydrology by increasing peak flows, stream flashiness and flood frequency, and degrade water quality through increases in sediment and nutrient concentrations. Stroubles Creek, a tributary of the New River, was listed on the 1998 Virginia 303(d) list for impairment of the benthic macroinvertebrate community. Construction and streambank erosion, along with organic matter and nutrients, were cited as significant stressors. Extensive streambank erosion in the downstream sections of Stroubles Creek is occurring due to increased flows from extensive urban development in the headwaters. In response to water quality and quantity issues within the Stroubles Creek watershed, the Town of Blacksburg and Virginia Tech are implementing innovative stormwater best management practices (BMPs) to reduce stormwater runoff and nutrient loading. The goal of this project is to evaluate the effectiveness of two low impact development (LID) BMPs: a bioretention cell and a structural soil infiltration trench. BMP construction was completed in July 2007. Stormwater monitoring, consisting of stormwater runoff, nutrient, sediment, and bacteria concentrations, and temperature, is ongoing. Study results will be used to develop design standards and to estimate runoff volume and pollutant load reductions for water quality management.

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A STREAM FLOW RESTORATION PROJECT FOR THE POTOMAC HEADWATERS

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KEY WORDS: alluvial groundwater, piezometer, beaver

ABSTRACT

Concerns over both quantity and quality of our surface and ground water resources are widespread in the United States. This project is exploring the feasibility of increasing surface water flow and enhancing alluvial groundwater storage by installing a series of small, in-stream structures in two West Virginia headwater watersheds. The structures used were similar to the cross vanes common in natural stream restoration. Cross vanes were chosen because they accomplished the primary purpose of raising the level of water behind the structure, without inhibiting the movement of aquatic life or destabilizing the stream channel. Primary testable hypotheses in this project were: the structures will elevate the local alluvial water table; the structures will increase surface flow in local streams during base flow periods; and the structures could be constructed at low cost, require little maintenance, and not destabilize the stream. Conceptually, the project represents an ecologically guided attempt to partially restore the natural hydrologic functions, flow regimes and ground water levels that likely existed prior to the colonial era extermination of beavers - and the elimination of once ubiquitous beaver dams on headwater streams. This paper provides an overview of the project design and results to date.

INTRODUCTION

Concerns over both quantity and quality of our surface and ground water resources are widespread in the United States. In the Potomac Headwaters region of West Virginia, these issues are exacerbated by economic development in the absence of effective community planning, and a large integrated poultry industry that utilizes significant amounts of groundwater. An assessment of water resources in Hardy County, WV found that the poultry industry “may impart a significant burden” on the county’s groundwater resources (MSES, 2002). Meanwhile, the population of Hardy County is expected to grow by 17% over the next two decades, with much of that growth in the region dependant on groundwater for water sources.

There is anecdotal concern in this area that “we are sucking the rivers dry” and recent experience indicates that new wells must be drilled deeper to get the yields previously obtained from shallower wells. This project is exploring the feasibility of increasing surface water flow and enhancing alluvial groundwater storage using a technique tested in the West and modeled after the hydrological impact of beaver dams.

When we imagine the eastern-American landscape of 300 years ago, most people picture a landscape of nearly continuous forests bisected by free flowing streams. The reality was very

different. Before the fur trade decimated beaver populations in early colonial times, beavers were abundant on streams throughout the Mid-Atlantic region (Ambrose 1996). The wetlands created by beavers supported diverse flora and fauna and may have maintained perennial flows in streams in headwater regions. During the time when beavers were extirpated from the Chesapeake basin, contemporary maps show many perennial streams gradually became intermittent or ephemeral, possibly due to lowering of water tables as beaver dams were no longer present to recharge groundwater. Although other human disturbance, including channelization, deforestation and other activities might also have lowered groundwater tables, current research in the West affirms that the return of beavers can result in a return of perennial flows to headwaters areas and a more natural stream flow regime (Clark, 1998), and can augment alluvial groundwater levels a considerable distance downstream of beaver dams (Westbrook et al, 2006).

It is currently unrealistic to reintroduce beavers for the purpose of restoring natural hydrologic conditions within Eastern watersheds, in large part due to the public's often negative reaction to beaver activity. However, researchers in the western US have investigated low-cost methods to construct beaver dam-like structures (BDS) to restore riparian areas and increase groundwater storage (Skinner et al, 1988; Skinner et al., 1991; Clark et al., 1998; Warren, 1991). These BDS have been built using a variety of materials, such as: woven wire, steel posts, synthetic erosion mat and discarded tires (Skinner et al., 1991); logs (Warren (1991); and woody brush (Norton et al., 2002). By retaining eroded material behind the structure, the BDSs reduce downstream sedimentation and augment alluvial water storage capacity as they mature.

With respect to stream flow, Stabler (1985) reviewed the literature on increasing summer flow in small streams, citing a number of cases where a series of erosion control check dams and gabions were constructed in ephemeral flow gullies, and perennial flow unexpectedly developed over time. Zuni Indians have been installing structures made from woody brush for the same purpose for over 2000 years (Norton et al., 2002). Stabler hypothesized that small dam construction within small valleys can restore stream flow by increasing the zone of saturation within the valley bottom and in sediments trapped behind the structure; the water thus stored is slowly released through the ground back to the stream channel. Groundwater levels in alluvial aquifers fluctuate with stream stage (Workman & Serrano, 1999; Chen and Chen, 2003). These structures serve to increase effective stream stage along streams segments.

In this continuing project, we are assessing the effects of in-stream structures on surface flow and alluvial ground water levels of headwater streams in the Potomac Headwaters region of West Virginia. Primary testable hypotheses in this project remain: the structures will elevate the local alluvial water table; the structures will increase surface flow in local streams during base flow periods; and structures can be constructed at low cost, require little maintenance, and not destabilize the stream. The structures represent an ecologically guided attempt to partially restore the hydrological effects of what was once a ubiquitous component of the ecosystem – beaver dams.

METHODS

This study used a paired watershed design, with a period of baseline data collection. Data analysis was based on comparisons between sites, (in particular, between control and experimental sites), between upstream and downstream locations within the experimental study areas, and between pre and post treatment data for individual sites. Also, digital pictures provide a visual record of changes.

Data collected for this project include flow, groundwater level, stream height, precipitation, and water temperature. Data analysis was based on comparisons between sites, in particular, between control and experimental sites, and between upstream and downstream locations within the experimental study areas.

Flow and precipitation data collection began in the fall of 2003. Precipitation data was collected using All Weather Rain Gauges (Productive Alternatives, Inc.) that were placed at all of the study sites, with cumulative precipitation data collected on sampling days. An additional gauge, located within a few miles of all the study sites, was checked daily.

Stream flow sites were located at the top and bottom of each study site. The very small, meandering streams selected for this project provided particular challenges for accurate measurement of stream flow across all ranges of flow, and a mixture of methods suitable for different conditions were utilized. Flow measurement equipment included a Global Flow Probe Model FP101 and a portable "Insta-Weir." The custom designed Insta-Weir was created for use by this project to measure flow where stream depth or flow rate precluded use of the Global Flow Probe or other flow measurement devices. The Insta-Weir ultimately proved unsuitable

Alluvial groundwater was measured using a network of piezometers. Piezometers were constructed from 1¼ inch PVC pipe, with nylon mesh is placed on one end of the pipe and secured in place using a 1¼ inch PVC cap which is perforated with six 1/4 inch holes. Spring steel measuring tapes were cut to length and placed in the piezometers and a water-soluble ink daubed on the tapes served as a crest height recorder between direct measurements. Piezometers were installed according to two protocols. The first was a longitudinal network spaced every 100 ft along the length of the stream segment, 10 ft from the edge of the stream (if possible), with the bottom of the piezometer level with the stream's thalweg. Piezometer nests were installed every 400-ft of stream length (where possible). The second piezometer protocol consists of a grid of three rows of three piezometers (if possible) used to measure changes in groundwater levels across a width of floodplain area caused by the installation of a structure. Stream height was measured relative to the top of piezometers using a hand sight level and a pocket rod.

Structures were designed with three key attributes in mind: use of esthetically acceptable materials, stability, and low cost. Structures were modeled after a variant of the cross vane that we observed during a visit to a stream restoration project in Big Bear, PA. Log structures at that site had been installed on a sizeable, high-gradient stream by two men in one day, using no heavy equipment. These structures have successfully withstood a number of major floods. Structures were built to or below bankfull elevation at the edges and lower in the center of the stream to provide a spillway. The cross vane's inverted-V design directs the force of the water

away from the banks and toward the center of the channel, reducing bank erosion and enhancing long-term stability.

A schematic of the cross vane used for this project is shown in Figure 1. While construction methods at each site varied, in general the method was as follows: 1) The primary members of the cross vane were driven or cut into the stream banks at the appropriate angle and elevation. 2) The upstream edges of the logs were trimmed and fitted to each other, and then either wired, through-bolted, or nailed together. 3) At the point where the cross vane members enter the bank, model 68-DB-1 duckbill earth anchors (1,100 lbs. holding capacity) were driven into the bank at a right angle to the member, with the cable fastened to the member using a cable clamp. 4) Galvanized wire mesh and erosion cloth were stapled to the upstream edge of the exposed portions of each member, and run 1'-2' upstream of the structure. The erosion cloth was included to reduce seepage through the structure, and is held below the line of sight for aesthetic reasons. 5) Cobble, gravel and sand were placed on the full length of the wire mesh upstream to the height of the top of the members.

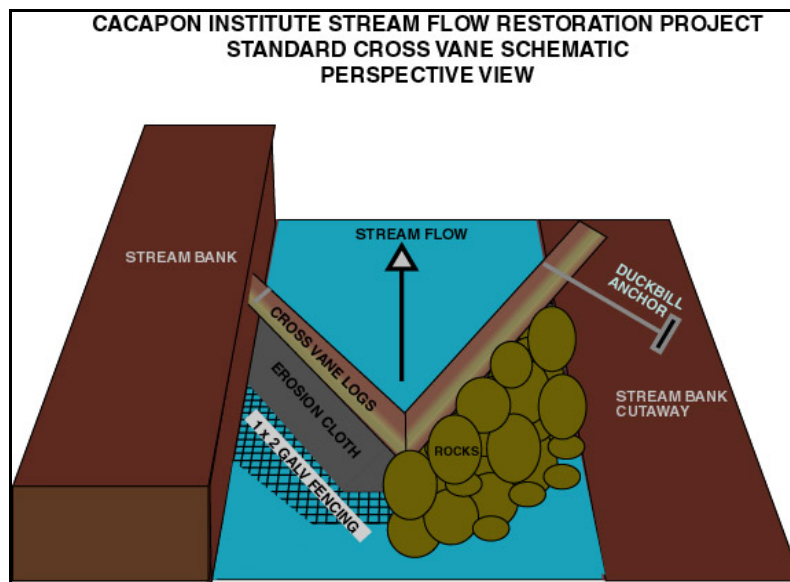


Figure 1. Schematic of structure installation.

Structures were installed longitudinally in series in the experimental streams to create a progression of small pools, rather than single isolated structures. We anticipated that alluvial water storage caused by upstream pools will extend periods of flow to pools further downstream and create a multiplier effect, like batteries wired in series.

The grant time frame constrained the collection of pre-treatment data; however, the paired watershed and upstream/downstream designs were intended to compensate for this limitation to the extent possible. Once suitable sites were selected, data collection began. Sites were visited at least twice monthly throughout the period of study. Frequency of visits was increased at times to capture changing conditions.

RESULTS

Four study sites were selected. **Site 1** (referred to as "the Meadow" below) is a broad valley with a sinuous stream and extensive wetlands. There are no roads or agriculture in the floodplain. The site's stream length is 2100 feet, with a slope of 1.7%; the watershed area at the base is 511 acres. Twenty-two "regular" piezometers were installed along the length of this stream, as well as six piezometer nests and two structure grids with a total of 20 piezometers. Fifteen structures were installed here. The results will focus on this site.

Site 2 is located roughly 2000 feet upstream of Site 1, is totally forested, with pasture and a highway at the headwaters of the drainage. The stream section is 850 feet long with a slope of 2.7%; the watershed area at the base is 309 acres. Ten piezometers were installed along the length of this stream. Structures were not installed here after it became apparent that this stream was unstable. Data from this site was considered non-treatment data.

Site 3 is the "control" site. It has only a small, forested section and is mostly grassy. The site was far from ideal as a control, but no other stream in the area was more suitable. It winds through a narrow valley, and passes back and forth across a road. The stream has a slope of 3.2%; the watershed area at the base is 342 acres. Eight "regular" piezometers were installed along the length of this stream, as well as three piezometer nests. There is a pond near the headwaters of this stream - a pond that has in the past blown out and created scouring downstream flows. In fact, during the study, the stream experienced several severe scouring events that lowered the thalweg by more than six inches in many sections.

Site 4 is a tiny stream in an upland meadow. The land is enrolled in the USDA Conservation Reserve and Enhancement Program (CREP). The stream has a slope of 2.4% and a narrow area with alluvial deposits; the watershed area at the base is 119 acres. Eleven "regular" piezometers were installed along the length of this stream, as well as three piezometer nests and two structure grids with a total of 14 piezometers. Ten stone structures were installed. This site has a side stream, about 400 feet above the bottom of the study reach, that can deliver significant flow to the system. There have been many problems with this site, including the owner's decision to install a culvert stream crossing in the middle of the project, and the fact that the stream is so small that slumping grass can create numerous small dams during the fall and winter.

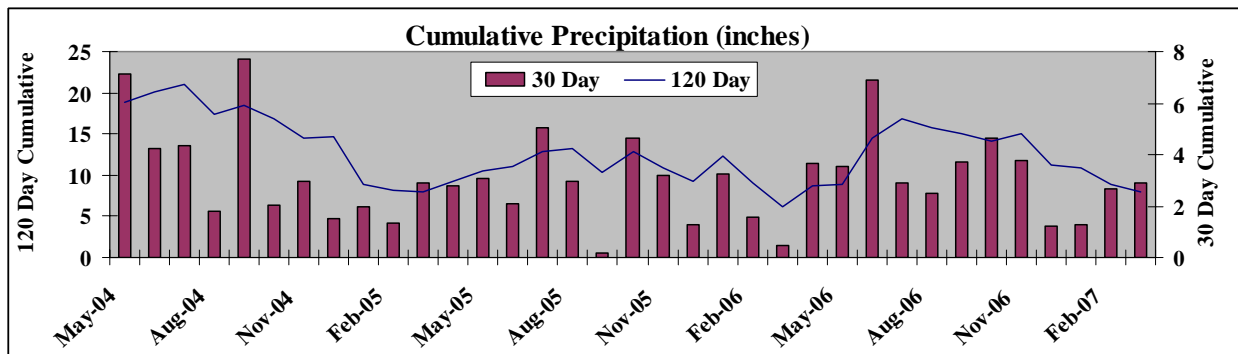


Figure 2. Cumulative precipitation.

Precipitation

Cumulative precipitation data is provided in Figure 2 to provide a measure of antecedent moisture conditions throughout the study period. The pretreatment period through September 2004 was very wet in comparison to most of the post treatment period. This unfortunate fact had important ramifications on data analysis. For example, one of the testable hypotheses of this project was that the anticipated increase in alluvial groundwater due the structure effects would in turn augment flows during low flow periods. Comparisons of pretreatment and post treatment flow data are required to determine if this occurred. However, because the pre treatment period was very wet, there were few opportunities to collect low water stream flows.

An interval of very low precipitation in August 2004 provided the opportunity to document groundwater levels as the subject streams became dry, and repeated periods of high flow to low flow in September due to tropical storm precipitation allowed documentation of each system's response to these events. Collection of these data series led to the decision to begin installing structures in October 2004. Fifteen structures were installed in the meadow section of Site 1 in October and early November 2004. The following results focus on that site.

Flow

Strong statistical relationships between control and experimental, and between upstream and downstream flow sites were required to assess changes in flow that might occur due to structure installation. Strong correlations were found between the four flow stations in Sites 1 and 2, with r-values ranging from 0.940 to 0.997. These sites were all also strongly correlated with the two flow stations in the control area (Site 3), with r-values ranging from 0.88 to 0.98, and very strongly correlated with the upper two flow stations at Site 4, with r-values ranging from 0.935 to 0.994.

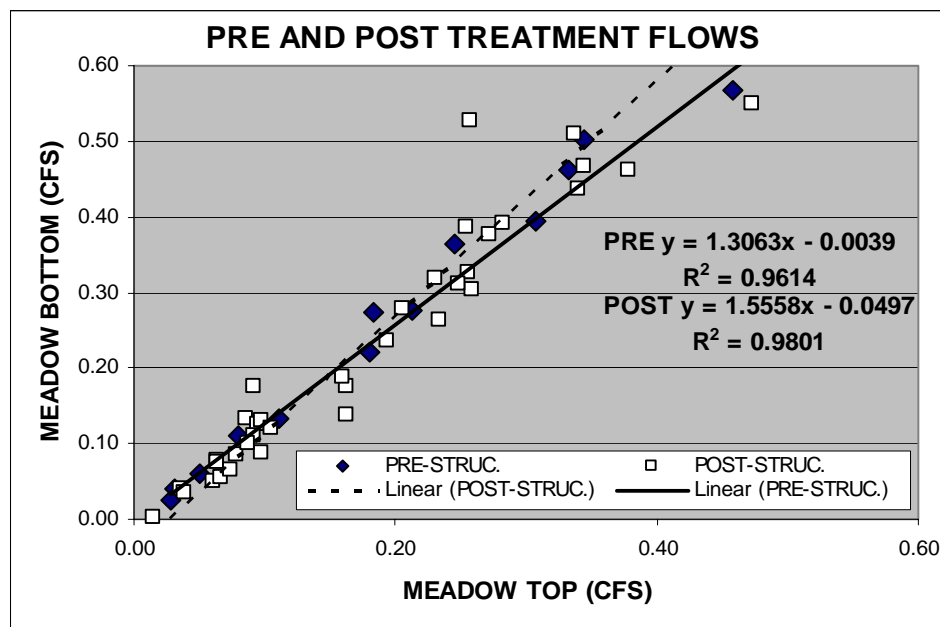


Figure 3. Stream flows below 0.6 cfs for the two flow stations in the Meadow (Site 1).

Figure 3 graphs stream flows below 0.6 cfs for the two flow stations in the Meadow (Site 1). Meadow Top is above all of the structures, Meadow Bottom is below all of the structures. No difference between pre and post treatment periods is apparent. Graphs of the Meadow Bottom against other control sites in Site 2, Site 3 and Site 4 indicate that flows below the treatment section may be slightly, but not significantly, lower during the post treatment period. However, as noted previously, the lack of low flow data during the pre treatment period is problematic.

Groundwater data

The piezometer data was far less predictable than the flow data discussed above, and impossible to generalize. The one pattern that was reasonably consistent was that most piezometers had significantly higher water levels during the dormant season (October through April) than the growing season (May through September).

Otherwise, four major patterns in piezometer data were observed. Piezometers that: 1) usually had water, with little difference between daily and maximum levels and a relatively small range of water levels; 2) usually had water, with little difference between daily and maximum levels and a relatively large range of water levels; 3) were often dry, with large difference between daily and maximum levels and a relatively large range of water levels; and 4) were in a condition of dynamic change that had nothing whatsoever to do with the installation of structures (may include the defining characteristics of 1-3 above). The differences in patterns 1-3 appear to relate to the speed of the hydraulic connection to the stream, and piezometers in proximity to one another often "behaved" quite differently. For example, Figure 4 is a time series graph of two piezometers located fifty feet apart. Both are at the tow of the slope at the edge of the floodplain farthest from the stream. This represents an extreme example of piezometer variability.

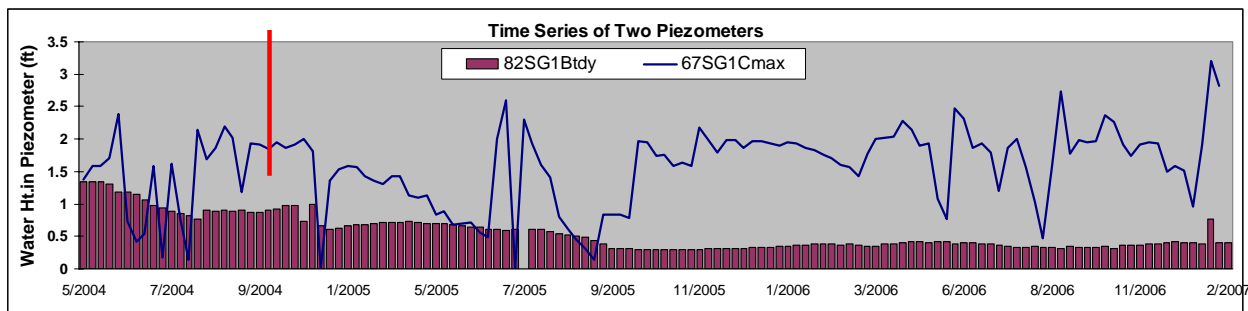


Figure 4. Time series graph of water height in two piezometers at the Meadow site. The site represented by the bars was located 50 feet down gradient of the site represented by the line.

The presentation will provide the results of analyses on the alluvial groundwater data. Some of this analyses explored individual piezometer behavior. For example, analysis of variance was used to compare water levels across seasons over time. Correlation analysis within season (growing and dormant) and year was used to identify piezometers that behaved in a similar manner. Cluster analysis was used on the correlation matrices to find piezometer groupings, and GIS was used to map similar clusters, and visualize how groupings changed over time.

There were strong indications of groundwater response to structure installation at a number of locations. However, anomalies at control sites cloud the issue. For example, the control stream had log jams from which acted like structures by increasing the effective stage height, thereby increasing connectivity to the floodplain in the vicinity and, apparently, raising water in nearby and downstream piezometers. The control stream also had a number of scouring events that dropped much of the channel by more than six inches, thus reducing its connection to the floodplain. This was not helpful.

SUMMARY

This is a continuing project. Data collection will continue for at least one more year. Adequate surface and groundwater supplies are a prerequisite for sustainable human communities and healthy wildlife habitat. While the scope of this project is small, it will provide important information concerning the feasibility of restoring watershed hydrologic functions and augmenting groundwater storage in a way that is low-cost, easily replicable, environmentally sensitive and culturally acceptable.

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Session I-B

Gaining Insight into Microbial Communities in Aquatic Systems

- Characterization of Phytopathogenic Bacteria from Irrigation Reservoirs at Two Commercial Nurseries in Eastern Virginia
- Bacterial Communities from a Headwater Stream at an Abandoned Arsenic Mine
- Effects of the Antimicrobial Agent Triclosan on Bacterial Resistance to Chlorination in Wastewater Treatment Processes
- Detection of Norovirus in Sewage Effluent and Its Persistence in Estuarine Water Using Real-time PCR

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CHARACTERIZATION OF PHYTOPATHOGENIC BACTERIA FROM IRRIGATION RESERVOIRS IN EASTERN VIRGINIA

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KEY WORDS: phytopathogenic bacteria, irrigation water, polymerase chain reaction, 16S rRNA, sequence analysis, sedum, iris.

ABSTRACT

Several bacterial species were isolated from irrigation reservoirs at the commercial ornamental plant nurseries in eastern Virginia by streaking water sample on nutrient agar and incubating at 28°C for 36-48 h. Resultant bacterial colonies first were differentiated into different types based on morphological characters. Representative cultures then were subjected to Gram's reaction by potassium hydroxide solubility test and pathogenicity tests by detached leaf bioassay on *Sedum* and *Iris* plant leaves. Several types of bacteria isolated from irrigation reservoirs were Gram negative, and they were positive for pathogenicity tests. These bacteria are plant pathogenic in nature. The cultures were further subjected to biochemical characterization tests including, Arginine dihydrolase test, Gelatin hydrolysis test, Levan formation, Lipase activity, Catalase activity, Kovac's oxidase test and tobacco hypersensitive reaction. One pathogenic bacterial culture was subjected to FAME analysis. Further, isolation and amplification of purified culture DNA by polymerase chain reaction at 16S ribosomal RNA regions followed by sequence analysis was performed. Morphological characters, biochemical characterization tests, FAME analysis and sequence analysis confirmed the pathogenic bacterial identity as *Pseudomonas*

syringae and *Erwinia carotovora*. The threat to irrigation water health consequently to nursery crop health posed by these bacterial pathogens will be discussed.

INTRODUCTION

Bacteria live all around us and within us. The air is filled with bacteria, and they have even entered outer space in spacecraft. Bacteria live in the deepest parts of the ocean and deep within Earth. They are in the soil, in our food, and on plants and animals. Even our bodies are home to many different kinds of bacteria. Our lives are closely intertwined with theirs, and the health of our planet depends very much on their activities. Bacteria play a major role in recycling many chemical elements and chemical compounds in nature. Without such bacterial activities as the recycling of carbon dioxide (CO₂) life on Earth would be impossible. Plants use CO₂ to grow and in the process they produce the oxygen humans and other animals breathe. Moreover, we would drown in garbage and wastes if bacteria did not speed the decomposition of dead plant and animal matter.

Plant diseases reduce yields of food and cash crops, mar the beauty of the ornamental plants, and reduce our ability to shelter and feed ourselves. In some cases, pathogen can produce toxic compounds that poison our food. Plant diseases limit where crops can be grown and determine what foods are available to us (Schumann and D'Arcy 2006). The major group of microbes, which causes plant diseases, includes fungi, bacteria, viruses and nematodes. Bacterial group of microbes are tiny single celled prokaryotes, which can devastate the crop plants by infecting them and causing the diseases.

One factor common to all crop production is the need of irrigation water. Plant pathogens in irrigation water were recognized early in the last century as a significant crop health issue. This issue has increased greatly in scope and degree of impact since that time and it will continue to be a problem as agriculture increasingly depends on the use of recycled water. Over the past several years, numerous studies have been conducted on phytopathogenic bacteria within water bodies such as rivers, lakes and irrigation water reservoirs. There is substantial evidence demonstrating that contaminated irrigation water is a primary, if not the sole, source of inoculum for many plant diseases of nursery, fruit, and vegetable crops. Most of these studies have dealt with relatively clean water sources, often in relationship to potato production. These findings pose great challenges and opportunities to the plant pathology community. Aquatic ecology of plant pathogens is an emerging field of research that holds great promise for developing ecologically based water decontamination and other strategies of pathogen mitigation. Pathogen detection and monitoring as well as biological and economic thresholds are much-needed IPM tools and should be priorities of future research. The problem of plant pathogens in irrigation water is aggravated by the increasing use of recycled water. Most nursery growers are familiar with foul, fishy smell of the volatile compounds released by bacteria such as *Erwinia* spp. while breaking down the plant tissues (Norman *et al.* 2003). Due to rapid population growth of cities, there are direct competitions with agriculture for the limited clean water supply. Surface water is considered to be a possible alternative for ornamental plant producers. Teaming with hydrologists, agricultural engineers, ecologists, geneticists, economists, statisticians, and farmers

is essential to effectively attack such a complex issue of growing global importance (Hong and Moorman 2005).

As we had observed some of the bacterial disease symptoms on ornamental plants – sedum and iris, during our routine visits to nurseries prompted us to take up these studies with the objective of isolation and characterization of phytopathogenic bacteria from the irrigation reservoirs of eastern Virginia.

MATERIALS AND METHODS

Collection of water samples from the irrigation reservoirs

Water samples were collected randomly in the irrigation reservoirs at 10 different spots and pooled them. Samples were stored in the refrigerated conditions until used.

Isolation and characterization of phytopathogenic bacteria

Water samples were thoroughly mixed and 50 µl were spread on Nutrient agar plates. The plates were incubated at 28⁰C for up to 72 h for the bacterial growth. Morphologically different bacteria were separately isolated from single colonies and subjected for further biochemical characterization tests. Representative cultures were subjected to Gram's reaction by potassium hydroxide solubility test, Starch hydrolysis test, Arginine dihydrolase test, Gelatin hydrolysis test, Levan formation, Lipase activity, Casein hydrolysis test, H₂S production test Kovac's oxidase test, Catalase activity, and O/F tests using standardized procedures (MacFaddin 1981, Goszczynska *et al.* 2000). These bacterial colonies are also subjected to growth at different temperatures, by differential medium, observation under UV light, pathogenicity tests by detached leaf bioassay and also intact leaf infiltration on *Sedum* and *Iris* plant leaves with *Nicotiana tobaccum* (tobacco) hypersensitive response. Hypersensitive response and pathogenicity tests were performed by infiltrating the bacterial suspension (10^7 CFU mL⁻¹) to tobacco and sedum plants. Sterile distilled water and the reference/authentic cultures were used as negative and positive controls respectively. All these characterizations tests were performed in four replicates each and repeated three times with appropriate reference cultures.

Extraction, amplification of bacterial DNA by PCR and sequencing

Genomic DNA of all the phytopathogenic bacterial cultures was extracted using ultra clean microbial DNA isolation kit (MO BIO Laboratories, Inc, CA, USA). The yield of sample DNA was estimated by electrophoresis in a 1 % agarose gel, followed by staining with ethidium bromide. The DNA from all the phytopathogenic bacteria was amplified using PCR with bacterial 16S primers (Nubel *et al.* 1996). PCR amplification assays were performed using Eppendorf Master Cycler Gradient (Thermal Cycler, Brinkmann, Germany). The reaction contained a total volume of 50 µl of mixture including 1 µl of template DNA extract, 2.5 mM each of the dNTPs (4 µl) , 5 µM each of upstream and downstream primers (5 µl), 1x PCR buffer and 0.25 µl of *Taq* DNA polymerase (TaKaRa Mirus Biocorporation, Madison, WI, USA) (5 U/µl) in the buffer provided by the manufacturers. The PCR was programmed by denaturing the DNA for 2 min at 96⁰C, followed by 40 repeated cycles of melting, annealing and DNA

extension at 94⁰C for 30 sec, 55⁰C for 25 sec and 72⁰C for 50 sec respectively. For the last cycle, the extension time was increased to 10 min. Aliquots (5 µl) of PCR products were analyzed on 1 % agarose gel, stained in ethidium bromide and were visualized with an UV transilluminator (Epi Chemi II, UVP Laboratory products, CA, USA). All the PCR products were cleaned up using wizard SV gel and PCR clean up system (Promega, Madison, WI, USA) and sequenced (MCLab, San Francisco, CA, USA). The sequence of PCR products were analyzed for the identity of bacteria by BLAST search in GenBank (<http://www.ncbi.nlm.nih.gov>)

The bacterial isolate with ambiguous identity was subjected to Fatty Acid Methyl Ester (FAME) analysis with the use of MIDI microbial identification system (Microbial ID, Newark, DE 19713, USA).

RESULTS AND DISCUSSION

Six bacteria were isolated from the water samples were phytopathogenic in nature. Characterization of these bacteria using morphological and biochemical tests, categorized them in to two distinct pathogenic genera. Pathogenicity tests on both sedum and iris plants showed that both the genera are pathogenic. They induced tobacco hypersensitive reactions within 2-5 days. Morphologically, the colonies of the first bacterial isolate were creamy white, flat, circular on Kings B medium, and produced a pale fluorescent pigment under UV light. They are Gram negative, negative for oxidase test, pectolytic activity, and arginine dihydrolase test but positive for gelatin liquefaction, levan production, catalase activity (Table 1). The bacterial culture did not grow beyond 37⁰C. Further, a PCR amplification with 16S rDNA primer produced fragment of about 430 bp. Sequence analysis of this fragment revealed the best match to *Pseudomonas syringae*, which is confirmed by morphological, biochemical characters and FAME analysis (with 86 % similarity index). The other group of phytopathogenic bacterial isolate is positive for catalase, oxidase, O/F tests, tobacco hypersensitivity and pathogenicity tests (Table 1). Similarly based on the morphological, biochemical tests and PCR amplification with 16S rDNA, the bacterial isolate is identified as *Erwinia carotovora*.

In this study, we have found out the presence of the phytopathogenic bacterial isolates from the nursery retention ponds, where irrigation water is recycled. Similar results are obtained by Norman *et al.* (2003), who recovered phytopathogenic *Erwinia* from irrigation ponds, infecting ornamental plants, in Florida. Populations of *Erwinia* spp. with in the retention ponds of Florida were substantially smaller than those found in other states. Harrison *et al.* (1986) detected large quantity of phytopathogenic bacteria in Colorado River water and Cappaert *et al.* (1988) in Oregon irrigation water. Specific species of *Erwinia* found in water sources have been found to vary with geographic location and water source. Harrison *et al.* (1987) collected large number of phytopathogenic bacteria from rivers, ponds and reservoirs in different states. Of these isolates, 98 % were *E. carotovora*. Similar results were obtained by McCarter-Zorner *et al.* (1984) who examined surface water in Colorado and found that 95 % of the water samples contained phytopathogenic *E. carotovora*.

A number of techniques have been employed to genetically identify phytopathogenic bacteria, such as bacteriophages, serology, protein profiles, FAME analysis, RAPD-PCR (Maki-Valkama

and Karjalainen 1994) and RFLP (Nassar *et al.* 1996). Some of the techniques used in these studies have been useful for distinguishing populations distributed geographically and by host of origin. In this study a PCR with universal primers was utilized to characterize both the phytopathogenic bacteria. A number of studies have been conducted on variations in virulence has host range of the phytopathogenic bacteria. However, we have not measured the virulence of these bacteria, but they are positive for tobacco hypersensitive response and pathogenic on both sedum and iris plants. With the high susceptibility of ornamentals to these bacteria, it would seem inadvisable to use untreated water from these irrigation reservoirs. Also it appears that, these bacterial isolates increase within water recycled from nursery retention ponds. More research is needed to identify a direct link between the use of contaminated irrigation water and an increased incidence of bacterial diseases in nurseries.

Table 1. Reaction of *Pseudomonas syringae* and *Erwinia carotovora* to various biochemical characterization tests.

Biochemical tests	Bacterial Pathogens	
	<i>Pseudomonas syringae</i>	<i>Erwinia carotovora</i>
Gram's reaction	-	-
Gelatin hydrolysis	+	NT
Arginine dihydrolase test	-	NT
Levan formation	+	NT
Pectolytic activity	-	NT
Kovac's oxidase test	-	+
Catalase activity	+	+
O/F test	NT	+
Tobacco HR test	+	+
Pathogenicity on Sedum	+	+
Pathogenicity on Iris	+	+

All these characterization tests are the average of three independent experiments of four replicates each, + = Positive, - = Negative, NT= Not Tested.

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BACTERIAL COMMUNITIES FROM A HEADWATER STREAM AT AN ABANDONED ARSENIC MINE

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ABSTRACT

Analysis of bacterial communities in the environment has consistently presented obstacles and difficulties to scientists who study them. Microbial communities are complex, and many microbes have yet to be discovered due to their inability to survive outside of their natural habitat for culturing. Microbes also have the ability to thrive in nearly any environment, including those with conditions detrimental to other organisms such as the presence of heavy metals. Our research objectives are to use molecular techniques to isolate and analyze members of microbial communities contaminated with heavy metals and to track the composition of the communities over time. We use sediment samples collected from a headwater stream adjacent to an abandoned arsenic mine in Floyd County, Virginia. The concentration of arsenic at this site is quite variable with concentrations measured at below the detectable limit to more than a thousand-fold greater. We use molecular techniques including Polymerase Chain Reaction (PCR) and gradient gel electrophoresis to create fingerprints (profiles) of community composition with the bacterial community DNA extracted from these sediment samples. In these gels, different banding patterns correspond to different bacterial communities. We isolate individual members of the community and identify them using ribosomal DNA sequence data. Identifying and documenting members of these bacterial communities will further our understanding of complex microbial communities in aquatic systems, and will further our understanding of communities that have potential for heavy metal remediation in stream systems.

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EFFECTS OF THE ANTIMICROBIAL AGENT TRICLOSAN ON BACTERIAL RESISTANCE TO DISINFECTION IN WASTEWATER TREATMENT PROCESSES

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KEY WORDS: Triclosan, disinfection, chlorine, UV, wastewater

ABSTRACT

The objective of this research was to investigate the effect of Triclosan on bacterial susceptibility to disinfection in wastewater treatment. An active sludge bacterial community was acclimated over several sub-cultures in the presence of different concentrations of Triclosan, ranging from 0.0 to 20 mg L⁻¹. Acclimated bacteria were then exposed to sub-lethal doses of chlorine (5.0 mg L⁻¹ min Cl) or UV (2.5 mJ cm⁻²). Determination of bacterial viability by direct counting and by flow-cytometry showed that microbial communities exposed to Triclosan exhibited an *increased* susceptibility to chlorination and UV by comparison to non-exposed bacteria. Characterization of the microbial community by restriction fragment length polymorphism showed that acclimation on Triclosan resulted in a significant reduction of the bacterial diversity, suggesting that the change of susceptibility can be explained, at least partly, by a change of the community structure. The most prevalent species detected in Triclosan-acclimated cultures were identified as *Pseudomonas fluorescens* and *Serratia marcescens*, two potentially important opportunistic pathogens previously documented as highly resistant to Triclosan. This paper reports, for the first time, that exposure to the antimicrobial agent Triclosan can modify the susceptibility of activated sludge bacteria to wastewater disinfection processes.

INTRODUCTION

Introduction of emerging contaminants, such as pharmaceuticals and personal care products (PPCPs) and antimicrobial agents into sewage represents a new challenge for wastewater treatment processes (Koplin et al., 2002). 2-(2,5-Dichlorophenoxy)-5-chlorophenol or Triclosan is a powerful germicidal agent included in a multitude of products, including detergents, toothpaste, and footwear. Triclosan has been detected worldwide in water, soil, sediments, and aquatic organisms (Singer et al., 2002; Halden and Paull, 2005). Although Triclosan is relatively innocuous for higher organisms, it is suspected to have potential detrimental effects on the environment. Triclosan is toxic for bacteria and for aquatic organisms, such as algae and fish (Orvos et al., 2002; McBain et al., 2003). Even though Triclosan can be metabolized by activated sludge bacteria, it usually resists complete biodegradation, resulting in significant concentrations in receiving waters (Singer et al., 2002). In the presence of hypochlorite and/or sunlight, Triclosan can also be transformed into more toxic and persistent derivatives, such as chlorophenols, polychlorinated biphenyl ethers, and dioxins (Mezcua et al., 2004). Finally, by its

antimicrobial properties, Triclosan is also suspected to induce cross-resistance to antibiotics (Yazdankhah et al., 2006).

The objective of this research is to investigate the effect of Triclosan on the susceptibility of activated sludge bacteria to the disinfection processes, chlorination and UV. Disinfection is usually the last step of wastewater treatment, which inactivates potential pathogens before the treated effluent is discharged into receiving waters. Our central hypothesis is that Triclosan, by its effect on activated sludge bacteria, can modify the susceptibility to disinfection, potentially increasing the risk that pathogens are discharged into the environment.

MATERIALS AND METHODS

Bacterial suspensions

Samples of wastewater were collected in a nearby wastewater treatment plant (Morgantown Utility Board, Star City, WV). Bacterial suspensions were prepared from the activated sludge by homogenization to disaggregate bacterial flocks, followed by filtration on Whatman paper (11- μm pore size). The optical density (O.D.) at 600 nm was measured to estimate the cell concentration. Bacterial suspension were stored at 4 °C and used for Triclosan acclimation and disinfection experiments.

Acclimation of bacterial suspensions to Triclosan

Bacterial mixtures were cultivated in liquid synthetic sewage feed (EPA, 1996) spiked with concentrations of Triclosan ranging from 0.0 to 100 mg L⁻¹. Cultures were incubated under agitation at 150 rpm, 25 °C. At the end of the exponential growth phase (24 - 30 hrs), a 100- μL aliquot of each culture was collected and used as an inoculum to initiate sub-cultures being incubated under similar conditions in the presence of Triclosan. Three subsequent sub-cultures were conducted for acclimation before to be used in disinfection experiments.

Disinfection experiments

Suspensions of Triclosan-exposed bacteria were washed three times in sterile phosphate buffer solution (PBS) and diluted to a final O.D.₆₀₀ = 0.05 AU (about 10⁷ cells mL⁻¹). Diluted suspensions were exposed to increasing doses of chlorine and UV. Chlorine was added as sodium hypochlorite (NaClO) solution (commercial bleach, 5% free chlorine). Ten mL of diluted bacterial suspensions were exposed to increasing doses for 5 min under magnetic stirring. Chlorine was added at doses ranging from 0.0 to 25 mg L⁻¹ min Cl. Residual free chlorine was neutralized by an excess of sodium thiosulfate (Na₂S₂O₃). Initial chlorine demand will be determined by the diethyl-*p*-phenylenediamine (DPD) method. UV disinfection was performed using a UV collimated beam unit with a low-pressure mercury lamp with a maximum emission at 254 nm (Atlantic Ultraviolet, Peabody, MA). Fifteen-mL aliquots of cell suspensions were exposed to increasing doses of UV in shallow quartz dishes under magnetic stirring. UV intensity at the surface of the liquid was measured (0.1 – 0.2 mW cm⁻²) and samples were exposed for increasing periods of time, corresponding to doses ranging from 0.0 to 200 mJ cm⁻².

Determination of cell viability

The effect of disinfection on cell viability was determined by plating serial dilutions on synthetic sewage feed agar (EPA, 1996) and colony forming unit (CFU) counting, after incubation at 25 °C for 24 h. Because only a slight percentage of environmental bacteria are cultivable, cell viability was also determined by flow-cytometry using the dye exclusion binary system propidium iodide-Syto9 on a BD FACStar Plus flow cytometer (Becton-Dickenson, Franklin Lakes, NJ).

Characterization of the microbial community

The microbial community structure was characterized by restriction fragment length polymorphism (RFLP) analysis. Total DNA was extracted using DNeasy[®] Tissue Mini Kit (Qiagen, Valencia, CA) using the modified protocol for Gram+ bacteria. PCR amplification of 16S rDNA was performed on a MasterCycler Gradient[®] thermocycler (Eppendorf, Westbury, NY) using HotStarTaq[®] Master Mix Kit (Qiagen) and the bacterial universal primers, 27f and 1492r. Clone libraries were constructed using TOPO TA Cloning[®] Kit (Invitrogen, Carlsbad, CA). Plasmids were extracted using QIAprep[®] Miniprep Kit (Qiagen) and the inserts were re-amplified using primers M13. PCR products were digested using the restriction enzymes *Hae*III (20 U/rxn) and *Hha*I (20 U/rxn) (New England Biolabs, Ipswich, MA) and analyzed by agarose gel electrophoresis. 16S rDNA clones were identified by sequencing (Davis Sequencing, Davis, CA) and search against public databases (<http://www.ncbi.nlm.nih.gov/> and <http://rdp.cme.msu.edu/>).

Triclosan analysis: Samples were prepared by centrifugation to remove bacterial cells, mixed with one volume of acetonitrile, and filtered on 0.2 µm. Triclosan was analyzed by reverse phase HPLC using a HP Series 1100 (Hewlett-Packard, Palo Alto, CA) on a C₁₈ Supelcosil LC-18 column (Supelco, Bellefonte, PA). The mobile phase was a gradient system of acetonitrile:H₂O operating from 50 to 90% acetonitrile in 20 min, at a flow rate of 1.0 mL min⁻¹. Triclosan was detected by the absorbance at 254 nm.

RESULTS AND DISCUSSION

Exposure of an activated sludge microbial community to Triclosan

A complex bacterial mixture from an activated sludge was exposed to increasing concentrations of Triclosan in agitated batches. Aliquots of the cell suspensions were collected and sub-cultured two times in the presence of Triclosan for acclimation. Figure 1 shows the growth curves of the bacterial suspensions during the three sub-cultures as measured by the O.D. at 600 nm.

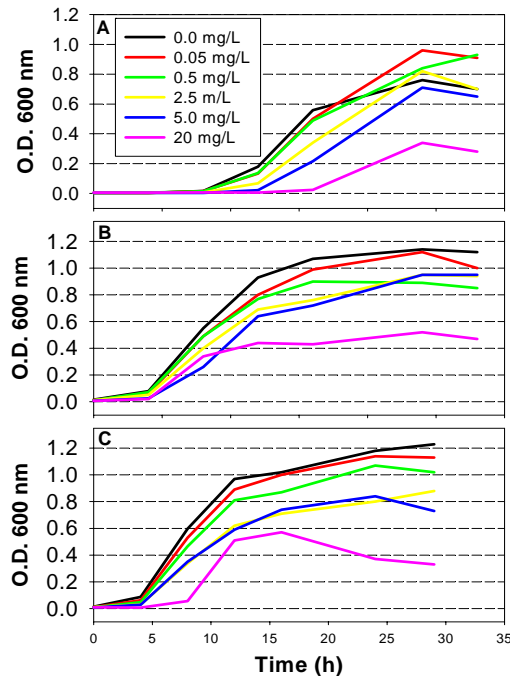


Figure 1: Growth curves of activated sludge bacterial suspensions exposed to different concentrations of Triclosan, as measured by the optical density (O.D.) at 600 nm: (A) Initial culture, (B) 1st sub-culture, (C) 2nd sub-culture.

Exposure to Triclosan resulted in a decrease of the rate of growth in a concentration-dependent fashion, even though a positive growth was still observed at the highest level (20 mg L^{-1}). Also the lag phase was significantly reduced after the first exposure, indicating acclimation of bacteria on Triclosan. These results are consistent with previous reports. For instance, studying the effect of Triclosan on 11 different bacterial strains, McBain et al. (2004) reported Triclosan minimum inhibitory concentrations (MIC) ranging from 0.1 to 21 mg L^{-1} . The authors also observed that previous exposure to Triclosan resulted in a higher resistance in 7 out of the 11 strains tested. Acclimated strains were resistant to up to 39 mg L^{-1} Triclosan.

Effect of Triclosan on bacterial susceptibility to chlorination

A control, non-exposed activated sludge bacterial suspension was first treated with increasing doses of chlorine (from 0.0 to $25 \text{ mg L}^{-1} \text{ min Cl}$) in order to determine sub-lethal doses being used in further disinfection experiments. Bacterial survival numbers were determined by flow-cytometry and CFU counting (Figure 2). The results show that chlorine concentrations above $1.0 \text{ mg L}^{-1} \text{ min Cl}$ exerted a significant effect on the bacterial growth. However, even very high concentrations of chlorine (up to $25 \text{ mg L}^{-1} \text{ min Cl}$) did not inhibit bacterial growth. To test further the effect of Triclosan acclimation on susceptibility to chlorine, cell suspensions were exposed to a sub-lethal dose of 5.0 mg L^{-1} . Typically only 1 to 15% of bacteria from activated sludge are cultivable, therefore cell viability was tested using in parallel direct colony counting *and* flow-cytometry, which is a culture-independent technique.

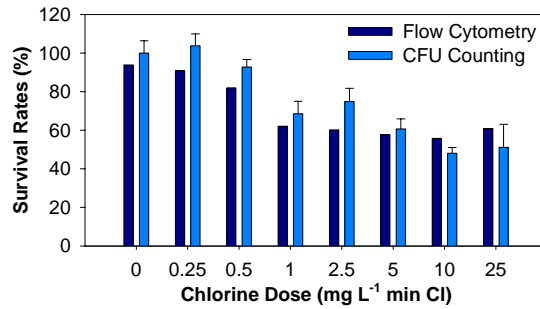


Figure 2: Survival rates of activated sludge bacteria exposed to increasing concentrations of chlorine (NaClO), as measured by flow-cytometry and by plating and CFU counting.

Activated sludge bacterial suspensions acclimated in the presence of different concentrations of Triclosan (0.0, 0.5, 2.5, and 20 mg L⁻¹) were exposed to a sub-lethal dose of chlorine (5.0 mg L⁻¹ min Cl). Bacterial survival rates were again determined by flow-cytometry and by CFU counting (Figure 3). The results showed that acclimation on Triclosan dramatically increased the susceptibility of activated sludge bacteria to chlorination. This can be explained by a change of the bacterial community structure toward strains less resistant to chlorine, or by physiological changes in bacterial cells, or both. It is interesting to note that the two methods for determination of cell survival numbers, plating and flow-cytometry, showed similar patterns, suggesting that susceptibility to chlorine was comparable in cultivable and non-cultivable bacteria.

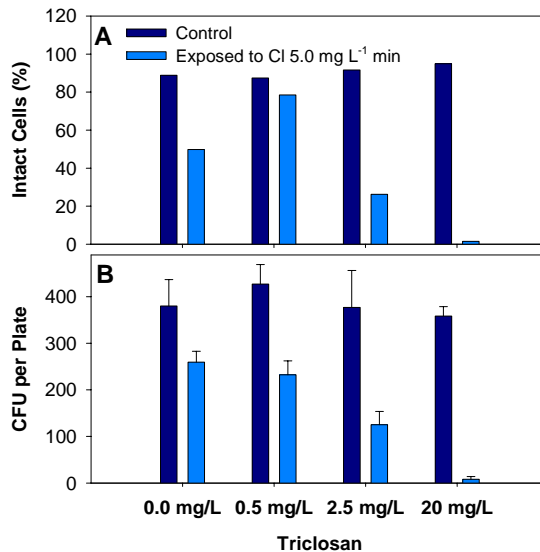


Figure 3: Survival rates of activated sludge bacteria acclimated on increasing concentrations of Triclosan and exposed to 5.0 mg L⁻¹ min of chlorine, as measured by flow-cytometry and CFU counting.

Effect of Triclosan on bacterial susceptibility to UV

In order to determine the sub-lethal dose being used in further disinfection experiments, a non-exposed bacterial suspension was first treated with increasing doses of UV (from 0.0 to 200 mJ cm⁻²). Bacterial susceptibility to UV was assessed by plating and CFU counting. Figure 4 shows that a dose of 5.0 mJ cm⁻² significantly impacted cells, while a dose of 100 mJ cm⁻² was

necessary to fully inhibit bacterial growth. To test further the effect of exposure to Triclosan on susceptibility to UV, cell suspensions were exposed to a sub-lethal dose of 2.5 mJ cm^{-2} . Flow-cytometry was not used to detect UV-exposed cells. Flow-cytometry cell viability analyses were based on the exclusion dye, propidium iodide, which penetrates only damaged cells. UV irradiation is known to impact primarily DNA and does not affect significantly the membrane structure, therefore preventing detection by flow-cytometry.

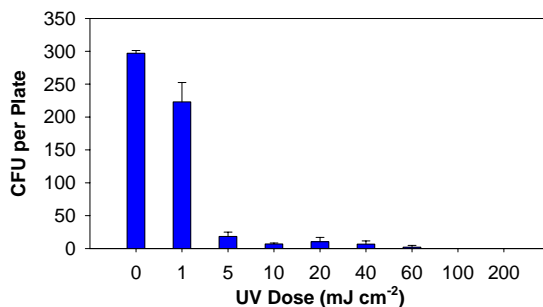


Figure 4: Effect of UV irradiation (low-pressure UV, 254 nm) on the survival of bacteria of an activated sludge, as measured by plating on solid medium and CFU counting.

Activated sludge bacteria sub-cultured three times in the presence of increasing concentrations of Triclosan ($0.0, 0.5, 2.5,$ and 20 mg L^{-1}) were exposed to a sub-lethal dose of UV (2.5 mJ cm^{-2}). Bacterial survival rates were determined by plating on synthetic sewage feed and CFU counting (Figure 5). The results showed that acclimation on the highest dose of Triclosan (20 mg L^{-1}) increased the susceptibility of activated sludge bacteria to UV irradiation. However, acclimation to 0.5 to 2.5 mg L^{-1} did not modify significantly cell susceptibility, as compared with non-exposed cells. As observed previously with chlorination, this result can be explained by a change of the bacterial community structure and/or by physiological changes of the cells.

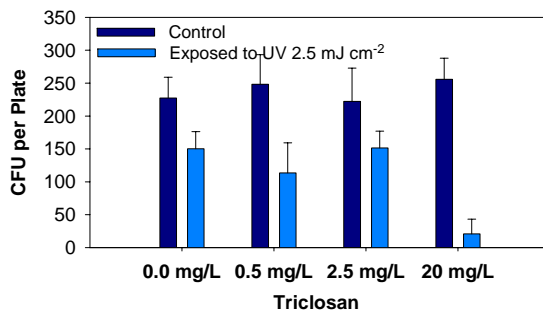


Figure 5: Survival rates of activated sludge bacteria acclimated to increasing concentrations of Triclosan and exposed to UV irradiation at 2.5 mJ cm^{-2} (254 nm), as measured by plating and CFU counting.

Effect of Triclosan on the bacterial community structure

In order to test the hypothesis that acclimation on Triclosan would affect the community structure, potentially explaining the observed increase of susceptibility to disinfection, the bacterial community was analyzed using RFLP. Two clone libraries were constructed, the first from a control, non-exposed bacterial mixture and the second from a mixture sub-cultured three times in the presence of 20 mg L^{-1} Triclosan. Table 1 shows that exposure to Triclosan resulted

in a significantly lower bacterial diversity, as compared to the non-exposed culture: 31 different strains out of a total of 78 clones analyzed have been isolated from the control mixture and only 5 different strains out of a total of 50 clones have been isolated from the Triclosan-acclimated sludge. The Shannon diversity index was 2.76 for the non-exposed culture and 1.45 for the Triclosan-acclimated culture. The two major strains detected in the activate sludge acclimated on Triclosan were isolated and cultivated in the lab as pure cultures. The strains were further identified by 16S rDNA fingerprinting as *Pseudomonas fluorescens* and *Serratia marscecens*.

Table 1: Bacterial strains identified from a control, non-exposed activated sludge mixture and a mixture acclimated on 20 mg L⁻¹ Triclosan.

Non-Exposed Activated Sludge				Acclimated on Triclosan			
Genus	#	Genus	#	Genus	#	Genus	#
<i>Pseudomonas</i> #1	21	<i>Janthinobacterium</i>	1	<i>Blastobacter</i>	1	<i>Serratia</i> #2	14
<i>Pseudomonas</i> #2	13	<i>Crocinitum</i>	1	<i>Nostoc</i>	1	<i>Pseudomonas</i> #6	14
<i>Comamonas</i>	6	<i>Alysiella</i>	1	<i>Xenohalotis</i>	1	<i>Stenotrophomonas</i> #2	11
<i>Acinetobacter</i> #1	4	<i>Trichococcus</i>	1	<i>Salmonella</i>	1	<i>Providentia</i>	10
<i>Flavobacterium</i>	3	<i>Phyllobacterium</i>	1	<i>Serratia</i> #1	1	<i>Caulobacter</i>	1
<i>Stenotrophomonas</i> #2	3	<i>Pseudomonas</i> #5	1	<i>Raoultella</i>	1		
<i>Sphingobacterium</i>	3	<i>Stenotrophomonas</i>	1	<i>Ralstonia</i>	1		
<i>Lactobacillus</i>	2	<i>Ruminococcus</i>	1	<i>Caulobacter</i>	1		
<i>Aeromonas</i>	1	<i>Microvirgula</i>	1	<i>Ochrobactrum</i>	1		
<i>Pseudomonas</i> #3	1	<i>Gloeothece</i>	1				
<i>Pseudomonas</i> #4	1	<i>Bergeyella</i>	1	Total	78	Total	50

Similarly, McBain et al. (2003) observed a reduction of the microbial diversity of a wastewater microcosm upon exposure to Triclosan. The authors also identified Triclosan-resistant strains in the groups of pseudomonads and stenotrophomonads. It is interesting to note that our two isolates from Triclosan-acclimated activated sludge are the same Triclosan-resistant strains that were previously isolated from a food processing facility (Moretro et al., 2006). By its antimicrobial properties, exposure to Triclosan can potentially modify cell properties, such as wall and membrane composition, possibly increasing susceptibility to disinfection. Particularly, the physiological action of Triclosan is the inhibition of the enoyl-acyl carrier protein reductase (ENR), a key enzyme involved in the membrane fatty acids biosynthesis (Yazdankhah et al., 2006). However this effect can be masked by a change in the microbial community structure (McBain et al., 2003). According to our results, the increase of bacterial susceptibility after acclimation on Triclosan can be explained, at least partly, by a change of the microbial community structure.

CONCLUSIONS

Instead of observing, as hypothesized, a decrease of the bacterial susceptibility after acclimation on Triclosan, our results showed an *increased* susceptibility to chlorination and UV. This overall change of susceptibility could be explained by physiological changes of the bacterial cells or by a change of the bacterial community structure. Indeed, phylogenetic analyses showed a significant change of the composition of the bacterial community, including a reduction of diversity, upon exposure to Triclosan. Further experiments will be conducted on selected

Triclosan-resistant strains, such as *P. fluorescens* and *S. marscecens*, in order to determine if exposure to Triclosan can induce physiological cell modifications, including a *decrease* of susceptibility to disinfectants, potentially threatening human health and the environment. To the extent of our knowledge, this study is the first to report that exposure to the antimicrobial agent Triclosan can modify the susceptibility of activated sludge bacteria to disinfectant processes.

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DETECTION OF NOROVIRUS IN SEWAGE EFFLUENT AND ITS PERSISTENCE IN ESTUARINE WATER USING REAL-TIME PCR

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ABSTRACT

Noroviruses are causative agents of epidemic gastroenteritis worldwide. They are highly contagious and primarily spread by direct human contact, water or food, with contaminated shellfish serving as a vector for transmission. Despite the acknowledged public health importance of noroviruses, little is known about their occurrence, removal/inactivation during sewage treatment or their fate in environmental receiving waters. This project examined the use of real-time PCR to determine the presence and densities of norovirus in representative treated sewage effluents discharged to Chesapeake Bay receiving waters. We describe our evaluation and optimization of methods for sewage effluent sample concentration and norovirus RNA extraction. FRNA coliphages, a proposed indicator for enteric viruses, were also quantified simultaneously using a culture-based method to evaluate if FRNA coliphage presence and densities can be used as a surrogate indicator of norovirus presence. Additionally, using in situ and in vitro experiments, we examined the persistence of noroviruses and FRNA coliphages in estuarine waters in response to sunlight, water temperature and salinity. Preliminary data indicate that norovirus is more persistent than FRNA coliphage. Inactivation rates and T_{90} data will be useful in the assessment and management of water quality with respect to health risk in recreational and shellfish growing waters.



Session I-C

Algal Blooms, Biofuels, and Other Nutrient Features

- A Window to Reassess the Potential Impacts of Nutrients in Irrigation Runoff on Natural Water Resources
- Stimulation of Algal Growth by Effluent Organic Nitrogen Along a Salinity Gradient
- Photochemical Release of Labile Nitrogen from Natural and Effluent Organic Nitrogen

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A WINDOW TO REASSESS THE POTENTIAL IMPACTS OF NUTRIENTS IN IRRIGATION RUNOFF ON NATURAL WATER RESOURCES

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KEY WORDS: BMP, irrigation water, nutrient management, water health, water quality

ABSTRACT

In a research project investigating how plant pathogens survive and distribute in irrigation runoff water retention basins, we continuously monitored water quality using a Hydrolab DataSonde 5X. Major water quality parameters monitored included: (i) Chlorophyll a, (ii) conductivity, (iii) dissolved oxygen, (iv) oxidation-reduction potential, (v) pH, (vi) salinity, (vii) temperature, (viii) total dissolved solids, and (ix) turbidity. Measurements were taken hourly from November 16, 2006 to March 12, 2007. Two algal blooming events were observed. Dramatic changes in water pH, dissolved oxygen, and oxidation-reduction potential were closely associated with these algal bloom events. Both pH and dissolved oxygen concentration fluctuated coincidentally, whereas oxidation-reduction potential changed inversely with chlorophyll readings. Following each algal bloom event was a peak of turbidity. Total dissolved solids, salinity and conductivity also peaked after each algal bloom event, but they tended to increase over time. This research provides direct evidence demonstrating the potential impact of irrigation runoff water on natural water resources if not contained in retention basins. It also clearly indicates the need to monitor recycled water health and assess the impacts of algal bloom events and the subsequent water quality changes on

plant health and plant disease management at crop production facilities where surface water is recycled.

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STIMULATION OF ALGAL GROWTH BY EFFLUENT ORGANIC NITROGEN ALONG A SALINITY GRADIENT

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ABSTRACT

Wastewater treatment plants are point sources of nitrogen and phosphorus to natural waters, contributing to eutrophication of our waterways. Targeted phosphorus reductions have been successful in reducing nuisance algal growth in freshwater, however, nitrogen pollution continues to plague estuarine waters. Therefore, in the Chesapeake Bay 2000 Agreement, watershed nitrogen discharge must be reduced by 48% below 1985 levels by 2010. This represents an enormous expense to utilities and challenges existing technologies. Nitrogen in secondary treated effluent includes both inorganic and organic compounds. Inorganic nitrogen is more easily removed by the most common treatment systems by employing coupled nitrification/denitrification. As a result, a substantial fraction of the residual nitrogen in treated wastewater effluent is organic nitrogen, which we call effluent organic nitrogen (EON). Based on limited data regarding organic nitrogen bioavailability, EON has been assumed to be refractory, therefore biologically unavailable, leading some to propose that it not be included in discharge allowances. However, recent work in marine and estuarine systems suggests that dissolved organic nitrogen is available to bacteria and algae. In the Chesapeake Bay watershed, effluent discharged into the watershed may be transported from DIN replete freshwaters to N-limited estuarine waters along a salinity gradient. We conducted bioassays using natural water samples collected along a salinity gradient in coastal Virginia and the high molecular weight fraction from two effluents obtained from wastewater treatment facilities. Preliminary results indicate that high molecular weight EON stimulated algal growth in bioassay bottles and that this stimulation was greatest at the higher salinity stations.

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PHOTOCHEMICAL RELEASE OF LABILE NITROGEN FROM NATURAL AND EFFLUENT ORGANIC NITROGEN

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KEY WORDS: photochemical organic nitrogen effluent

ABSTRACT

Research in aquatic systems has provided evidence that photochemical processes can result in the release of labile nitrogen from dissolved organic nitrogen (DON). The degree that photochemical release occurs when effluent organic nitrogen (EON) is released into the environment and exposed to sunlight is unknown. We quantified rates of photochemical release of ammonium (NH_4^+), dissolved primary amines (DPA), and nitrite in three environments: the Savannah and Altamaha Rivers in Georgia (spring, summer, and fall), the South Atlantic Bight (SAB; summer) and in EON collected from the Virginia Initiative wastewater treatment plant in Norfolk, VA. Water or EON from each site was filtered through a 0.2 μm Supor filter and incubated in quartz tubes using natural sunlight. Treatments included samples incubated in both the light and the dark. Photoproduction of NH_4^+ was observed in five of ten incubations in the rivers, in three of four incubations in the SAB and during two of the three sampling periods in the EON incubations. Photoproduction of DPA was observed in six of ten incubations in the river but was not observed in the SAB or the EON. Release of NO_2^- was only seen in one incubation in the Savannah River but sustained NO_2^- consumption was observed in the EON incubation.

INTRODUCTION

Recent findings in freshwater and marine systems indicate that photochemical processes can affect the release of labile nitrogen (N) moieties from dissolved organic matter (DOM; reviewed in Bronk 2002). Bushaw *et al.* (1996) demonstrate that DON from a freshwater pond is a source of labile N for microbial processes, but only after the DON is irradiated with sunlight and that

wavelengths in the ultraviolet (UV) region (280 - 400 nm) produce these compounds from DOM sources most efficiently. This photochemical reactivity can alter the bioavailability of DON. However, photochemical reactions can affect the lability of organic material along estuarine gradients (Bushaw *et al.* 1996; Minor *et al.* 2006) and readily convert refractory DON to labile forms. A recent paper shows that biologically recalcitrant DOM can be converted into bioavailable forms via photochemical reactions and subsequently stimulate N-limited microbial food webs (Vähätalo and Järvinen 2007). Additionally, previous work has shown that nitrite (NO_2^-) and ammonium (NH_4^+) can be released from DON photochemically (e.g. Kieber *et al.* 1999; Koopmans and Bronk 2002). This release may explain why bacterial growth efficiency, bacterial nutrient demand, and bacterial biomass and respiration rates are influenced by light (McCallister *et al.* 2005). Similarities between the composition of naturally occurring DON and organic N in effluent suggest that the EON may also be susceptible to photochemical breakdown.

The objective of this study was to quantify photoproduction rates of NH_4^+ , dissolved primary amines (DPA), and NO_2^- in a continental shelf, two riverine systems, and EON from a local wastewater treatment plant to determine the ability of organic matter from different sources to produce labile N compounds that could fuel phytoplankton and bacteria production. We hypothesize that photochemical energy can result in the release of labile N from naturally occurring DON and EON.

METHODS

Site description

Experiments were performed on the continental shelf (South Atlantic Bight – SAB) and in two riverine (Savannah and Altamaha Rivers, GA) ecosystems. These environments represent a range of nutrient and light regimes, from the relatively oligotrophic SAB, with a deep photic zone, to the eutrophic rivers with extremely shallow photic depths.

The SAB is a region on the continental shelf of the southeastern US that is wide (up to 200 km) and relatively shallow along the coast of Georgia. Although waters of the inner shelf are turbid, the middle and outer shelf have significant light penetration of up to 50 m or more (Yoder 1985). Station locations in the SAB were $79^\circ 29.9\text{W}$, $32^\circ 44.9\text{N}$ (3-3-99) and $79^\circ 08.6\text{W}$, $32^\circ 28.8\text{N}$ (3-5-99). Both the Savannah and Altamaha Rivers empty onto the continental shelf of the southeastern United States. The Savannah River is much more heavily impacted by industry and has been extensively modified by dredging and channelization. All three experiments in the Savannah River were set up at the dock at the Skidaway Institute of Oceanography ($31^\circ 59\text{N}$, $81^\circ 01\text{W}$). The Altamaha River is the second largest river on the eastern coast of the United States. The Altamaha is relatively pristine with minimal inorganic N inputs associated with urban and agricultural land use; the dominant form of N is organic. Station locations in the Altamaha were approximately $31^\circ 8\text{N}$, $81^\circ 24\text{W}$ on all three cruises. For the EON incubations, effluent was obtained from Hampton Roads Sanitation District's Virginia Initiative Plant in Norfolk, VA on 12 June 2007.

Experimental design

Water was collected, using 10 or 30L Niskin or Go-Flo bottles, from just below the surface and from a number of depths in the SAB and from and 1 m off the bottom in the rivers. Whole water was used to avoid any potential artifacts with isolating different organic fractions. After collection, samples were filtered through 0.2 μm filters, and placed in dark bottles or quartz flasks. Samples were incubated in on-deck incubators with flow-through seawater. In the EON incubations, triplicate of light and dark treatments were sacrificed 4, 21, and 28 hours, which equates to 4, 11, and 18 hours of direct sunlight received, respectively. At the end of the incubation, the concentration of NH_4^+ , DPA, and NO_2^- was measured in the dark controls and the quartz flasks. Photoproduction was considered statistically significant if the light treatment increased above that of the dark control and the error bars in the two treatments did not overlap.

Chemical analyses of N concentrations

Concentrations of NH_4^+ were measured manually with the phenol/hypochlorite technique (Grasshoff *et al.* 1999). Concentrations of NO_2^- were measured with an autoanalyzer using a standard colorimetric technique (Parson *et al.* 1984). Concentrations of DPA were measured with the fluorometric OPA method (Parsons *et al.* 1984).

RESULTS

Photochemical ammonification

In the SAB, rates of NH_4^+ photoproduction were fairly high (21 to 91 $\text{nmol N l}^{-1} \text{h}^{-1}$; Table 1), but NH_4^+ concentrations decreased in one of the four incubations. In the Rivers, NH_4^+ concentrations increased in the light treatment over the control in seven of 10 experiments, and five of those were significant. The mean NH_4^+ production rate in the two rivers was 24.1 $\text{nmol N l}^{-1} \text{h}^{-1}$ (Table 1). Photochemical ammonification was observed during the later two time points in the EON incubation (Table 2, note the change in units from Table 1).

Photochemical DPA production

In the SAB, DPA concentrations decreased in three of the four incubations – only one of them were significant (Table 1). In the rivers, DPA concentrations increased in the light treatment in nine out of 10 incubations, and six of these were significant resulting in a mean DPA photoproduction rate of 16.0 $\text{nmol N l}^{-1} \text{h}^{-1}$ in the two rivers. Consumption of DPA was observed at each time point in the EON incubations (data not shown).

Photochemical NO_2^- production

In the SAB, concentrations of NO_2^- were below the limit of analytical detection in all incubations in both the light treatments and dark controls. In the rivers, loss of NO_2^- was observed in six of

nine incubations, and four of these were significant (Table 1). Concentrations of NO_2^- were extremely high in the EON treatments (Figure 1).

DISCUSSION

The data from these experiments show that UV radiation exposure from sunlight can influence the decomposition of some components in DON and support the hypothesis that photochemical energy can result in the release of labile N from DON and, in the case of NH_4^+ , from EON. These data are significant because the labile N form is available for phytoplankton and bacteria uptake. If the amount of labile materials and the rate at which they cycle can be quantified, a better understanding of the amount and time frame at which inorganic and labile DON materials are available for uptake can be established. Observations also suggest that exposure to UV light can result in the consumption of some labile N forms, namely DPA and NO_2^- . A more comprehensive understanding of DON and EON can then be applied to making decisions regarding the allowable N release from wastewater treatments plants.

Table 1. Rates of NH_4^+ , dissolved primary amines, and NO_2^- photoproduction measured in the South Atlantic Bight (SAB) and two rivers and Georgia. Data are mean \pm standard deviation. NM = not measured. BD = below detection, meaning that an increase in concentration was observed in the treatment but it was not significant. L = a decrease in concentration was observed in the treatment relative to the control but it was not significant. Means were calculated using a rate of 0 for any samples that was BD or where a loss of substrate was observed.

Date	Depth (m)	Incub. Time (h)	Photoproduction Rate (ng-at N l ⁻¹ hr ⁻¹)		
			NH_4^+	DPA	NO_2^-
South Atlantic Bight					
3/3/99	1	7.0	91.4 \pm 12	BD	BD
	12	7.0	20.9 \pm 3.7	L	BD
3/5/99	1	8.6	L	L	BD
	35	8.6	29.0 \pm 2.0	-1.3 \pm 0.1	BD
Mean \pm std			35.3	0.0	0.0
Savannah River					
3/31/98	1	7.8	1.3 \pm 9.5	19.0 \pm 14.1	2.8 \pm 2.6
	3	7.8	L	BD	L
7/18/98	1	9.3	BD	3.5 \pm 2.3	L
	3	9.3	19.1 \pm 19.2	9.6 \pm 4.2	BD
10/27/98	1	9.5	BD	BD	-2.9 \pm 2.8
	3	9.5	L	BD	-3.8 \pm 2.3
Altamaha River					
3/28/97	1	7.1	15.5 \pm 16.4	6.9 \pm 5.5	NM
7/16/98	1	9.5	36.3 \pm 27.5	L	L
	3	9.5	L	10.2 \pm 1.5	-8.6 \pm 3.8
10/29/98	1	8.3	36.0 \pm 17.8	37.8 \pm 13.5	-3.5 \pm 1.6
Mean \pm std			24.1	16.0	0.0

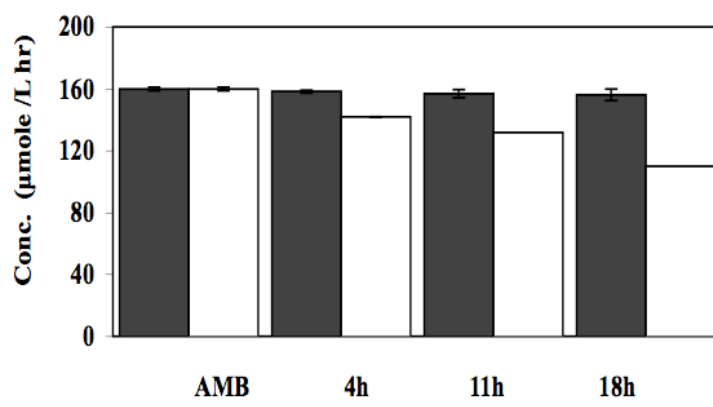


Figure 1. Concentrations of NO_2^- at the start of a photochemical incubation (AMB) and after exposure to sunlight for 4, 11, and 18 hours. Black bars denote dark incubations and white bars were incubated in quartz tubes.

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Session II-A

Assessing Drinking Water from Source to Consumer

- Drinking Water Assessment at Underserved Farms in Virginia's Coastal Plain
- Low Pressure Propagation Analysis in an Experimental Plumbing Rig
- New Chlorine Resistant Membranes for Reverse Osmosis and Nanofiltration Water Purification
- Formation and Sorption of Trichloromethanes in Chlorinated Polyvinyl Chloride and Polyethylene Pipes Used in Drinking Water Distribution Systems

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DRINKING WATER ASSESSMENT AT UNDERSERVED FARMS IN VIRGINIA'S COASTAL PLAIN

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KEY WORDS: drinking water, underserved, rural homes, Virginia's coastal plain.

ABSTRACT

This study assessed the chemical and microbiological qualities of drinking water samples collected from 185 locations covering 22 counties along the Coastal Plain of Virginia. All samples were taken from rural wells or springs where underserved farms and families in the region obtain their drinking water. Separate samples were collected for biological and chemical analyses and screened for selected water quality indicators. Total coliform, fecal coliform, and *Escherichia coli* were detected in 34, 11, and 6% of the samples, respectively. The levels of microbial contamination would be high during summer months when recharge is at minimum. Chemical analysis showed that 25% of the total wells surveyed were near the Atlantic Ocean and the Chesapeake Bay. Among them approximately 10% had salt-water intrusion, as indicated by sodium content exceeding USEPA limits for drinking water. Shallow wells in close proximity to agricultural fields would be expected to be high in nitrate and phosphorus, but most of the values obtained in this study were not considered detrimental to human health based on USEPA's drinking water standards. The pH of well

waters ranged from 4.5 to 8.5 depending on location. Survey results indicated that color, taste, and turbidity were the most common complaints reported by homeowners.

INTRODUCTION

Underserved (limited resource) farmers and ranchers are defined by the United States Department of Agriculture (USDA) as follows: (1) a person with direct or indirect gross farm sales not more than \$100,000 in each of the previous two years, and (2) a person with a total household income at or below the national poverty level of a family of four; or less than 50% of the country median household income in each of the previous two years (USDA-NRCS, 2006). Perhaps a reasonable definition of the underserved was given by a an earlier, 2003 edition, as follows: “underserved are those farmers and ranchers who, when compared with other farmers, ranchers, and farm operations in a given geographic area, such as a state, county or project area, have distinct disadvantages in obtaining U.S. Department of Agriculture program assistance” (USDA-NRCS, 2003). Recognizing the above to be the fundamental definitions for the underserved, this study identified communities and farms in the Coastal Plain of rural Virginia that best fit those definitions while collecting well water samples from homes and farmsteads.

In general, drinking water quality protection is a concern for all rural areas, farms, and communities, where the socio-economic status (poverty level) demographic distribution (primarily consisting of elderly black folks) and education levels (usually not beyond the elementary school) are vastly different from more affluent communities. Well water may be exposed to runoff and other types of pollution from local farming operations. It is not uncommon to find underserved farmers who do not know the threats that fertilizers, pesticides, and herbicides pose when not stored or disposed of properly. Another major dilemma is the location of septic tanks and outdoor toilet facilities near drinking water wells. Uninformed farmers usually don't realize that such proximity could lead to fecal coliform and *Escherichia coli* contamination. A report by Poff and Blake (2000) indicated that Virginia's ground water is heavily contaminated from various sources including agricultural chemicals, septic systems, and mining activities. Consequently, rural households may be at risk, if the only source of drinking water comes from shallow and hand-dug wells, as it is the case in most underserved farm communities. A separate research done by Reichenberger (1990) and Ross et al., (2000), had confirmed this concern to be true for most rural drinking water resources in Virginia. However, both studies did not target the underserved communities in their drinking water assessment. The main objective of this study was to assess drinking water quality of underserved rural households from 22 counties in the Coastal Plain of Virginia using sample analysis and survey data.

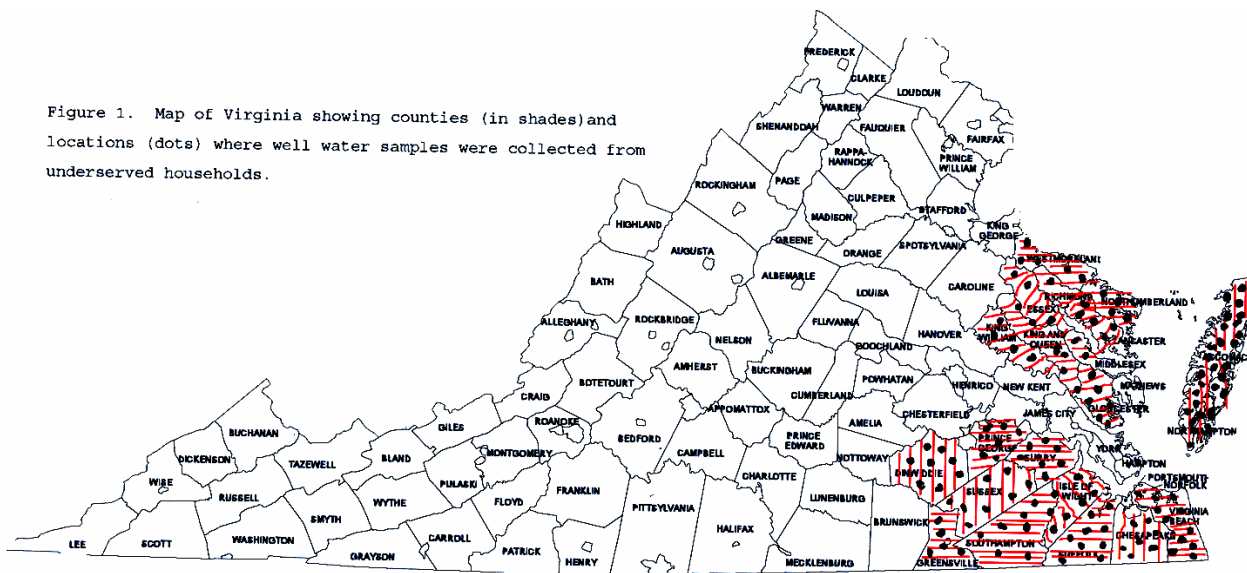
MATERIALS AND METHODS

A total of 185 samples from 22 counties in the Coastal Plain physiographic region of Virginia (Figure 1) were collected with collaboration from Natural Resources Conservation Service (NRCS) Research Conservation and Development (RC&D) Coordinators, volunteers, and extension personnel following USEPA procedures for well water sampling (USEPA, 1981). Two sets of samples were collected: one for chemical and the other for biological analysis. Samples for chemical analysis were collected in washed and acid rinsed, clean, amber glass bottles containing 10 mL 1N HNO₃ as preservative. For biological analysis, samples were collected in

EPA approved water sample collection vials (Biotrace, Bothell, WA), each containing one 10 mg sodium thiosulfate tablet that served as preservative. All samples were immediately sealed and transported on ice. Microbial testing was done within 24 h of sampling while samples for chemical analysis were kept frozen until analysis. Water samples (100 mL) were analyzed for metals following U.S.EPA procedure (EPA, 1981). Total coliform, fecal coliform, and *E. coli* bacteria were determined according to standard water testing methods (Greenberg et al., 1992). The term fecal coliform was defined as gram-negative facultative rods that ferment lactose at 44.5 °C (Pao and Brown, 1996).

During water sampling, homeowners were also asked 35 questions, which they were encouraged to answer. The questions included social, economic, and demographic information that would help assess the relationships among these parameters and drinking water quality at underserved households. No household member was obliged to answer any of the questions, if he or she did not feel comfortable answering. However, the significance of answering them was explained to them as being important in providing available assistance to protect drinking water quality and safety.

Figure 1. Map of Virginia showing counties (in shades) and locations (dots) where well water samples were collected from underserved households.



RESULTS AND DISCUSSION

Table 1 shows parameters that EPA uses as drinking water quality indicators along with values found in water samples collected from rural underserved households. Based on information gathered from literature and governmental sources (USEPA, 2002; Ross et al., 2000) most metals are not detrimental to human health. However, a critical component of drinking water assessment is determination of bacteria. Coliform bacteria are microorganisms commonly found in surface water, soil, and in the feces of humans and animals. They do not usually cause disease; however, their presence in the water indicates that fecal wastes may be contaminating the ground; this means that pathogenic organisms causing gastrointestinal diseases, hepatitis, or other human disease could be present in the water (Pao and Brown, 1996). In this study we found one or more coliform bacteria per 100mL water in more than 33% of the samples collected (Table 1), indicating many drinking water wells used by the underserved households at Virginia’s Coastal

Plain were contaminated with unknown microorganisms. The average coliform counts of the water samples collected from the coastal plain region were respectively about 0.73, 0.72, and 0.20 log CFU/mL, respectively. Water samples collected in the summer had higher coliform counts when compared to other seasons of the year. Fecal coliforms, including *E. coli*, are bacteria that originate from the intestines of warm-blooded animals (Entry and Farmer, 2000). These bacteria usually have a strong association with fecal contamination from warm-blooded animals. If one fecal coliform per 100mL of water is detected, the water is considered unsafe to ingest (Greenberg et al., 1992; USEPA, 2002). In this study we found that one or more fecal coliform and *E. coli* per 100mL water in 10 and 5% of the water samples collected (Table 1), indicating a significant number of underserved farmers and families of this tested region (especially in Dinwiddie County) were using water unsafe for human consumption.

It is to be recognized that coliform bacterial detection is simply an indication of the possible presence of pathogenic, or disease-causing organisms. Detection of coliform bacteria is confirmed by a total coliform analysis result above zero. Coliforms are always present in the digestive systems of all warm-blooded animals and can be found in their wastes. Coliforms are also present in the soil and in plant material. While a water sample with total coliform bacteria present may have been inadvertently contaminated during sampling, other possibilities include surface water contamination due to poor well construction, contamination of the household plumbing system, or water table contamination. To determine whether or not the bacteria were from human and/or animal waste, positive total coliform tests can be followed by an analysis for *E. coli* bacteria. Therefore, most probable number quantitative bacteria counts can be obtained for both total coliform and *E. coli* bacteria. The fact that generic *E. coli* (indicator bacteria) was found in 23% of tested samples from the Coastal Plain, made it evident that a significant number of drinking water wells used by the underserved farmers and families in this region were directly contaminated by either human or animal fecal matters and may cause serious harm to health. Additional study is warranted to investigate the source of *E. coli* and fecal coliform in affected wells; also to assess these wells for the presence of more serious pathogenic microorganisms (such as *Salmonella enteria*, *E. coli* O157:H7, etc).

Table 1. Drinking water quality indicators in well water samples collected from underserved rural households in Virginia’s Coastal Plain.

Parameters Measured	EPA Limits (mg/L)	Measured Concentrations (mg/L)			EPA Limits Exceeded
		Mean	Minimum	Maximum	No. of Samples
Iron (Fe)	0.3	0.26	0	17.0	15
Manganese (Mn)	0.05	nd ⁺	nd	nd	0
Copper (Cu)	1.0	nd	nd	nd	0
Sodium (Na)	30-60	970	67	21,438	22
Potassium (K)	--	266	50	1,438	--
Calcium (Ca)	--	306	4	6,760	--
Magnesium (Mg)	--	45	8	780	--
Arsenic (As)	0	nd	nd	nd	0
Lead (Pb)	0	nd	nd	nd	0
Mercury (Hg)	0.002	nd	nd	nd	0
Fluoride (F)	2.0	1.5	0.05	45.26	11
Chloride (Cl)	250	83	0.7	4953	2
Nitrate (NO ₃)	10	5.5	0.09	61.8	25
Nitrite (NO ₂)	1	0.15	0.02	11.9	4
Sulfate (SO ₄)	250	795	0.37	7,735	4
Phosphate (PO ₄)	--	7.9	0.02	342	--
Acidity (pH)	6.5 - 8.5	6.69	1.39	8.44	53
TDS (mg/L)	500	211	17.5	1803	15
Hardness (mg/L)	--	79.3	0	250	--
Saturation Index	--	-1.74	-8.95	0.75	--
Turbidity (NTU)	5	1.15	0.13	16.72	10
Alkalinity mg/L)	--	87.5	0	240	--
Total coliforms	0	4.38	0.37	51	71
Fecal coliforms	0	3.66	0.10	23	22
<i>E. coli</i>	0	1.96	0.04	23	12

⁺nd = not detected

Most of the targeted audiences lived on farms in rural areas, while some lived on remote rural lots; only 10% lived in housing developments. Out of the 179 households who cared to respond to the survey questionnaire, 60% identified themselves as African-Americans, 34% Caucasians, and 6% described themselves as other. The age distribution in these residences ranged from 36-45 years (54%) to over 65 years (16%). About half (48%) did not want to tell us their level of education; among those who were willing to tell us, 39% had completed either elementary or secondary school education and 13% had college level education. About 40% have been living in their current houses longer than 20 years. The majority of residents (92%) got their drinking water from private drilled wells while only 4% obtained water from hand dug wells. Many respondents did not know how deep their wells were, but some (27%) indicated that depths varied from less than 20 feet to over 500 feet. The survey results also indicated that 39% of the wells tested had no wellhead protection, and 53% of the respondents were unsure if their wells

had protection. A more general survey on the use of wellhead protection in Virginia's rural waters has revealed that most wellheads are either severely damaged or grossly neglected (IEN, 1992). Almost all respondents stated that they had not experienced any shortage of water from their wells. The plumbing in these rural houses was mostly copper (23%) and plastic (44%), with a few houses having a combination of the two; however, some (27%) did not know the type of plumbing they had in their houses.

A great majority (86%) of the homeowners in the sampling area did not use any kind of treatment technologies prior to consuming their well water, even though 42% of the residents described their water as being turbid or colored with observable particulates. Only 29% of the survey results showed that residents had their water tested at least once in the last five years for water quality indicator chemicals and bacteria, which meant about 70% of the household never had their water tested. Approximately 52% of the respondents saw no staining of utensils used in carrying water, but some (14%) have seen evidence of staining due to rust, blue-green, or brown sediments. The observed rust (staining) might be due to the high iron and/or manganese content of the soil in the region, or it could be indicative of the presence of some oxidizing bacteria or blue green algae in the wells. Questions dealing with environmental impacts were not answered satisfactorily. For instance, when people were asked if they had above or below ground fuel tanks, 43% responded as having neither. Moreover, those that admitted to having storage tanks didn't know how far the fuel tanks were from the drinking water wells. Most (67%) of the respondents did not have a designated storage facility for fertilizers and pesticides outside their houses. Normally, separate structures are recommended for storing such chemicals in farms to avoid food and drinking water contaminations. An encouraging aspect of the survey was revealed when a great majority (93%) of the homeowners did not dispose grease, oil, or leftover household chemicals down the drain. Therefore, the likelihood for petroleum-based chemicals entering the drinking water wells would be negligible.

Among the water parameters tested in the well waters, survey responses showed that total coliforms (TC) fecal coliforms (FC), nitrate (NO_3), fluoride (F1), and iron (Fe) were of concern (Figure 2). Demographically, TC counts were the highest in water samples taken from black homeowners than other demographic groups. However there was no significant difference in TC counts when compared by income levels. Comparison by education level showed samples from households with high school education had the worse contamination of TC. The type of plumbing used in houses had significant effect on TC counts, whereby plastic piping was more conducive for TC than either copper galvanized metal or lead piping. Even though 22 samples showed fecal coliforms count beyond the USEPA guideline (Table 1), these values were not discernable when compared by demography, income level, education or the type of plumbing used. However, among the 185 water samples collected from the Coastal Plain region, 71, 22, and 12 samples exceeded the EPA limits for Total coliform, fecal coliform, and *E. coli*, respectively (Table 1). Survey results also indicated that NO_3 , F1, and Fe counts were affected by well age, well depth, well type, and well casing (Figure 3). Nitrate level was predominantly higher in shallow wells (50 feet depth) than deeper (>100feet). However, variations in F1 and Fe levels were not too different regardless of depth. Well age showed variability in NO_3 levels, whereby wells drilled between 1970 and 1990 indicating high NO_3 content in water samples followed by older wells (1950 – 1970) and the more recently drilled wells (1990-2000). Interestingly enough, hand dug wells showed less NO_3 contamination compared with drilled

wells. Well casing had greatly reduced NO₃ contamination in water samples. The same is true for Fl and Fe contents.

CONCLUSION

After analyzing the chemical and microbiological qualities of the drinking water samples, it was found that drinking water resources at underserved farms located in certain parts of the Coastal Plain of Virginia are in serious need of water safety risk prevention. The overall results showed that of the 185 locations sampled and tested, coliform, fecal coliform, and *Escherichia coli* organisms were detected in about 33, 10, and 5% of the samples, respectively. Wells in close proximity to agricultural fields would show higher nitrate and phosphorus levels. Animals that roam in the vicinity of drinking water wells, geographic location of septic tanks, fertilizer storage, and manure disposal and storage facilities on the farm are potential sources of well water contamination.

The current findings indicated the presence of both biological and chemical contaminants in wells, although the data are insufficient to confirm that such contaminants would be detrimental to human health. A survey questionnaire filled out by farmers indicated that most complained about the color, taste, and turbidity of the water supply. Installation of filtration system could alleviate the problem, however, underserved farmers unlikely have the financial resources for such facilities. Nonetheless, results found from this study were provided to the homeowners along with additional information about educational and other resource assistance.

Identification of sources of contaminants to drinking water resources in these communities and finding solutions to the problems are sorely needed. Most of the residents of such communities lack sufficient income to afford expensive water treatment facilities. Consequently they are in great need of either state or federal assistance to overcome the problem. In addition, water quality education needs to be made available to the underserved communities to better protect drinking water resources.

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LOW PRESSURE PROPAGATION ANALYSIS IN AN EXPERIMENTAL PLUMBING RIG

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KEY WORDS: contaminant intrusion, pressure transients, plumbing systems, water hammer.

ABSTRACT

America's water distribution infrastructure system is old and deteriorating. A water system with its myriad appurtenances (including pumps and valves and tanks) is susceptible to hydraulic transients resulting in high and low pressure waves alternatively passing through the network. While both kinds of pressures structurally tax the already weak system, there is copious evidence to intrusion of contaminants into the drinking water pipes from the pipe exterior environment as the low pressure is unable to withstand the high pressure push. These contaminants are bound to enter into the house as the home plumbing system is a passive recipient from the water main. The major municipal system is readily recognized as a vast infrastructure system of nearly 1,409,800 km of piping within the United States, the minor plumbing system that is at least 5 to 10 times larger is generally not well addressed. An experimental plumbing rig is set up within the hydraulics lab of the Civil and Environmental Engineering (CEE) department to mimic the range of street level pressures encountered in a real system. This paper addresses how a transient triggered in a plumbing system within a house can impact the street lateral with a possible suction effect. A LabVIEW, an engineering software from National Instruments, based data acquisition system (DAQ) along with pressure transducer and flowmeter are employed to capture the variations in pressure and flow patterns at the service lateral. The analysis is expected to provide guidelines for a safer design of a household plumbing system and will also yield recommendations for developing better design and maintenance practices for the municipal system.

INTRODUCTION

The growth in bottled water consumption, point of use water treatment (including various kinds of filters, distillation, ion exchange, and reverse osmosis) indicates citizens' concern regarding the quality of distributed water at the tap. Cost advantages of public water supply, higher maintenance problems related to the point of use devices and relative marginal improvement in water quality result in the municipal water supply remaining as the top-ranked water supplier for maintaining established drinking water standards. It was widely believed that because a drinking water distribution system is pressurized the water can only come out. However, there is evidence that pump trips, opening and closing of fire hydrants, valve closures or malfunctioning, pipe break, and sudden change in demand and resonance can induce significant transients that lead to a lower pressure within drinking water piping and a greater external pressure can lead to intrusion. Tests of surrounding soil and pipe specimens from repair locations show the presence of pathogens. In 2000 alone in the US 6,988 water systems affecting about 10.5 million people violated drinking water standards for microbial standards.

Le Chevallier et al. (2003) define intrusion as the specialized backflow situation in which nonpotable/contaminated water from the environment outside of the distribution piping flows into the pipe through an available opening. Kirmeyer et al. (2001) and Friedman et al. (2004) point out physical mechanisms grouped as *transitory contamination* due to low pressure propagation in the system permitting a push-through of contaminants from the exterior surroundings with a higher pressure, *cross connection* between potable water system and a source that can potentially introduce contaminants into the potable water, *pipe break, repair and installation* that expose the distribution system to the externalities as routes of entries. Storage facilities both covered and uncovered, purposeful contamination, growth and resuspension serve as additional sources for pathogen intrusion. Karim et al. (2003) reported bacteria and viruses in 66 soil and water samples collected next to drinking water pipelines in eight utilities in six states. Friedman et al. (2004) documented intrusions and low pressures of the order of negative 10 psi (gage). They also emphasize that the intruded contaminant is not re-extruded out of the pipe but a portion of it is carried downstream. Distribution mains downstream of pumps, high elevation areas, low static pressure zones, areas far away from overhead tanks, segments of pipes upstream and downstream of active valves in high flow areas are the most susceptible for low or negative pressures. Locations with frequent leaks and breaks, high water table regions, flooded air vacuum valve vaults, and high risk cross connections have the highest potential for intrusion. Gullick et al. (2004) observed most surge events as the result of pump operations and outages.

In this paper we discuss efforts needed to simulate water hammer related transients within a plumbing system using an experiment rig and examine possible contamination scenarios at the household level. Since the home plumbing system is a passive recipient from the water mains, if there is contamination at street level, it is bound to enter into the house. We plan to explore varying street level boundary conditions and the conditions inside of the house, whether the low pressures can be induced within the plumbing system and the street level lateral becomes an intrusion point for the contaminants from the front lawn and the surrounding in general.

METHODS

Theory

The analysis of plumbing systems differs markedly from the analysis of municipal systems. It is a typical practice to treat the flow in municipal systems as steady flow and the continuity and energy equations are solved for head (H) and flow (Q); however, in plumbing systems because the demands last for hardly a few minutes transient analysis is more appropriate and the continuity and momentum equations are solved. As mentioned, the water hammer is a transient flow phenomenon introduced in pipe flow systems by suddenly obstructing the flow. As a consequence, there is a pressure rise and fall and the pattern is repeated until the transients decay. For completeness, we present the water hammer equations here as:

$$\text{Continuity equation: } \frac{\partial p}{\partial t} + V \frac{\partial p}{\partial x} + c^2 \rho \frac{\partial V}{\partial x} = 0$$

$$\text{Momentum equation: } \frac{\partial V}{\partial t} + V \frac{\partial V}{\partial x} + \frac{1}{\rho} \frac{\partial p}{\partial x} + g \sin \alpha + \frac{f}{2D} V|V| = 0$$

in which: p = pressure, V = velocity, c = wave speed, ρ = density, g = acceleration due to gravity, α = angle of inclination of pipe, f = friction factor, D = diameter, x = spatial dimension, t = time. These equations have to be solved for a pipe network that incorporates suitable interior boundary conditions for appurtenances such as valves, pumps, and junctions of several pipes and external boundary conditions for street level lateral, tanks, and faucets. Wylie and Streeter (1993) provide detailed accounts of the solution methodology. The solution of these equations yields the pressure, $p(x, t)$ and velocity, $V(x, t)$ as functions of x and t with x – dimension is taken along the length of the pipe. The pressure can be high positive and negative and the velocity can be negative indicating flow reversal.

Experimental rig

Three scenarios, referred to as *Transient I, II, and III* (see Figure 1) that can trigger a transient in a service lateral are considered. Transient I: In this case, transients are triggered due to actions initiated from inside of the house (e.g. by shutting off the valve, shower heads, or automatic on/off of solenoid valve from laundry room). Transient II and III: In these cases, transient causing actions are initiated from the municipal water systems (e.g. pump trips, opening and closing of fire hydrants, valve closures, pipe break, sudden change in demand and resonance, and malfunctioning of valves). Depending on the flow direction in the system, there are two different transients which influence the pressure variation at the service lateral.

An experimental plumbing rig is built in the hydraulics lab of CEE department at Virginia Tech to simulate these proposed three scenarios. Dotted oval shape indicates the plumbing system inside house and solid line is for municipal water system outside of a house (Figure 2). High sensitive sensors such as pressure transducers (with stainless steel diaphragm) and insertion type magnetic flowmeter were adopted to observe the spatial and temporal variation of the flow patterns. Table 1 gives specifications of the sensors employed including cost numbers. Three pressure transducers and one flowmeter were installed at locations of interest (Figure 2). The

signals from the sensors were collected simultaneously by LabVIEW based DAQ systems using sample and hold technique every 0.01 second. There is no delay in acquiring data from each sensor. To visualize the phenomenon inside the piping system, a 5.3' section of clear plastic pipe was installed (see Figure 2). A high definition Video camera is used to capture the gaseous cavitation.

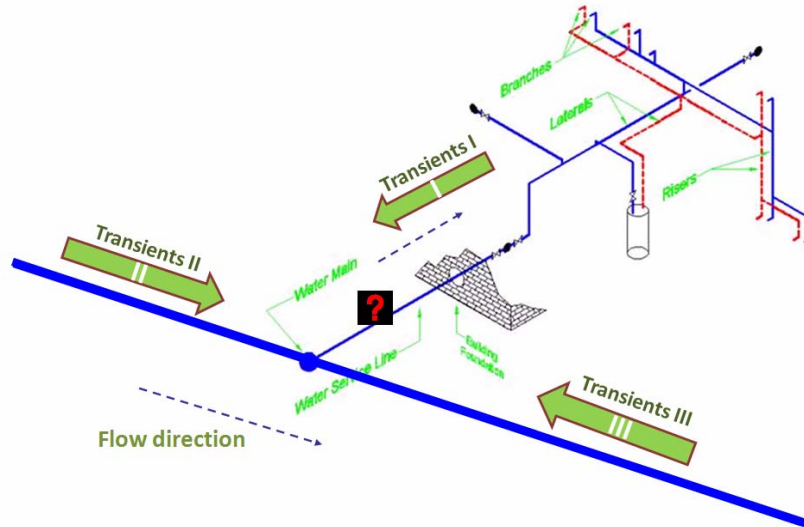


Figure 1. Scenarios of transients that can occur in drinking water distribution systems (modified from Ladd, 2005).

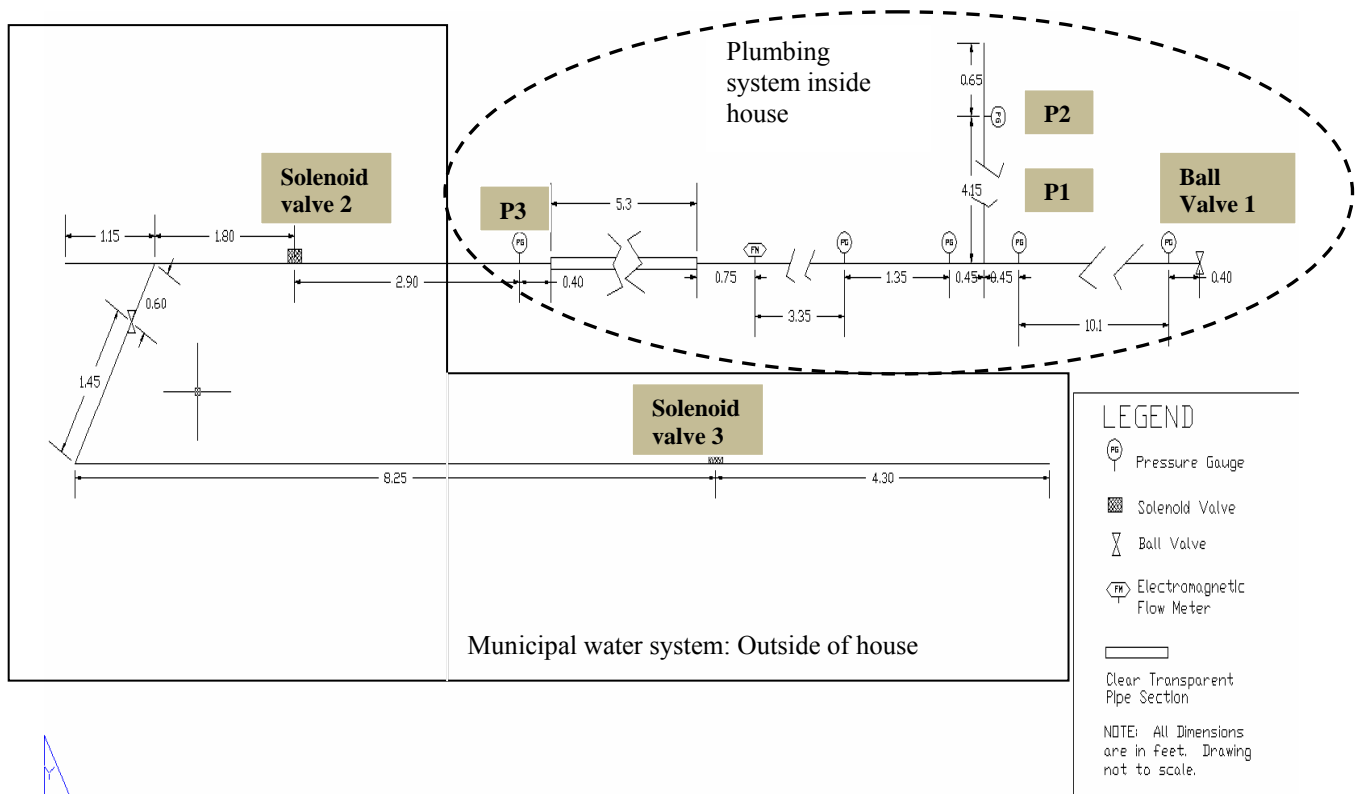


Figure 2. Schematic of the experimental plumbing system.

Table 1. Specification of the sensors.

Apparatus	Specification	Cost
Dynamic Pressure Transducer	Output: $\pm 5V$, range: ± 500 psi, sensitivity: 10mV/psi, resolution: 10mpsi, frequency ≥ 500 kHz,	\$729/each
Data Acquisition System and Modules	Sample and hold; <u>NI 9233 4-Channel</u> : 24-Bit, ± 5 V, 50 kS/s per channel; Analog Input Module; <u>NI 9203 8-Channel</u> : ± 20 mA, 200 kS/s, 16-Bit Analog Input Module	Total \$2,700 including modules and chassis
Flow meter	Output: 4-20 mA, 3/4" fitting, Flow range: 0.15-16 fps, fluid: drinking water, no pressure drop, one-direction	\$1,300 including power supply, meter, and fittings

RESULTS AND DISCUSSIONS

The experiment plumbing rig was connected to the water main in hydraulics lab to simulate a real household water supply situation. The average static pressure was 80 ± 5 psi when all the valves were closed. The authors opine that this fluctuation comes from water usage in nearby faucets or weak transients from the municipal systems. When the faucets were fully opened, however, the residual pressure was reduced to 45 psi. The level of residual pressure could be controlled by maneuvering the valve at the water main. Initially, we set the system in a steady state of 45 psi. Then we suddenly closed the solenoid valves. The valve closing time was known to be 0.3 seconds according to the manufacturer.

In figure 3, the pressure fluctuation at locations P1, P2, and P3 (see Figure 2) are shown when solenoid valve 2 is suddenly closed (in relative pressure difference from steady state (45 psi)) which simulate *Transient II* scenario. So, the pressure at three locations goes below 0 psi (gauge) for a fraction of a second. When the valve 1 is suddenly closed (scenario of *Transient I*), however, the pressure goes up to 100 psi (gauge) but it is not creating very low pressure as was observed in solenoid valve 2.

While experiments are on-going at the time of this writing, our preliminary findings are that water hammer triggered from inside the house cannot cause low pressures to cause contamination intrusion at the service lateral. However, transients triggered at the municipal system are highly likely to cause low pressure for possible intrusions. When the pressure goes below 0 (gauge pressure) psi, we observed gaseous bubbles forming for about 1 second in the clear plastic pipe section; then the bubble disappeared when pressure goes above gas saturation pressure (see Figure 4). It is again noted that water hammer induced homogeneous gaseous cavitation (from bulk liquid) was observed in this experiment. Experiment work is continuing to understand this phenomenon fully.

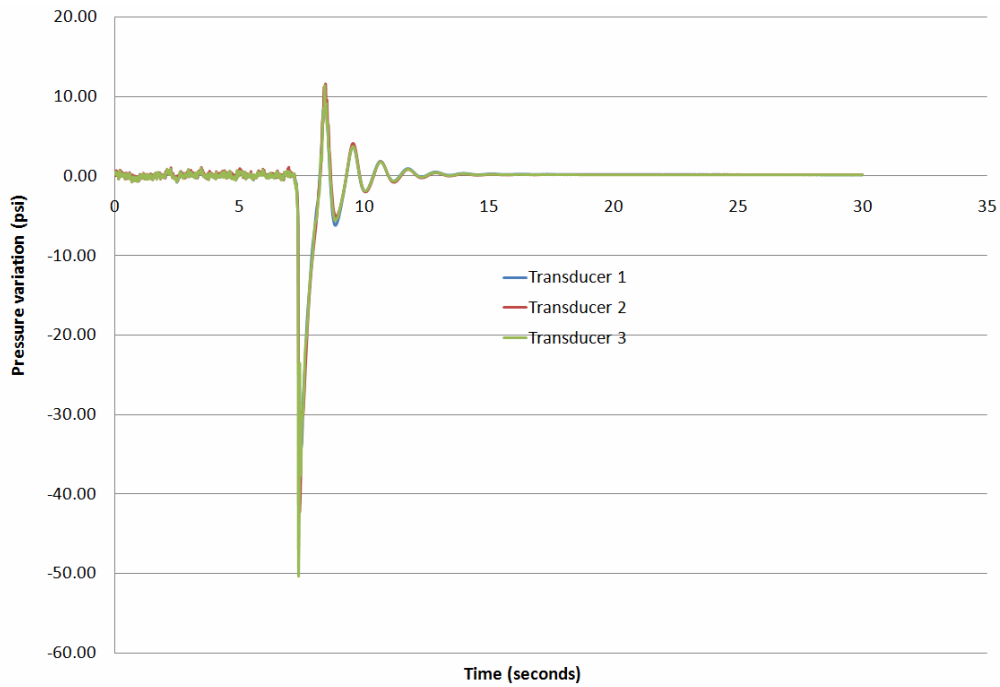


Figure 3. Pressure variation due to sudden closure of solenoid valve 2. Pressure shown in relative to 45 psi (gage) steady state conditions.



Figure 4. Water hammer induced homogenous gaseous cavitation.

SUMMARY

In this paper, pressure transient results from an experimental plumbing are presented corresponding to typical street level pressures encountered in a real water supply system. This research has potential to address the following issues: (1) how the low pressure wave moves through the plumbing system as the results of a street level transient and (2) how a transient triggered within a house can impact the street lateral with a possible suction effect. The authors are working on the numerical solution of corresponding governing equations and transient results will be compared with the observed values. An understanding of the pressure and flow pattern within a plumbing system should lead to a safer design not only within the house but also better design and maintenance practices for the municipal system.

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NEW CHLORINE RESISTANT MEMBRANES FOR REVERSE OSMOSIS AND NANOFILTRATION WATER PURIFICATION

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ABSTRACT

We have synthesized and characterized new disulfonated copolymer membranes based on chemistry that is entirely different from conventional post-polymerization sulfonation technology. Using direct copolymerization of disulfonated and other monomers, reproducible sulfonated copolymer membranes can be prepared for reverse osmosis and nanofiltration [1-3]. This synthesis method overcomes the problems of conventional sulfonation technology such as molecular weight reduction during post-polymerization sulfonation. This presentation will discuss the preparation and evaluation of several families of sulfonated polymers such as random or block copolymers, blends and crosslinked polymers. These sulfonated polymers or sulfonated polymer-coated thin-film composite membranes exhibit high tolerance to widely used chlorinated disinfectants, which is in contrast to conventional desalination membranes such as these based on aromatic polyamides or cellulose acetate. They also exhibit high water flux and moderate salt rejection. To delineate structure-property relations for these materials, solubility and diffusivity of water and various salts were evaluated for a series of sulfonated polymers. These intrinsic properties were compared with those of commonly used cellulose acetate and polyamide membranes. From this study, it is clear that high salt rejection is obtained via high diffusivity selectivity, and evidence is presented for the existence of “upper bound” correlations between water permeability and water/salt selectivity. This fundamental study of structure-property relations provides guidelines regarding material selection for reverse osmosis membranes. The new membranes also show good potential for arsenic removal and oily water separations. The authors like to acknowledge ONR N00014-06-1-1109, ONR- N00014-05-01-0772 and NSF-IIP-0650277 for their support and funding.

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2. Z. Zhang, G. Fan, M. Sankir, H.B. Park, B.D. Freeman, J.E. McGrath, *Synthesis of di-sulfonated poly(arylene ether sulfone) random copolymers as novel candidates for chlorine-resistant reverse osmosis membranes. PMSE Preprints, 95 (2006) 887-888.*
3. H.B. Park, B.D. Freeman, J.E. McGrath, *Novel sulfonated poly(arylene ether) copolymer membranes and their use in reverse osmosis: fundamental water and salt transport study, chlorine stability and anti-fouling characteristics, North American Membrane Society, Chicago, Illinois, May 12-17, 2006.*

FORMATION AND SORPTION OF TRIHALOMETHANES IN POLYMERIC PIPES USED FOR POTABLE WATER

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KEY WORDS: cPVC, PEX, chlorine, drinking water, trichloromethane

ABSTRACT

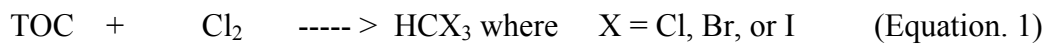
Polymeric pipes provide an alternative to metal pipes used in residential drinking water distribution systems. This study investigated the leaching of total organic carbon (TOC) in the presence of free chlorine and the potential formation and/or sorption of trichloromethane from plumbing materials including PEX-a, PEX-b and cPVC. The study involved new commercial piping filled with standard drinking water. The pipes were stored with a reference drinking water at room temperature for 72 hours to allow organic carbon to leach from the pipes to the water; this water was then reacted with 2 mg/L free chlorine and formation of trichloromethane was investigated. All three pipes leached a significant amount of TOC into the water ranging from 236 µg/L for cPVC, 249 µg/L for PEX-a to 376 µg/L for PEX-b. This TOC reacted with the disinfectant residual to form trichloromethane. During a 72 hour contact period of the leached TOC with free chlorine, the trichloromethane concentrations reached 11 µg/L for cPVC, 13 µg/L for PEX-b to 16 µg/L for PEX-a. New pipes were also filled with drinking water containing 25 µg/L of trichloromethane to measure sorption of this compound during a 3-4 day contact period. During the trichloromethane sorption phase, PEX-a and PEX-b sorbed 5-10 µg/L trichloromethane while cPVC had no change in concentration. This report contains more detailed information and data supporting formation and sorption capabilities of PEX-a, PEX-b and cPVC piping for trichloromethane.

INTRODUCTION

The American Water Works Association (AWWA) defines potable water as “water free from disease-causing organisms, free from minerals and organic substances that may produce adverse physiological effects and aesthetically pleasing with respect to turbidity, color, taste and odor” (AWWA, 1990). In order to make most water potable, i.e. “free from disease-causing organisms,” the process of disinfection is implemented which is defined as, “...killing, selectively if necessary, those living organisms that can spread or transmit infection through or in water” (Fair et al. 1968). Although there are various methods of disinfecting water, chlorination, one of

the first mass disinfection methods documented back in 1908, has become “the most cost-effective means to prevent the spread of waterborne infections and has been a common public health method for almost a century” (AWWA, 2006; Mills et al., 2000). Disinfection serves its purpose as safeguarding drinking water, but chlorination creates potential threats to community health (Bukaveckas et al., 2007).

Most public water supplies use chlorination as a method of disinfection, which potentially produces water containing chloroform (trichloromethane) and other trihalomethanes (THMs) (Lindstrom and Pleil, 1996). Trichloromethane, bromodichloromethane, dibromochloromethane, and tribromomethane are four chemicals that form when chlorine-based disinfectants react with naturally occurring organic matter in water (Davis and Masten, 1972). THMs are formed from the following reaction in which organic carbon reacts with free chlorine, (Equation 1)



where TOC is total organic carbon that may come from the water distribution system or be naturally occurring fulvic and humic acids in the environment.

Trichloromethane is a widespread concern because it occurs in the highest concentration in drinking water and is known to be carcinogenic to laboratory animals and potentially humans (Nazir and Khan, 2006; Health Canada, 1999). In addition to the risk of cancer, disinfection by-products (DBPs) have also been associated with spontaneous abortion in pregnant women, low birth weight and multiple birth defects (Mills et al., 1998). The health risks associated with disinfection by chlorination has caused the Environmental Protection Agency (EPA) to lower the maximum contaminant level for total trihalomethanes (TTHMs) from 100 µg/L to 80 µg/L (EPA, 1998). In the drinking water industry, a protocol called National Sanitation Foundation-61 (NSF-61) is used to test materials in contact with drinking water for their propensity to leach USEPA regulated contaminants including THMs.

The formation of THMs has potentially serious health risks, but the sorption of THMs into the distribution system materials creates another health risk pertaining to THM analysis. A 1999 study demonstrated that consumption and exposure to trichloromethane in a typical house could be into these categories: 68% consumed was from inhalation, 18% from dermal and 14% from digestion (Khanal, 1999). This information is essential in assessing human exposure to possible contaminants that result from materials use. If THMs sorb into the pipe materials during NSF-61 testing, then the materials might artificially pass leaving a concern that THM sorption affects THM exposure. Heim and Dietrich (2007a) recently reported that trichloromethane was produced when chlorinated water was in contact with new epoxy lined copper pipes. They also found, in a different study dealing with new high density polyethylene (HDPE), that 44% of the trichloromethane in water sorbed into the pipe matrix (Heim and Dietrich, 2007b). These results will be combined with the results from this study to generate a matrix of information comparing the different properties and effects various polymeric pipes have on drinking water quality.

The goal of this study was to compare trichloromethane production and sorption for different polymeric pipes (c-PVC, PEX-a and PEX-b) used in drinking water distribution systems using an NSF-61-related procedure. The study had two distinct parts: 1) investigate the potential

formation of THMs by reacting free chlorine with organic carbon leached from the material; 2) investigate the potential of the material to sorb pre-existing trichloromethane.

METHODS

National Sanitation Foundation (NSF) approved PEX-a, PEX-b and cPVC pipes were acquired from local building supply companies. Reference tap water, that represented tap water similar to those found in the Eastern United States, was prepared from Nanopure® (Barnstead Nanopure Filter) water combined with six salts equaling 8 mg/L Mg^{2+} , 46 mg/L SO_4^{2-} , 20 mg/L Na^+ , 0.05 mg/L Al^{3+} , 11 mg/L Ca^{2+} , 2.6 mg/L SiO_3^{2-} , 4 mg/L K^+ , 1.4 mg/L NO_3^- as N, 0.002 mg/L PO_4^{3-} as P. No other organic matter was added and the pH was adjusted to 8.

The pipes were used as obtained from the supply store and they were not pre-disinfected or rinsed with chlorinated water. Three foot lengths of 5/8 " diameter pipe were cut and filled with reference tap water leaving no headspace and then capped with Teflon-lined caps. The pipes were stored at room temperature and pressure for 72 hours. After 72 hours, the water was drained and an aliquot placed in 20 VOA vials to measure initial total organic carbon (TOC). Other aliquots were placed in 40 mL VOA vials and prepare for testing the formation of THMs by reacting with free chlorine. Chlorine was added to a concentration of 2 mg/L free chlorine; solutions were prepared with sodium hypochlorite. Triplicate samples of TOC leachate and free chlorine were reacted for 6, 24, an, 48, and 72 hours; the reaction was quenched with liquid sodium thiosulfate. Samples were analyzed for THM concentrations and residual chlorine and TOC after the reaction. This process was repeated for a second contact period with the same pipe lengths to determine how THM formation changes over multiple flushes in the same pipes.

An investigation of THM sorption into the pipe wall was conducted by cutting 6" lengths of the different pipes and filling them, leaving no headspace, with reference tap water containing 25 $\mu\text{g/L}$ THMs. Amber VOA vials were also filled with the same water as control samples. These pipes were stored at room temperature and samples were obtained at 6, 24, 48 and 72 hours. THMs were determined and the difference between the initial and final concentrations was used to calculate sorption capability.

Total organic carbon was measured using a Sievers 800 Portable TOC analyzer, and following the guidelines described in Standard Method 5310C (APHA, 2005). TOC tests were done at 0 hours and 72 hours to determine how the total organic carbon concentration changed over time. All reported TOC concentrations were corrected for the background TOC level in the Nanopure® water which was typically 300-500 $\mu\text{g/L}$.

Free chlorine levels and residuals were determined by measuring the concentration at 0, 6, 24, 48 and 72 hours to determine the amount that reacted over time. Concentrations were determined using DPD power pillows, 10 mL of the sample, and a HACH Pocket Colorimeter II or HACH DR/2400 Portable Spectrophotometer. Chlorine naturally decays over time so the chlorine values for consumption by the pipe materials were adjusted by subtracting the chlorine decay for control samples of reference water with no additional TOC for a pipe material.

THMs were measured according to USEPA Method 502.2 or Standard Method 6232D (APHA,

2005). The instrument used was a Tremetrics 9001 gas chromatograph with a Hall 1000 detector, Tekmar 3000 purge trap and concentrator and Tekmar 2016 Purge Trap auto-sampler. THM were measured at 0, 6, 24, 48 and 72 hours to determine THM formation and sorption capability over time.

Hypothesis testing was performed using standard student t-test methods with $\alpha=0.05$. For the p values, null hypotheses were accepted for $p > \alpha$ and rejected for $p < \alpha$, accepting the alternative hypotheses.

RESULTS

All three pipes, PEX-a, PEX-b and cPVC leached organic carbon as shown in Table 1. The large standard deviations show that not every pipe section was made the same and that there is substantial variability in the manufacturing of these polymeric pipes. There was no statistical difference ($p > 0.06$) for TOC concentrations between PEX-a and PEX-b during the first contact periods meaning that all materials leached about the same amount of TOC. During the second 72 hour contact period of reference water and pipe, PEX-a and PEX-b both leached statistically ($p \ll 0.05$) more TOC than cPVC. Overall PEX-a and PEX-b leached more TOC than cPVC leaving the potential formation of more THMs depending on chlorine consumption.

Table 1. TOC leached into water from polymer pipe materials.

Material	Total Organic Carbon (TOC) $\mu\text{g/L}$ ^a	
	1 st 72 hr Contact Period	2 nd 72 hr Contact Period
PEX-a	249 \pm 52 ^b	207 \pm 6.5
PEX-b	376 \pm 92	202 \pm 5
cPVC	236 \pm 59	79 \pm 9.5

^a Concentrations were adjusted for background levels of TOC of the controls which were reference water stored in glass vials; the controls ranged from 300-500 $\mu\text{g/L}$ TOC.

^b Values are means \pm standard deviation for n=3

Chlorine decay in leachate water from the three pipes was faster than for the control samples which contained only reference water and free chlorine in glass. Water exposed to all three pipes, PEX-a, PEX-b and cPVC, showed significantly less chlorine residual than the controls. Figure 1 also shows that during the first contact period with chlorinated water, the pipes consumed chlorine. In the first contact period, PEX-a consumed more Cl_2 than PEX-b or cPVC. Chlorine consumption results from reaction with the pipe wall and with TOC leached. When the pipes were contact with chlorine for a second period, less chlorine was consumed and the amount was in the range of 0-0.4 mg/L Cl_2 (data not shown).

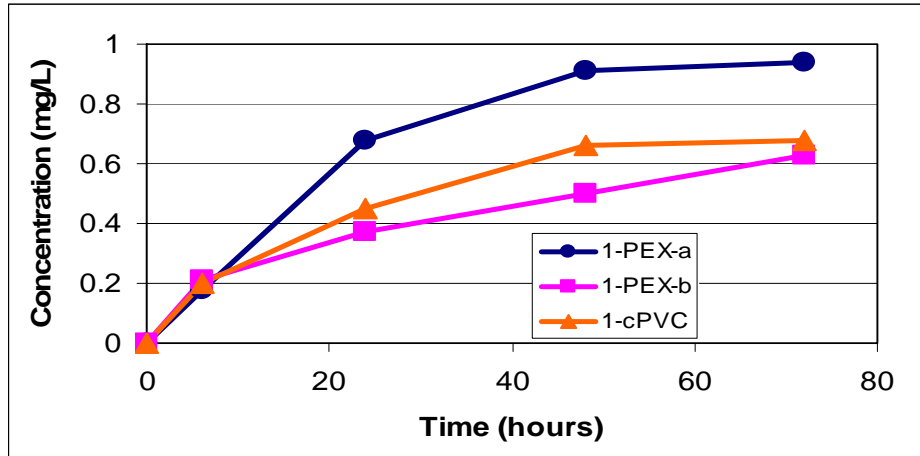


Figure 1. Chlorine consumption by pipe material for first contact period of pipe with chlorinated water. The values for chlorine consumed by the pipe materials were adjusted for background chlorine decay in the controls. Initial chlorine dose was 2 mg/L Cl_2 .

Trichloromethane was the only THM detected as a result of the reaction of leached TOC and free Cl_2 . The concentrations of trichloromethane formed during the first flush were all statistically larger than zero. Although they all produced trichloromethane, Figure 2 shows the differences in the amounts produced and how they formed over time. Statistically ($p > 0.115$) TOC leached from PEX-a, PEX-b and cPVC pipes produced the same amount of trichloromethane over the 72 hour contact period.

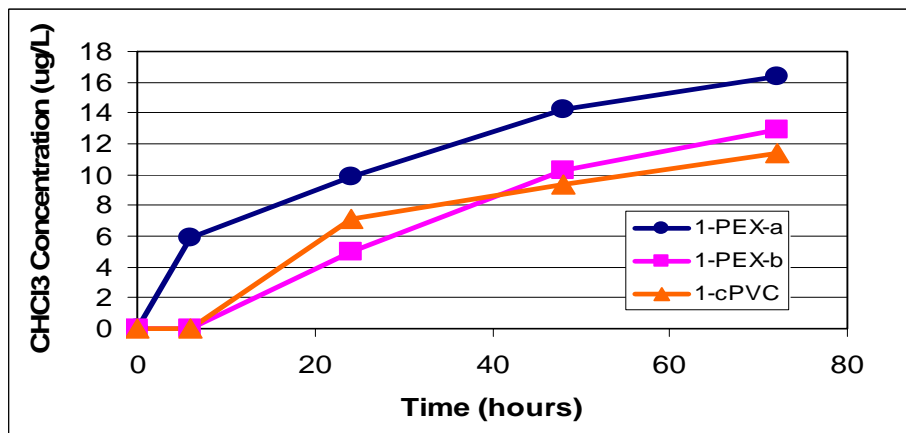


Figure 2. Trichloromethane formation from reaction of leached TOC with free chlorine.

Trichloromethane was the only THM formed during the formation stage, likely due to the absence of bromide in the reference water. Trichloromethane was the only THM investigated in the sorption phase. As can be seen in Figure 3, when trichloromethane containing water is stored in the pipes, there is a slight tendency of the initial concentration of trichloromethane to decline over indicating sorption to the pipe material. Statistically ($p < 0.025$) PEX-a and PEX-b sorbed trichloromethane. PEX-a had a significantly ($p = 0.027$) smaller remaining concentration of trichloromethane than PEX-b meaning PEX-a sorbed more trichloromethane than PEX-b. cPVC

had no statistical ($p=0.099$) change in beginning and ending trichloromethane concentration meaning no sorption took place.

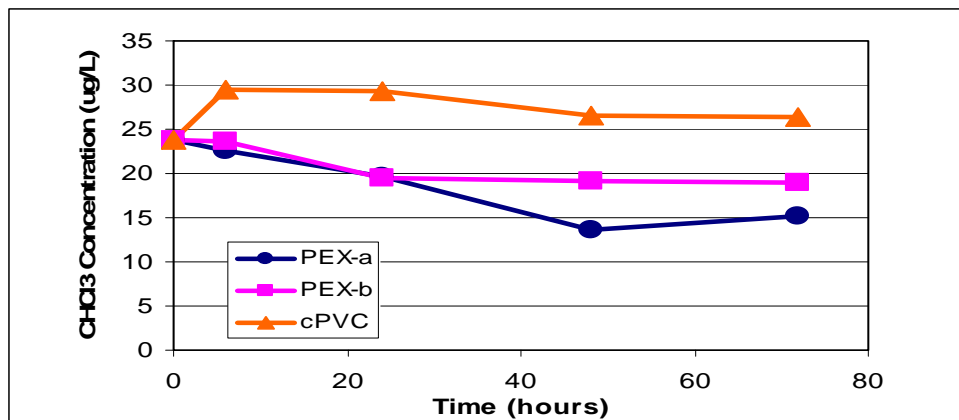


Figure 3. Change in aqueous concentration of trichloromethane due to contact with pipe material. Declining concentration indicates sorption.

DISCUSSION

This research demonstrates the impacts that PEX-a, PEX-b and cPVC have on the formation and sorption of trichloromethane in tap water. Sections of new commercial 5/8" pipe of PEX-a, PEX-b or cPVC all leached a significant amount of TOC in reference water throughout a 72 hour contact period. When this leached TOC was separately reacted with free chlorine, PEX-a, PEX-b and cPVC all consumed a significant amount of free chlorine during a 72 hour contact time that was coupled to a THM formation study. PEX-a, PEX-b and cPVC all formed significant concentrations of trichloromethane. A range of 8-16 $\mu\text{g/L}$ trichloromethane was formed by these materials during a 72 hour reaction time. In another experiment that investigated trichloromethane sorption into the pipe material, PEX-a and PEX-b were shown to sorb similar $\mu\text{g/L}$ concentrations of trichloromethane. cPVC had no sorption ability over a 72 hour time period.

This research demonstrates the effect that new commercial plumbing materials have on the formation and sorption of trichloromethane. This information is important for consumers to assess when choosing and installing plumbing materials for use in their homes.

ACKNOWLEDGEMENTS

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Session II-B

Measuring and Modeling Bacteria in Aquatic Systems

- Comparison of Membrane Filtration Using m-Colibblue Media with Coliscan Easygel for Enumerating E. coli and Total Coliforms in Donaldson Run
- Uncertainty of Predicted In-stream Fecal Coliform Concentration for TMDL Development Using HSPF: Generalized Likelihood Uncertainty Estimation Approach
- Assessing Alternative Methods Used to Model Fecal Coliform Direct Deposit

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COMPARISON OF MEMBRANE FILTRATION USING M-COLIBLUE MEDIA WITH COLISCAN EASYGEL FOR ENUMERATING E-COLI AND TOTAL COLIFORMS IN DONALDSON RUN

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KEY WORDS: E coli, water monitoring, coliforms

ABSTRACT

Coliscan Easygel from Micrology Laboratories is becoming a common technique to enumerate E-coli and total coliforms in surface water. We compared this technique with the EPA approved method of membrane filtration using m-Colibblue broth from Hach. Four sites on Donaldson Run were tested on multiple days with different conditions of temperature and precipitation. Three repeats were performed for each sample. In general, the m-Colibblue method produced higher counts of E-coli and the Coliscan method produced higher counts of total coliforms with the sites varying in which technique produced higher counts. Some colonies from each technique were tested with Enterotube II from Becton Dickinson and both techniques were found to produce false positives and false negatives. Both techniques were also compared by testing with Watewatr Coliforms samples from Environmental Resource Associates and the two techniques agreed for the standard samples.

INTRODUCTION

In 2005, the Virginia Department of Environmental Quality began a citizen monitoring program for *E. coli* using Coliscan Easygel (Virginia Department of Environmental Quality). The technique was selected by the Virginia DEQ for its ease of use and low cost. One area of monitoring included Four-Mile Run in Arlington County. Donaldson Run is another stream in

Arlington County, which is in the Potomac River watershed, but not in the Four-Mile Run Watershed. Donaldson Run is in a suburban area. It begins just down hill from an Arlington County mulch pile and salt depot. The area sampled was originally developed in the 1950's. Run off, from the streets and houses, is directed into the stream, and the stream is impaired by both polluted run off and hydrologic impacts. (EPA) Membrane filtration methods had previously been used to enumerate *E. coli* and total coliforms in Donaldson Run. Coliscan Easygel would be a less expensive method to test for bacterial contamination.

METHODS

Four sites were selected on upper Donaldson Run. Site 1 has evidence of anaerobic conditions and always has the lowest dissolved oxygen level. Site 2 has evidence of Iron bacteria *Sphaerotilus natans* and *Gallionella ferruginea*. Site 3 is just past the convergence of two branches. Site 4 is after a culvert on Vermont Ave. Samples were collected in sterile Nalgene containers. The water temperature, dissolved oxygen, pH, nitrates and phosphates were also recorded. The water samples were taken to the microbiology laboratory at Marymount University and tested by Coliscan Easy Gel and Membrane Filtration within one hour of sampling.

For Coliscan-Easy Gel from Micrology Laboratories, a 1-5 mL sample was pipeted into the media. The sample size was determined by estimating the bacteria present, for a rainy day the smaller sample size was used. After they were mixed, the sample plus media were poured into a Petri dish, which caused the media to solidify. The media uses the enzyme galactosidase to differentiate coliforms from other bacteria and glucuronidase to distinguish *E. coli*. *E. coli* will give a blue + red = purple colored colony. Other coliforms will give a red colony and noncoliforms will give a white colony. The samples were incubated at 37°C for 24 hours.

For the membrane filtration technique, 100 mL of sample were aseptically filtered onto a membrane, which was placed on an absorbent pad saturated with m-colibblue media purchased from Hach, in a Petri dish. The media inhibits non coliforms with methylene blue, erythromycin and an azide. Coliforms are highlighted by a non selective dye, 2,3,5-triphenoltetrazolium chloride, which produce a red color. *E. coli* again produce a blue color as the result of a glucuronidase reaction. The samples were incubated at 37°C for 24 hours.

The two techniques were also compared by using known *E. coli* samples purchased from Environmental Resource Associates (ERA). The samples were from the same lot and contained *Escherichia coli* ATCC 35218. The membrane filtration sample was divided into two 50mL samples. 50 mL of phosphate buffer was added to each. The sum of the *E. coli* measured was 13 coliforms/100 mL. The other sample from the same lot was divided into 20, 5mL samples and tested using the Coliscan method. A total of 7 coliforms was found over the 20 plates. Both of these values fell within the QC limits (6-92) for Fecal Coliforms determined by ERA.

Colonies from both the Coliscan method and the membrane filtration method were also tested with Enterotube II diagnostic system from Becton-Dickenson. The Enterotube II performs 12 different biochemical tests when inoculated with bacteria.

RESULTS AND DISCUSSION

The results for counts of *E. coli* and the other coliforms for both membrane filtration and the coliscan techniques are presented in Table 1, with the date the experiments were performed. Also included on the table is the precipitation for the date of the test. For most of the test dates no precipitation occurred for the previous 48 hours. However, for March 29, 2006 light rain occurred 24 hours previously, and on June 28, 2006 sampling was done during heavy rain.

Table 1. Results for *E. coli* and other coliforms for both techniques.

<i>Date</i>	<i>Precipitation</i>	<i>E. coli</i>		<i>Other coliforms</i>	
		Filtration	Coliscan	Filtration	Coliscan
Site 1					
2/10/2006	No	253	11	511	1166
2/15/2006	No	350	100	623	3867
3/29/2006	24 hours	2	0	8	489
6/28/2006	Yes	159	100	623	600
9/26/2006	No	784	589	414	744
10/25/2006	No	511	133	1723	600
Site 2					
2/10/2006	No	127	0	555	211
2/15/2006	No	316	67	297	633
3/29/2006	24 hours	4	0	33	33
6/28/2006	Yes	11	516	61	1817
9/26/2006	No	75	67	33	333
10/25/2006	No	311	0	1172	333
Site 3					
2/10/2006	No	58	0	285	22
2/15/2006	No	39	0	1109	67
3/29/2006	24 hours	13	0	79	433
6/28/2006	Yes	190	3517	618	3916
9/26/2006	No	385	67	79	516
10/25/2006	No	238	100	959	233
Site 4					
2/10/2006	No	19	0	70	11
2/15/2006	No	29	0	949	0
3/29/2006	24 hours	8	0	79	2198
6/28/2006	Yes	201	966	677	2733
9/26/2006	No	0	0	0	0
10/25/2006	No	0	0	0	0

In general, Coliscan gave a lower number for *E. coli*, than the membrane filtration technique. The exceptions are sites 2, 3, and 4 on June 28, 2006, which was the warmest sampling day and the only precipitation event. Coliscan, in general, gave a larger number of other coliforms than

the membrane filtration method, the exception being the samples taken on February 15, and October 25. The results of the Enterotube II test of the various colonies are given in Table 2 for Site 1, Table 3 for Site 2, Table 4 for Site 3 and Table 5 for Site 4. The colonies for the Enterotubes were from the June 28, September 26, and October 26, 2006 samples.

Table 2 Results from Enterotube samples for Site 1.

Membrane Filtration Red	Membrane Filtration Blue	Coliscan Red	Coliscan Blue
undetermined	undetermined	undetermined	E.coli
Hafnia Alvei	undetermined	Citrobacter freundii	
Enterbacter aerogens	undetermined	Klebsiella pneumoniae	E.coli
undetermined	undetermined	undetermined	E.coli
undetermined	E.coli	undetermined	Enterobacter Cloacae *
			Yersinia
			Enterocoloticia
undetermined	undetermined	Citrobacter freundii	E.coli
Enteric Group 60 *	undetermined	E.coli	E.coli *
Yersinia Kristensenii			Klebsiella Ozaenae
undetermined	E.coli	Klebsiella Ozaenae	E.coli
Enterbacter Cloacae	Actinobacter LWOFFII *	undetermined	E.coli
	Pseudomonas		
	Maltophilia 2k-1		

Table 3 Results from Enterotube samples for Site 2.

Membrane Filtration Red	Membrane Filtration Blue	Coliscan Red	Coliscan Blue
undetermined	Salmonella subgroup	Citrobacter freundii	E.coli
Salmonella cholerae	undetermined	Citrobacter freundii	E.coli
undetermined	undetermined	undetermined	undetermined
Serratia Fonticola *	undetermined	Citrobacter freundii	E.coli
Enterobacter Aerogenes	Salmonella SUBGP 3B (Arizona)	undetermined	E.coli
undetermined	undetermined	undetermined	E.coli
Enterobacter Aerogenes	E.coli	Shigella Serogroups A,B,or C*Enterobacter Agglomerans* Yersinia Pseudotuberculosis	E.coli
Enterobacter Cloacae	E.coli	Shigella Serogroups A,B,or C*Enterobacter Agglomerans* Yersinia Pseudotuberculosis	E.coli
Actinobacter LWOFFII *	E.coli	Actinobacter LWOFFII *	undetermined
Pseudomonas Maltophilia 2k-1		Pseudomonas Maltophilia 2k-1	

Table 4 Results from Enterotube samples for Site 3.

Membrane Filtration Red	Membrane Filtration Blue	Coliscan Red	Coliscan Blue
salmonella subgroup	Salmonella	Klebsiella ozaenae	E.coli
undetermined	undetermined	Klebsiella pneumoniae	E.coli
Salmonella	Salmonella	Serratia plymuthica	E.coli
undetermined	undetermined	Klebsiella ozaenae	E.coli
undetermined	E.coli	E.coli	E.coli
		Yersinia Pestis * Shigella	
		Serogroups A, B, or C * E.coli*	
		Klebsiella Ozaenae* Enterobacter	
Klebsiella Ozaenae	E.coli	Agglomerans* Yersinia	
Klebsiella Pneumoniae *		Enterocoliticia	E.coli
Enterobacter			
Aerogenes	undetermined	undetermined	E.coli
undetermined	undetermined	Citrobacter freundii	E.coli
			Enterobacter
			Cloacae *
			Yersinia
undetermined	E.coli	undetermined	Enterocoliticia

Table 5 Results from Enterotube samples for Site 4.

Membrane Filtration Red	Membrane Filtration Blue	Coliscan Red	Coliscan Blue
undetermined	Enterbacter aerogens	Enterobacter cloacae	E.coli
undetermined	undetermined	E.coli	Klebsiella pneumoniae
Enterbacter aerogens	Salmonella	undetermined	E.coli

Based on the results of the Enterotube II tests, the coliscan technique gives fewer false positives for E. coli than the membrane filtration technique with the m-Colibblue media. If the Enterotube II results confuse Klebsiella with E. coli, (Martin et. Al. 1971) then it is possible that the coliscan technique produces false negatives with E. coli showing red colonies. It is clear from the Enterotube II results that the membrane filtration technique produces a higher rate of false positives for E. coli than the Coliscan method.

CONCLUSIONS

Coliscan appears to work as well for high bacteria measures as membrane filtration. For lower numbers of bacteria, the small sampling volume decreases the sensitivity of the technique. Both techniques, however, give relatively high numbers of false positives, with the Coliscan technique give approximately 30% false positives and the membrane filtration technique giving more than 50% false positives. An extensive comparison of several methods with the EPA method 1603 for 223 samples showed reasonable agreement for the Coliscan technique. (O'Brien, 2006)

ACKNOWLEDGEMENTS

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UNCERTAINTY OF PREDICTED IN-STREAM FECAL COLIFORM CONCENTRATION FOR TMDL DEVELOPMENT USING HSPF: GENERALIZED LIKELIHOOD UNCERTAINTY ESTIMATION APPROACH

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KEY WORDS: fecal coliform, TMDL, uncertainty analysis, generalized likelihood uncertainty estimation, GLUE.

ABSTRACT

Generalized likelihood uncertainty estimation (GLUE) technique was used to estimate the distribution of input parameters for Mossy Creek watershed model developed using Hydrologic Simulation Program – FORTRAN (HSPF). These estimated distributions also referred to as “posterior” distributions were used to estimate uncertainty in predicted in-stream concentration of fecal coliform in Mossy Creek for two different pollutant load allocation scenarios. The results suggest that the numbers of violations of instantaneous fecal coliform criteria resulting from the two allocations scenarios are not significantly different from each other. However, when the 80% and 95% probability intervals for fecal coliform concentration were computed, the two allocation scenarios exhibit significantly different results from each other. The scenario with greater contribution of fecal coliform through direct deposit in the stream exhibited greater uncertainty in water quality concentrations compared to scenario with lower contribution from direct deposit. The numbers of violation of instantaneous fecal coliform criteria reported by 80 and 95% probability interval were also significantly greater for the allocation scenario with greater input from direct deposits. The results suggest that the added information about uncertainty in water quality modeling can be extremely important while assessing among different TMDL pollutant allocation scenario or watershed management plans.

INTRODUCTION

Water quality modeling is often used to develop watershed management plans like Total Maximum Daily Loads (TMDLs). A TMDL specifies the maximum amount of a particular pollutant a waterbody can receive and still meet applicable water quality standards. A margin of safety is often included within a TMDL to account for the inherent uncertainty present in water quality modeling which is often used to develop the TMDL. Uncertainty is always present in

modeling a natural system and is a result of limited knowledge of the system being modeled or stochastic variability (Beck 1987; Suter et al. 1987). However, there is a limited scientific guidance available to estimate the amount of uncertainty associated with water quality modeling which is often used to determine a TMDL. USEPA (2001) estimated that the costs of developing and implementing TMDL would reach more than \$1 billion per year for next several years. Indicator bacteria which are an indication of pathogen impairment being the second most widespread cause of water quality impairment will be responsible for a significant share of this expense (USEPA 2006).

Water quality modeling is often conducted using software that contains process based or a mix of process-based and empirical modules. The Hydrological Simulation Program – FORTRAN (HSPF) is commonly used in Virginia for modeling in-stream bacteria concentrations as a part of the TMDL development process. HSPF simulates various hydrologic and water quality processes, and produces a deterministic time-series of hydrology and water quality. The authors illustrated a two-phase Monte Carlo approach to estimate uncertainty in predicted in-stream fecal coliform concentration (Mishra and Benham, 2007) in a watershed model developed using HSPF. One of the other Monte-Carlo based approaches to estimate uncertainty in hydrologic modeling is the Generalized Likelihood Uncertainty Estimation or GLUE approach as proposed by Beven and Binley (1992). The GLUE approach is based on the premise that there is not one set of model parameter values that represent a “true” parameter set for a system. Instead, an assessment is made whether an input parameter set is an acceptable, or has a likelihood of being an acceptable simulator of the system.

In the GLUE approach, a large number of model runs are performed by generating different sets of parameters from what are called the “prior” distributions. The suitability of these parameter sets is assessed by comparing model simulations and the observed data for each model run. Based on this assessment, acceptable parameter values are used to compute a distribution of input parameters also called as “posterior” distribution, using Bayesian equation. The “posterior” distribution parameter sets can be used to conduct Monte Carlo simulations of the model and useful statistics can be drawn to validate the model or estimate uncertainty for a given prediction period. The GLUE approach has been used to conduct uncertainty analysis using different hydrologic software (Beven and Binley 1992; Freer et al 1996; Balin 2004). However, it has not been used to estimate uncertainty in water quality modeling. The objective of this research is to evaluate the uncertainty in predicted in-stream concentration of fecal coliform in a Virginia watershed for two TMDL allocation scenarios suggested in the TMDL developed for that watershed. The watershed model was developed using HSPF and the GLUE approach was used to estimate the uncertainty.

MATERIALS AND METHODS

Study area

The Mossy Creek watershed (4076 ha) located in Rockingham and Augusta counties in Virginia (figure 1) was selected for this research. Mossy Creek was listed as impaired in 1996 due to violations of Virginia’s Primary Contact Recreational standard’s fecal coliform criterion. The Department of Biological Systems Engineering (BSE) at Virginia Tech developed bacteria

TMDL for Mossy Creek (Benham et al 2004). The Mossy Creek watershed is characterized as a rolling valley with Blue Ridge Mountains to the east and the Appalachian Mountains to the west. The predominant landuse in the Mossy Creek watershed is agriculture. Mossy Creek was monitored monthly by the Virginia Department of Environmental Quality (DEQ) for various constituents between July 1992 and March 2003. BSE monitored Mossy Creek bi-monthly between February 1998 and December 2001. BSE-monitored constituents included fecal coliform, sediment, and various other chemical constituents. Daily flow measurements were conducted from May 1998 to December 2002 at the BSE monitoring site.

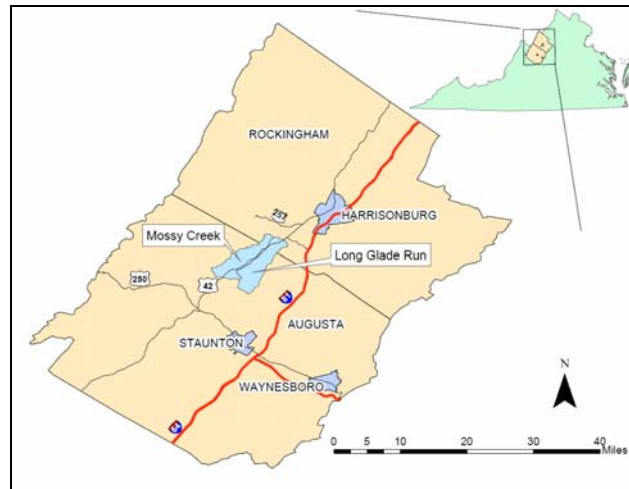


Figure 1. Mossy Creek watershed.

Mossy Creek watershed model

As part of the Mossy Creek TMDL study, a watershed model was developed using HSPF. The Mossy Creek watershed was divided into eight subwatersheds. Landuses were identified using aerial photographs from Virginia Department of Conservation and Recreation (VADCR) (table 1). Time-series inputs needed for the watershed (e.g., climate, deposition of fecal coliform directly into streams by livestock and wildlife, point sources inputs, and inflows from springs) was entered separately using “Weather Data Management (WDM)” file format. Rainfall data was collected from a BSE weather station located in the adjacent watershed Long Glade Run, some 2.5 km south of Mossy Creek. Other necessary climate inputs (e.g., solar radiation, wind and daily temperature) were obtained from nearby airport weather stations.

Table 1. Land use distribution of Mossy Creek watershed.

Landuse	Area (ha)	Percent of total area
Forest	1025.1	25.15
Cropland	556.0	13.64
Pasture	2347.6	57.59
Farmstead	55.0	1.35
Low Density Residential	87.0	2.13
High Density Residential	3.6	0.09
Loafing Lot	1.6	0.04

HSPF uses several hydrologic and water quality parameters to simulate the fate and transport of fecal coliform in a watershed model. Guidance to estimate the values of the hydrologic parameters is available from BASINS technical note 6 (USEPA, 2000). Calibration of these parameters is further conducted to estimate their values for a specific watershed model. Al-Abed and Whiteley (2002), and Lawson (2003) listed the hydrological parameters that are sensitive and are typically calibrated in the process of watershed modeling. For this research, most of the hydrological parameters were assigned a uniform or triangular distribution based on the typical lower and upper limits of parameters as illustrated in BASINS Technical Note 6 (USEPA 2000). GIS data was used to estimate the distribution of index to mean infiltration rate (INFILT). These distributions are termed as “prior” distribution as they reflect the prior knowledge of modeler regarding the system being modeled.

Water quality parameters

Simulation of in-stream fecal coliform concentrations by HSPF requires that the loading rates of fecal coliform to the land surface (ACCUM), the maximum possible accumulation of fecal coliform on the land surface (SQOLIM), and fecal coliform loading directly deposited in the waterbody be estimated. Other water quality parameters that are typically calibrated for water quality modeling are fecal coliform wash-off potential (WSQOP), and first-order decay rate (FSTDEC). For the Mossy Creek TMDL, ACCUM values and direct deposit of fecal coliform in waterbodies were calculated using the Bacteria Source Load Calculator (BSLC) (Zeckoski et al., 2005).

According to the Mossy Creek bacteria TMDL, dairy cattle, beef cattle, and poultry are responsible for more than 94% of fecal coliform production in the watershed, and pasture, loafing lots, and cropland receive largest amount of fecal coliform loading (Benham et al., 2004). Therefore, the uncertainty from only these pervious areas was considered in this study. Deterministic values for ACCUM calculated using the BSLC were used for the other landuses present in the watershed – forest, low and high density residential, farmstead and other impervious areas.

The ACCUM values depend upon several factors including species-specific feces production rates, fecal coliform fecal densities, fecal coliform die-off rates, and the fraction of time livestock are confined (Zeckoski et al, 2005). A comparison of the fecal coliform production rates of dairy cattle, beef cattle and poultry as cited in various TMDL reports and in the literature (ASABE Standards, 1998; Geldrich 1978, and Yagow 2001) shows that these values can vary by several orders of magnitude. We surmised that the uncertainty in the fecal coliform production rates from dairy cattle, beef cattle, and poultry would have an affect on the uncertainty in the model output and that it would mask the uncertainty associated with other factors. ACCUM values for cropland, pasture, and loafing lots were assumed to be distributed by a log-triangle distribution. The mode of this distribution was calculated as the average of ACCUM calculated by BSLC for each land use and the lower and upper limits were obtained by dividing and multiplying the mode by 10, respectively (table 2). For cropland, similar distribution was defined for the month of January and the values for remaining months were calculated by multiplying it to a monthly factor. This monthly factor was obtained by observing the trend of ACCUM-PERLND

calculated by BSLC for cropland for Mossy creek. The values reported in literature and previous TMDL reports were used to define distributions for SQOLIM Factor, WSQOP and FSTDEC.

Table 2. Summary of water quality parameters for Mossy Creek Watershed model.

Parameter	Landuse	Type of Distribution
ACCUM-PERLND (cfu day ⁻¹) (Accumulation of FC on pervious land per day)	Pasture Loafing Lot Cropland	Log-triangle (1 x 10 ⁹ , 1x 10 ¹⁰ , 1 x 10 ¹¹) [†] Log-triangle (1.12 x 10 ¹¹ , 1.12 x 10 ¹² , 1.12 x 10 ¹³) Log-triangle (2 x 10 ⁶ , 2 x 10 ⁷ , 2 x 10 ⁸) Uniform (2.5, 11.5) [‡]
SQOLIM Factor (Factor which is multiplied to ACCUM values to obtain SQOLIM)		
SQOLIM-PERLND (Maximum accumulation of FC on pervious land)		ACCUM-PERLND (for each land use) x SQOLIM Factor
WSQOP-PERLND (Rate of surface runoff that will remove 90% of stored bacteria from pervious land surface)		Uniform (0.5, 2.4)
FSTDEC (day ⁻¹) (First order decay rate of bacteria)		Triangular (0.12, 1.1, 2.52) [*]

[†]Log-triangle distribution implies that the logarithm of ACCUM-PERLND follows a triangular distribution. The values in parenthesis show lower limit, mode and upper limit of the distribution. [‡]Numbers in parenthesis show lower and upper limit of the uniform distribution, respectively. ^{*}Numbers in parenthesis show lower limit, mode, and upper limit of the triangular distribution, respectively.

To simulate the fate and transport of fecal coliform deposited directly in waterbody, HSPF requires that data be input as an hourly time series. The direct deposit sources responsible for fecal coliform in Mossy Creek estimated in the TMDL study are cattle, wildlife, straight pipes directly discharging from household to streams, and one permitted point source. In Mossy Creek, the fecal coliform production by wildlife is less than 1% of the total fecal coliform production and the fecal coliform production by entire human population is less than 2% of total fecal coliform production. The point source does not discharge more than permitted amount of fecal coliform in the stream. As a result, uncertainty in direct deposit fecal coliform load was assumed to be due to cattle direct deposit only. A direct deposit cattle load distribution was determined by multiplying the cattle direct-deposit time series as calculated by the BSLC by a factor which varied as log-triangle distribution with a mode of 1 and limits of 0.1 and 10.

Generalized Likelihood Uncertainty Estimation (GLUE)

To conduct the GLUE approach, a Monte Carlo simulation of Mossy Creek watershed model for the calibration period- 1 September 1998 to 31 December 1999 was conducted. 20000 model runs were performed with different sets of input parameters chosen from their “prior” parameter distributions. A likelihood value was assigned to each input parameter set based upon a comparison of observed data simulated data. There are several ways to calculate the likelihood of an input parameter set as outlined in Beven and Binley (1992).

In the current research, the likelihood value for each simulation was calculated using the following formula,

$$L_e = (\sigma_e^2)^{-N}$$

Where, $\sigma_e^2 = \frac{1}{n} \left(\sum_{i=1}^n (O_i - Y_i)^2 \right)$, and

- L_e = likelihood value,
- σ_e^2 = variance of the residuals or mean square error
- n = number of data points
- O_i = observed data point
- Y_i = simulated data point
- N = shaping parameter, chosen by the user.

The above formula for computing the likelihood value is very useful with time series data and long-term modeling. N is a shaping factor for the response surface. A large value of N results in the differences of orders of magnitude among likelihood of similar residual variances. A smaller value of N is suggested as a starting point to make sure the model simulations bracket the observed data (Keith Beven, personal communication). A N value of 2 was used in the present research, and the resulting model simulations were just able to bracket the observed data.

Once the model runs are completed, the input parameter sets which were not an acceptable simulator of the system were rejected. There are several ways to decide the rejection criteria of input parameter sets. The input parameter sets with likelihood values not significantly different from zero are rejected or a certain percentage of parameter sets with lower likelihood values than others are rejected. In the present study, 90% of simulations with lower objective function than others were rejected. The likelihood values of remaining simulations were normalized to unity. The normalized likelihood values for each simulation were plotted against the parameter values resulting in dotty plots as shown in figure 3 for a parameter LZSN-Pasture (lower zone nominal storage). The likelihood values were used to estimate empirical posterior probability of each parameter using Bayesian equation (figure 2).

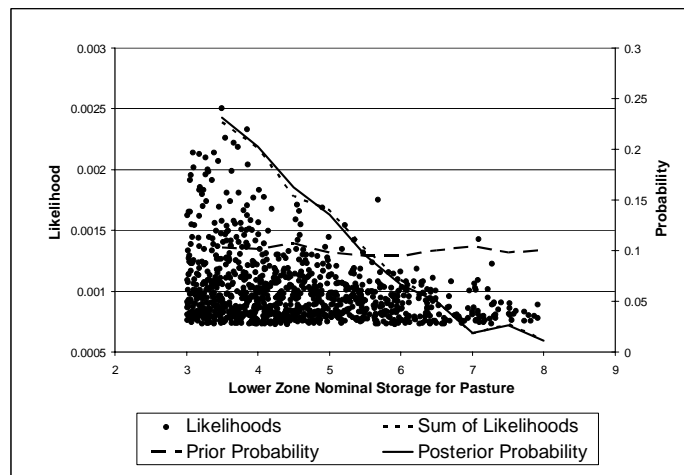


Figure 2. Likelihoods, prior and posterior probability distributions of the parameter lower zone nominal storage for pasture (LZSN-Pasture).

The posterior hydrologic parameter distributions were used to conduct Monte Carlo simulations for Mossy Creek watershed model for the validation period i.e. 1 January, 1999 till 30 September 2002. For each simulation, statistics recommended by Expert system for HSPF (HSPEXP) (Lumb 1994) were also calculated (table 3). Quantiles at 2.5 and 97.5% were computed for each statistics to validate the model. The quantiles were also computed had the prior distribution been used for Monte Carlo simulation for the validation period.

Table 3. Quantiles of the HSPEXP (Expert system for HSPF) statistics for the validation period when Monte Carlo simulations were conducted with “posterior” and “prior” distributions.

Calibration Sufficiency Statistics	Default criteria for percent error	Quantiles for validation period when “posterior” distribution was used to conduct Monte Carlo simulations		Quantiles for validation period when “prior” distributions was used to conduct Monte Carlo simulations	
		2.5	97.5	2.5	97.5
		Total Volume	±10	-10.25	8.20
50% Lowest Flows	±10	-4.69	16.84	-8.75	23.80
10% Highest Flows	±15	-16.94	1.00	-16.46	19.01
Storm Peaks	±20	-16.09	1.72	-15.43	32.01
Seasonal Volume Error	±30	0.23	11.11	0.88	16.31
Summer storm volume error	±50	-15.67	6.98	-19.13	15.28

Table 3 illustrates that using posterior distributions for water quality simulation produced improved statistics within 95% probability intervals for most of the criteria, except for 50% lowest flows and 10% highest flow. These results may imply that the likelihood function should include these different statistical measures and not only the difference between observed and simulated runoff volume for each time step. The posterior distributions of hydrologic parameters were accepted as 95% quantiles of most statistics were in acceptable range for the validation period.

To estimate posterior distributions of water quality parameters, posterior distribution of hydrological parameters and prior distribution of water quality parameters were used to conduct Monte Carlo simulations of Mossy Creek watershed model from 1 September 1998 to 30 September 2002. GLUE approach, as explained for hydrologic calibration was used to obtain posterior distribution of water quality parameters. The posterior distribution of water quality parameters was not validated against observed data as we did not have enough observed data available.

TMDL pollutant allocation scenarios

A TMDL allocation scenario allocates the pollutant loads among different sources and hence suggests the amount of reduction in pollutant loading from each source to meet the applicable water quality standard. The Mossy Creek TMDL included several allocation scenarios with two preferred allocation scenarios (table 4). The input of fecal coliform through direct deposits from cattle and wildlife in scenario 1 is less than that specified in scenario 2, whereas the input of fecal coliform from cropland is more in scenario 1 compared to scenario 2. However, it is important to note that uncertainty in wildlife direct deposit was not considered due to low production of fecal coliform by wildlife compared to other sources. The posterior distribution of hydrology and water quality parameters obtained using the GLUE approach were used to conduct Monte-Carlo simulations of Mossy Creek watershed model for the two allocation scenarios.

Table 4. TMDL pollutant allocation scenarios resulting in no violations of instantaneous criteria for E. Coli.

Scenario Number	Required Fecal Coliform load reductions in % to meet E. Coli Standards						All residential pervious land segments
	Cattle Direct Deposit	Cropland	Pasture	Loafing Lot	Wildlife Direct Deposit	Straight Pipes	
1	99	90	98	100	30	100	95
2	94	95	98	100	0	100	95

RESULTS AND DISCUSSION

The Generalized Likelihood Uncertainty Estimation (GLUE) approach was used to obtain the posterior distribution of input parameters used to model the Mossy Creek watershed. The posterior distribution of the input parameters were used to conduct Monte Carlo simulations of Mossy Creek watershed model for two different TMDL pollutant allocation scenarios as suggested in Mossy Creek TMDL. The daily average in-stream fecal coliform concentration resulting from all the simulations were used to obtain average time series and 2.5%, 10%, 90% and 97.5% quantiles for each day. These quantiles and the average time series were plotted for the two allocation scenarios. Figure 3 shows the 80 and 95% probability intervals and average time series of fecal coliform concentration resulting from one of the allocation scenarios. The number of violation incidences by each of the time series for the allocation scenarios are tabulated in table 5.

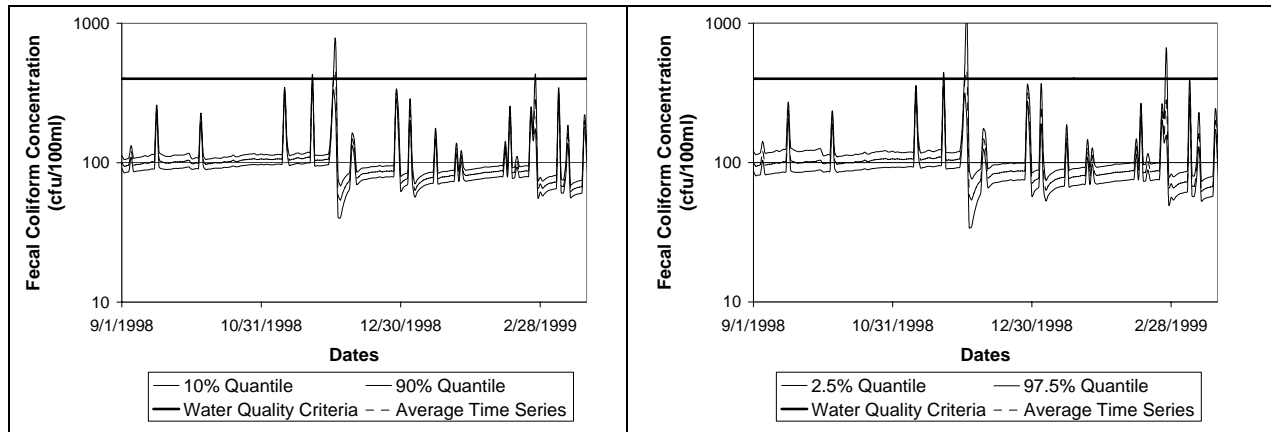


Figure 3. Average fecal coliform concentration and 80 (left) and 95% (right) probability intervals for TMDL allocation scenario 1.

Table 5. Number of violation incidences of instantaneous fecal coliform criteria by the average time series and the probability intervals for two TMDL allocation scenarios

TMDL Allocation Scenario	Number of violations by average time series	80% Probability interval	95% Probability interval
1	15	(2, 20)	(0, 31)
2	22	(7, 178)	(2, 276)

The numbers of violation incidences of instantaneous fecal coliform concentration criteria by average time series from both the allocation scenarios were not significantly different; however the numbers of violation incidences reported by outer limit of 80 and 95% probability interval from the two allocation scenarios were significantly different from each other. The number of violation incidences increased by as much as 9 times for allocation scenario 2 compared to allocation scenario 1 when we consider the 80% or 95% probability interval. The results show that the uncertainty in fecal coliform concentration and instantaneous fecal coliform criterion violation incidences increased with the increase in input of fecal coliform from cattle direct deposit.

CONCLUSIONS

The objective of this research was to quantify the uncertainty in in-stream concentration of fecal coliform in Mossy Creek, VA predicted by a water quality model developed with HSPF. The GLUE approach was used to estimate the average fecal coliform concentration and the associated uncertainty for two TMDL allocation scenarios suggested in the Mossy Creek TMDL. The numbers of violation incidences of instantaneous fecal coliform criterion resulting from average time series from the two allocation scenarios were not significantly different. However, the numbers of violation incidences reported by 80 and 95% probability intervals for the two allocation scenarios were significantly different from each other. The allocation scenario with greater input of fecal coliform by cattle direct deposit in the streams resulted in greater uncertainty than the scenario with lower input of fecal coliform by cattle direct deposit. The

numbers of violation incidences reported by the 90 and 97.5% quantiles increased by 9 times when the input of fecal coliform by cattle direct deposit was increased by 5% and the input of fecal coliform from cropland was decreased by 5%.

The type of information presented about uncertainty in water quality concentration and uncertainty in achieving a water quality criterion is extremely useful for decision makers and stakeholders. This information can be used for prioritization among the allocation scenarios, and set realistic targets for water quality achievements. Further research is aimed at estimating uncertainty in water quality using other techniques like Markov Chain Monte Carlo (Gilks et al, 1995) and parameter estimation software 'PEST' (Doherty 2001).

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ASSESSING ALTERNATIVE METHODS USED TO MODEL FECAL COLIFORM DIRECT DEPOSIT

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ABSTRACT

During dry weather, flow in low-order, upland streams is often minor and may stop completely. When developing a bacterial impairment Total Maximum Daily Load (TMDL) in an upland, rural watershed, direct deposit (DD) sources often drive the source-load reductions required to meet water quality criteria. Due to limitations in the application of existing watershed-scale water quality models, under low-flow conditions the models can predict unrealistically high in-stream bacteria concentrations. This study used Hydrological Simulation Program-FORTRAN to compare three low-flow direct deposit simulation methods; DD Stage Cut-off is a function of livestock behavior, while Flow Stagnation and Stream Reach Surface Area address DD through model representation. The study uses two Virginia watersheds where bacteria impairment TMDLs were developed with evidence of low-flow conditions. The Climate Generation (CLIGEN) program was used to stochastically generate climate inputs for multiple model simulations. Simulated daily average in-stream bacteria concentrations were compared to the Virginia single-sample water quality criteria for fecal coliform at varying loads used to represent different levels of bacteria source-load reductions. The violation rates of each reduction level – DD method were compared using a Two-Way-ANOVA. Results indicate there is a significant difference in the instantaneous violation rate between each method, at each level of DD load reduction, for both watersheds ($\alpha = 0.05$). Pairwise comparisons indicate Flow Stagnation has a strong influence on one watershed while Stream Reach Surface Area has a strong influence on the other watershed. This may be attributed to the amount of time each watershed experienced low flow conditions.



Session II-C

Connecting Nitrogen and Hydrologic Cycles

- Differential Rates of Nitrate Retention in Two Watersheds in the Fernow Experimental Forest: Norway Spruce vs. Native Hardwood Systems
- An Analysis of Solute Transport on a Harvested Hillslope in the Southern Appalachian Mountains
- Seasonal Variation of Ammonium Ion Deposition in the Shenandoah Valley
- Nutrient Loads of Poultry Farming in the Shenandoah River

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DIFFERENTIAL RATES OF NITRATE RETENTION IN TWO WATERSHEDS IN THE FERNOW EXPERIMENTAL FOREST: NORWAY SPRUCE VS. NATIVE HARDWOOD SYSTEMS

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ABSTRACT

Nitrogen saturation of several forested watersheds in the northeastern US has been documented and is largely the result of N deposition from anthropogenic inputs into the environment. Excess forest N has important implications for nutrient cycling within forests, as nitrate leaches from soils into stream environments draining these systems. Two watersheds in the Fernow Experimental Forest, WV (native hardwood and Norway spruce systems) exhibit differential patterns of nitrate retention, where the hardwood system leaks high levels to the stream and the spruce is near zero. Dissolved organic nitrogen (DON) constitutes a large pool of N in most soils, but its role in nutrient cycling is poorly understood. The goal of the present study is to investigate the role of vegetation and subsequent soil characteristics on the production and cycling of the different pools of N in forested ecosystems. Soil lysimeters were installed at several distances from the stream to investigate water held within the soil for multiple pools of C and N. Water chemistry within the stream was analyzed and soil characteristics, including C and N pools, were examined. Results indicate that the larger percentage of C present in soils influenced by spruce has a large effect on the N fractions, as indicated by the elevated fraction of organic N in the spruce watershed and the very large fraction of inorganic N in the hardwood watershed. Microbial biomass and functional group analysis indicate larger microbial populations and very low nitrifying populations present in the spruce system relative to the hardwood. This research furthers understanding of the biogeochemistry of N cycling and the factors controlling N-saturation and the resulting loss of NO_3^- into streams.

AN ANALYSIS OF SOLUTE TRANSPORT ON A HARVESTED HILLSLOPE IN THE SOUTHERN APPALACHIAN MOUNTAINS

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ABSTRACT

Transport of dissolved nitrogen and carbon to streams following forest harvesting on steep slopes of the southern Appalachian Mountains and the role of riparian buffers are poorly understood. To quantify the movement of these two solutes on a recently harvested hillslope in the southern Appalachians, four transects of lysimeters were installed across harvested areas and within a 10-m-riparian buffer. Porous ceramic- cup lysimeters were installed in each transect 1, 4, 10, 16, 30, and 50 m away from the stream in the A horizon, B horizon, and saprolite layer. Samples are being collected monthly for one year beginning March 2007. To understand patterns of dissolved nitrogen concentrations on the hillslope and riparian zone, we will determine the nitrogen mineralization potential of A horizon soils. Results will identify sources and sinks of nitrogen from the side slope to the stream across this intensively sampled hillslope.

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SEASONAL VARIATION OF AMMONIUM ION DEPOSITION IN THE SHENANDOAH VALLEY

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KEY WORDS: ammonium, deposition, seasonal, Shenandoah, poultry

ABSTRACT

For the last four years rainfall data was collected from the James Madison University Farm in Rockingham County, Virginia as part of the National Trends Network and the National Atmospheric Deposition Program. Interest in this site is strong because of the heavy poultry industry concentration in the area and its impact upon the Chesapeake Bay. The JMU Farm site shows a strong seasonality of ammonium ion concentrations, with fall and winter near the overall national average of 0.15 and 0.20 parts per million, and spring and summer concentrations spiking as high as 0.65, but averaging 0.45 ppm. This seasonal trend was also found in data from Shenandoah National Park and Charlottesville, though the actual concentrations were significantly lower. Strong seasonality was also apparent in data from Clinton, North Carolina and Lancaster, Pennsylvania, but in these cases the overall concentrations were higher probably due to the presence of hog farming operations in addition to the poultry farms. The data indicates a strong impact of animal operations on ammonium ion concentrations and total deposition.

INTRODUCTION

Understanding the nature of nutrient cycles and human impact on these cycles is essential to solving problems of nutrient loading in water bodies such as the Chesapeake Bay. As part of their effort to reduce total nitrogen entering the Chesapeake Bay, the Chesapeake Bay Program (CBP) of the Environmental Protection Agency (EPA) has funded a number of studies on nitrogen sources throughout the watershed. James Madison University (JMU) is participating in this EPA funded effort by managing and monitoring a National Trends Network (NTN) site at the JMU Farm under the auspices of the National Atmospheric Deposition Program (NADP). The CBP has strong interest in the Shenandoah Valley because of the high numbers of poultry farms in the area, centered in Rockingham County.

The NADP has over 250 sites in the NTN system, each of which collects rainfall on a weekly basis according to a rigorous protocol. The sites use a lidded collection system that opens at the start of a rainfall and closes when the rain stops. This limits airborne contamination of samples. The sites also have rain gauges that continuously chart rainfall timing and volume each week. Collected samples are then sent to the Central Analytical Laboratory on the Illinois State University campus for analysis of nine key ions including sulfate (SO_4^{2-}), nitrate (NO_3^-) and

ammonium (NH_4^+). This last ion is the central focus of this study. This weekly data is then made available through their website after determining if sampling protocols were met.

Ammonium ions are a primary byproduct of the poultry production system dominant in the US today. They are a breakdown product of poultry manure, from both chicken and turkey operations, when there is inadequate organic carbon available in the poultry house, a common state of affairs in the vast majority of poultry houses. A standard broiler chicken house is around 450 feet long, 40 feet wide, single story, with a sloped, center-peaked roof about 15 feet high in the middle. They are entirely metal buildings and well-insulated to keep the young birds at the ideal temperature. Birds are introduced into the houses as day-old chicks, eat, drink and grow for 39-44 days depending upon the customer's preference, then hauled to a processing facility. Five major companies dominate poultry operations in the Shenandoah Valley: Tyson, Cargill, Pilgrim's Pride, Purdue and George's. The companies own the birds and control the operations, but private farmers own the chicken houses (or turkey houses, which are slightly different in size, ventilation system, and bird residence time, but otherwise similar) and bear responsibility for handling all waste produced during the growing cycle.

It is the waste that produces the ammonium. Early in the growing cycle the houses are generally odor free, but as the birds grow the ability of the bedding to absorb the nitrogenous waste is overwhelmed by the volume produced. A typical poultry house containing 25,000 birds, feeds the birds approximately 250,000 pounds of feed. About 110,000 lbs of this is converted into whole poultry that are shipped to processing facilities. The remainder is used for respiration or lies on the floor of the poultry house in the form of excrement mixed with approximately 8 tons of bedding material, commonly wood chips, peanut hulls, or other high carbon organic material. This litter mix form a cake or a crumble structure that can be scooped with a shovel. Analysis of the litter mix shows that it has a total nitrogen content between 3 and 4 percent by weight, and a Carbon:Nitrogen (C:N) ratio of 9:1 (Sistani 2003). The ammonium content of the litter is generally between 11.5 and 12.5% of the total N content, which in turn is approximately 4% of the weight of the litter (Sistani 2003)

Whenever a high nutrient content source has a C:N ratio of less than 25:1 that source has the potential to produce volatile ammonia off-gassing. Any visitor to a poultry house, especially toward the end of a poultry growth cycle, is aware of the strong ammonia smell coming off the litter. This off-gassed ammonia converts to ammonium ions when it comes in contact with water molecules in the air, which prompted JMU's effort at monitoring in Rockingham County, Virginia.

Seasonal variations

Ammonium ion release is not uniform in the Shenandoah Valley or elsewhere in the country. The sources of emissions are varied, but commonly are localized and related to agriculture or confined animal feeding operations. The primary indicator of emissions from these sources, including animal waste, is temperature, which increases chemical and biological activity leading to loss of ammonia as a gas in situations where the C:N ratio is low (Lehmann 2006). Because ammonia and ammonium ions are highly reactive they do not tend to travel far from their point of origin. They combine in the air with sulfate and nitrate ions, which are common pollutants

from coal burning and internal combustion engines often found as acids associated with water (Lehmann 2006). When humidity is sufficiently high these may settle in dry deposition particulates, or in rainfall (Scudlark 2005).

Temperature alone does not explain the entire range of seasonal variation. Management activities also have an impact. According to state guidelines, farmers are not allowed to spread poultry litter or other animal manures during winter months when the ground is frequently frozen. This reduces the possibility of surface runoff or leaching losses in times when plants are not actively growing. Spreading usually begins in March in the Shenandoah Valley, and is at its peak in the April and May period. Spreading activity diminishes over the summer, though some may continue on pasture land, and the levels ammonium ion during this season are best explained by temperature.

Farmers spend most of the fall season harvesting crops and do not spread poultry litter or other animal manures unless needed for pasture land. Most litter taken from poultry houses during fall or winter is stored in covered litter storage buildings to prevent nutrient losses during this less active growing period.

Data analysis

The data analyzed in this study is available on-line from the National Atmospheric Deposition Program's website. It is derived from their chemical analysis of the weekly samples collected from their 250 program sites (NADP 2007). Samples from the JMU Farm location, about 11 miles SSW of Harrisonburg, VA are included in the database. Data from December 2002-November 2006 was downloaded and analyzed on an Excel spreadsheet (Excel? Quattro-Pro?). Weekly and monthly data did not exhibit distinct patterns, but quarterly data proved more fruitful. The quarterly data was derived from monthly weighted means over the four year period.

Figure 1 shows these quarterly ammonium ion concentration means given in mg NH_4^+ per liter of solution. As can be seen in the figure, fall (September - November) and winter (December - February) are distinctly lower than spring (March - May) and summer (June - August) values, with spring consistently having the highest values. These differences are due to increasing temperature, which increases the volatility of ammonia in both soil and animal waste (Lehmann 2006), and increasing human management activities in active farming areas (Scudlark 2005).

Concentration is not the only factor affecting ammonium deposition. The total amount of ammonium deposition is also a function of rainfall. Quarterly JMU Farm records are provided in Figure 2. As seen, 2003 was an exceptionally high rainfall year, which included two large September events related to tropical storms. Concentration of ammonium ions in these storms is quite low, 0.13 mg/l, but the volume is high, though not high enough to boost overall deposition during the period. As a result, the variation in deposition, Figure 3, is seasonally distinct, much like that of concentration. However, in all years the winter quarter is consistently lowest in deposition, while the fall is typically lowest in concentration.

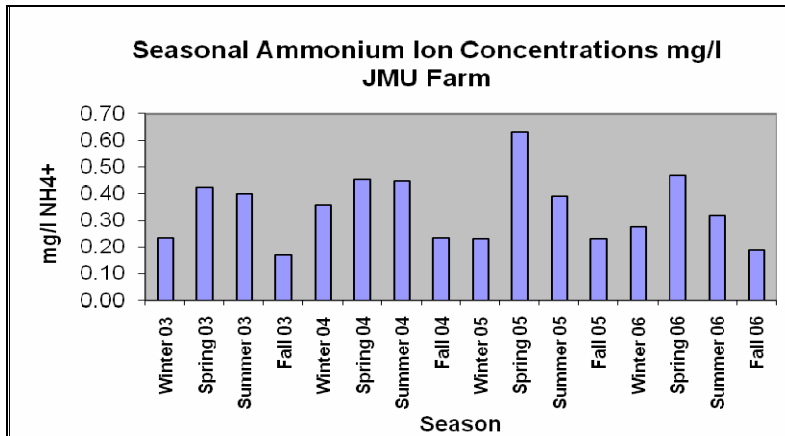


Figure 1: Seasonal ammonium ion concentrations at the JMU farm (VA27).

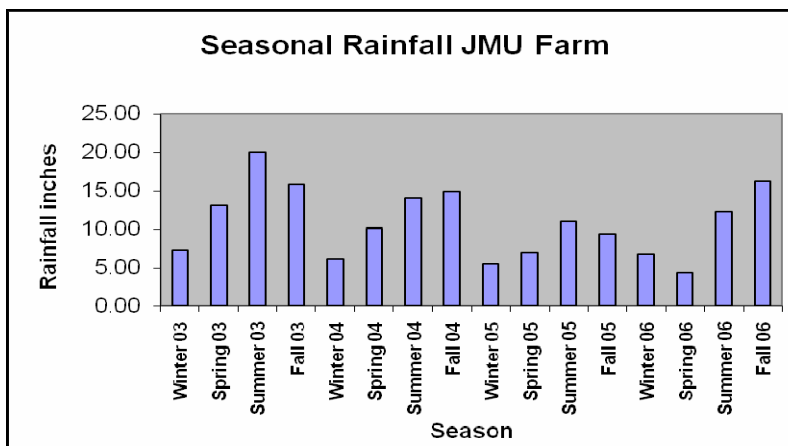


Figure 2: Seasonal rainfall totals in inches at the JMU farm.

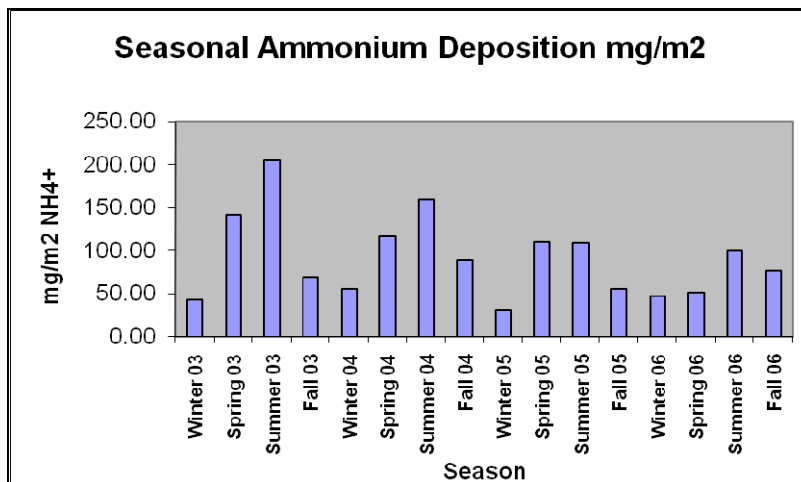


Figure 3: Total seasonal ammonium ion deposition in rainfall at the JMU farm.

Is the seasonality of the JMU Farm data a local phenomenon, or does it reflect a broader, regional pattern? Are there other regional sampling locations that show the same seasonal variation pattern as that found in the Rockingham County area of Virginia? In order to answer these two questions additional data was obtained from the NADP website. Two nearby sites, the first located in Big Meadows in Shenandoah National Park, part of the Blue Ridge Mountains, and the second near Charlottesville, VA were selected for comparison. The same data parameters were downloaded from these sites and quarterly weighted means were calculated. At the same time two sites with known heavy agricultural activity were downloaded. The first of these was Lancaster County, PA, the second most productive agricultural county in the eastern US in dollar terms, with significant poultry, hog and dairy farms (USDA 1997). The second is located in Clinton in Sampson County, North Carolina, the site of a major increase in hog production over the last 20 years (Walker 2000a). According to the 1997 Census of Agriculture Lancaster County ranked 5th in the nation in total value of livestock and poultry products sold, while Sampson County was 8th, and Rockingham County 16th (USDA 1997).

Quarterly variation in ammonium ion concentrations are seen in both the Big Meadows (VA28) and Charlottesville (VA00) sites, but the concentrations are consistently lower than those found at the JMU Farm, as shown in Figures 4 and 5. Both of these sites have higher annual average rainfall than the JMU Farm and deposition data shows less variation than concentration data because of this. Both Lancaster, PA (PA47) and Clinton, NC (NC35) have significant seasonal variation in their ammonium ion levels and these values are higher than at the JMU Farm as seen in Figures 6 and 7. This is best explained by the presence of confined hog production operations in these two areas; the liquid manure being even more prone to the off-gassing of ammonia than poultry litter (Walker 2000b).

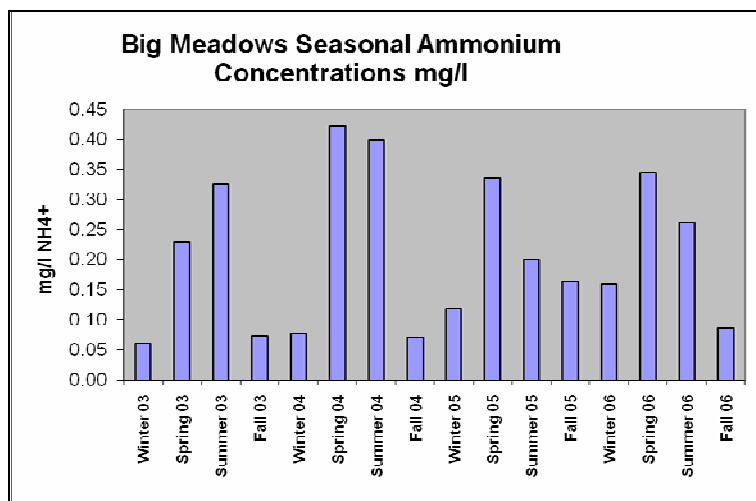


Figure 4: Seasonal ammonium ion concentrations at Big Meadows (VA28), Shenandoah National Park. Note that the scale is differs from that for the JMU farm in Figure 1.

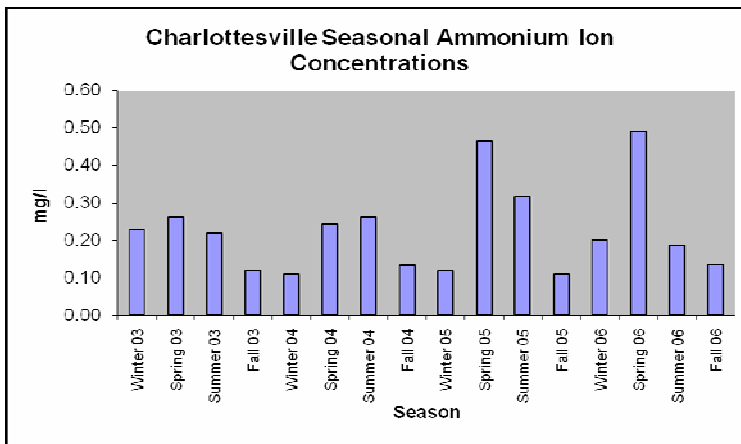


Figure 5: Seasonal ammonium ion concentrations near Charlottesville, VA (VA00).

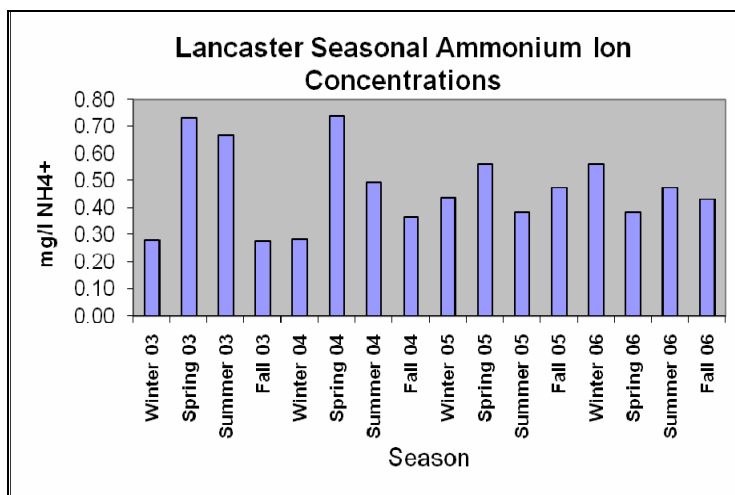


Figure 6: Seasonal ammonium ion concentrations in Lancaster, PA (PA47).

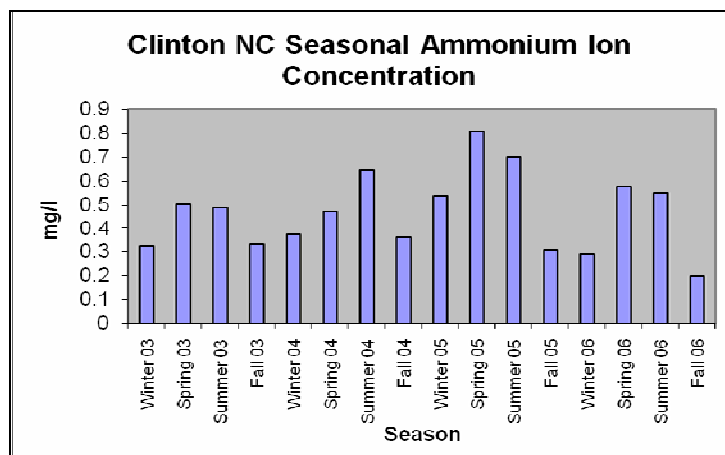


Figure 7: Seasonal ammonium ion concentrations in Clinton, Sampson Co., NC (NC35).

COMPARISON AND CONCLUSION

A comparison of all five sites analyzed is shown in Figure 8. The figure shows clearly that counties with heavy agricultural activities involving confined animal feeding operations also show increased ammonium ion concentrations compared to other sites. Ammonia and ammonium ions are highly reactive chemicals that combine readily with water and other airborne contaminants such as sulfates or nitrates and settle out of the atmosphere locally rather than regionally (Lehmann 2006). Sampson County, NC with its high concentration of hog operations, and Lancaster County with its more diverse hog, dairy and poultry operations have greater ammonium ion deposition problems than Rockingham County. Rockingham does show an increase in its own ammonium ion concentration over nearby sites. A thorough statistical analysis of the data is needed for the next phase of this work to confirm the implications of this study so far.

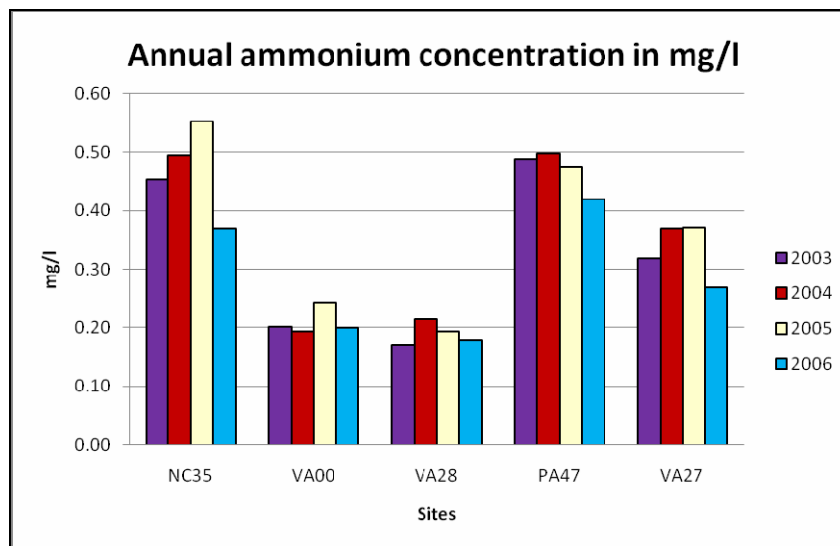


Figure 8: Annual weighted means of ammonium ion concentrations from five NADP sampling locations. In the order above these are: Sampson County, NC; Charlottesville, VA; Big Meadows, Shenandoah National Park; Lancaster County, PA; Rockingham County, VA. (NADP data access).

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NUTRIENT LOADS OF POULTRY FARMING IN THE SHENANDOAH RIVER

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KEY WORDS: nutrient, nitrate nitrogen, poultry litter, water quality

ABSTRACT

This study focuses on the role of agricultural practices, specifically poultry operations, and how they impact water quality. Rockingham County and parts of surrounding counties contain a high density of poultry operations, the highest within the Chesapeake Bay watershed. To study the relationship between the poultry operations and water quality, specifically nitrate nitrogen content, within the study area a water quality database and a geographic information system database was created. The water quality database, derived from data collected by Friends of the Shenandoah River, provided the information necessary for analyzing the statistical correlation of the relationship between poultry operations and nitrate nitrogen content within the watersheds studied. The geographic information system database provided the means necessary to locate poultry operations within the study area and determine watershed area and poultry house density. The time period studied is from January 2003 to May 2005. The results show that nitrate nitrogen content in a watershed is linked to poultry house density and is statistically significant.

INTRODUCTION

In order to study the sources of the nutrient loads to the Chesapeake Bay, this report will prove with limitations that poultry farming within the Shenandoah Valley contributes significantly to the nutrient loading to the Bay, which is under a plan to reduce nutrient loading by 40% (CBP, 2006). The Friends of the Shenandoah River (FOSR) is a strong source of information made up by a system of volunteers that collect water samples from 180 valley locations biweekly. All samples are sent the same day to a lab at Shenandoah University in Winchester, VA and are analyzed for a number of chemical variables including ammonium, nitrate nitrogen and phosphorus. The data is publicly available through the Water Window, which is maintained by the Pure Water Forum (www.purewaterforum.org/waterwindow) and James Madison University. This study will focus primarily on the nitrate-nitrogen data from the FOSR Water Window.

The nitrate-nitrogen data has little use without some understanding of the spatial nature of the smaller watersheds tested by the FOSR and the concentration of poultry houses in each of these watersheds. Geographic Information Systems (GIS) databases are available that provide

watershed delineation, watershed areas, and land use information within each watershed. This study will add a layer of data to the GIS database to detail the locations and density of poultry houses within each watershed. Geographically, Rockingham County sits at the upper end of the Shenandoah River Watershed and is mostly rural and heavily agricultural. It also ranks highly both statewide and nationally in poultry production (Census of Agriculture, 2002). This makes Rockingham County a prime study area for the relationship of nitrate nitrogen and poultry operations. Many of the small streams in the county have their entire watershed within county boundaries. Four of the twelve streams studied have major portions of their watersheds outside Rockingham County, specifically Augusta and Shenandoah County.

METHODS

A database was created including each of the twelve watersheds studied and their respective nitrate nitrogen water quality content, stream flow and poultry house density. The twelve watersheds were selected based on available water quality data and monitoring site location. Water quality data from FOSR, specifically nitrate nitrogen content had to be available between January 2003 and May 2005. Appropriate monitoring site locations are described as a monitoring site where water quality samples only measure the water draining from the watershed of study, typically before any confluence with another stream or river. Also the monitoring site should be located at the bottom of the watershed in order to capture the water quality of the entire watershed.

The USGS stream flow is measured by a continuous stream flow monitoring gauge. There are three stream flow monitoring locations used in this study; the USGS monitoring site at Burkettown, Virginia, which is used for all watersheds except two. Muddy Creek uses the monitoring station at Mount Clinton and Smith Creek uses the New Market station. This study uses a fractional stream flow rate based upon the watershed area in order to compare to the nitrate nitrogen data at each testing site. The fractional flow was calculated by multiplying the Burkettown flow by each watershed area divided by the Burkettown flow area of 379 square miles as reported by the USGS.

Poultry House density is the last component of the database. This database was created from existing county and city GIS data. County data was used to spatially locate and identify poultry houses, calculate each watershed area and determine poultry house density. Poultry houses were identified in Rockingham and Augusta County with their building use code. Shenandoah County poultry houses were identified from reported building sizes and aerial imagery. A typical poultry house is narrow and more than 400ft in length, and two are commonly seen side by side. The water quality and GIS database are the foundation for analysis on the correlation of poultry house density to nitrate nitrogen and was statistically measured in the software JMP IN.

The Nitrate Nitrogen data used in this study was collected by the FOSR using a Lachat QuikChem 8000 Flow Injection Analyzer and QuikChem Method 10-107-04-1-C / STD MTDS 4500 N03- to determine the nitrate nitrogen content in a sample. The FOSR states that none of the data collected under their organization may be used in a manner to create regulatory actions and is collected specifically for study by interested agencies and organizations.

Mossy Creek and Pleasant Run were excluded from some statistical analyses because they contain some anomalies. Mossy Creek is a gaining stream, meaning that the natural water flow amount in the stream is enhanced by springs where water flows into Mossy Creek from the Upper North River watershed area. This added water flow through the watershed dilutes the nitrate nitrogen content in the data producing an artificially low nitrate nitrogen value in Mossy Creek itself even though it has a relatively high density of poultry houses. Pleasant Run was excluded because there is a point source dairy farm operation just upstream of Friends of the Shenandoah River the sampling site (DEQ, 2001). The high nitrate nitrogen content shown in Pleasant Run comes from the dairy operation as the poultry house density is relatively low. These two watersheds are outliers in the data set and are excluded from some analysis in this report. All other watersheds in this study do not have any characteristics that would warrant exclusion from analysis.

RESULTS

The first part of this study characterizes each watershed in terms of total size, poultry house density, land use and TMDL impairments. The largest watershed in this study is Smith Creek at 105.11 square miles, as shown in Table 1. A larger amount of water flowing in a larger watershed has a diluting capability in terms of pollutant loading, when comparing two watersheds with different areas and the same total pollutant inputs. Therefore nutrient loading is based on the runoff volume and the pollutant concentrations in the runoff water (EPA STEPL, 2005).

Table 1: Watershed characteristics.

Watershed	Watershed Area (mi ²)	Watershed Area (acres)	Poultry House Count	Poultry House Density	Land Use Category	TMDL Impairment						
						Benthics	Fecal Coliform	E. Coli	pH	Temp	N	P
COOKS CREEK	24.87	15,916.62	138	5.55	AG	✓	✓	✓				
MUDDY CREEK	31.32	20,044.91	133	4.25	AG	✓	✓	✓			✓	
MOSSY CREEK	15.75	10,076.72	52	3.3	AG	✓	✓					
LONG GLADE CREEK	18.55	11,870.48	51	2.75	AG		✓					
CUB RUN	26.81	17,160.05	52	1.94	FR		✓			✓		
NAKED CREEK	22.93	14,673.69	37	1.61	AG		✓					
DRY RIVER (combined)	88.52	56,650.81	127	1.43	FR				✓			
SMITH CREEK	105.11	67,270.10	142	1.35	FR	✓	✓					
MILL CREEK	15.05	9,635.04	19	1.26	AG	✓	✓	✓		✓		
BRIERY BRANCH	49.49	31,673.77	54	1.09	FR			✓	✓			
PLEASANT RUN	8.3	5,308.84	9	1.08	AG	✓	✓					✓
BLACKS RUN	19.15	12,252.66	5	0.26	UR	✓	✓	✓				
LOWER DRY RIVER	15.72	10,057.62	-	-	FR				✓			
UPPER DRY RIVER	72.8	46,593.19	-	-	FR				✓			

Table 1 is ranked by poultry house density and shows that Cooks Creek and Muddy Creek have the highest poultry house densities both have elevated levels of nitrate nitrogen content. The twelve watersheds studied are placed in three different major land use categories, urban (UR), forest (FR) and agricultural (AG). Seven out of the twelve streams in this study are classified as agricultural. The agricultural nature of these streams impacts their water quality as shown by the Total Maximum Daily Load (TMDL) studies that are available from the Virginia Department of Environmental Quality (DEQ). Most impairments are attributed to non-point source runoff, primarily agricultural. These TMDL pollutants impact the streams for aquatic life use and recreation use. Notice that Muddy Creek is the only stream identified to have an overabundance of nitrogen and is also the number two ranking watershed in terms of poultry house density. Figure 1 is a time profile of each watershed nitrate nitrogen values where blank or missing data is displayed as the previous recorded value. This graph shows the overall comparison of the nitrate nitrogen in the twelve watersheds. Cooks Creek has the overall highest nitrate nitrogen values (most above 5 ppm) followed by Muddy Creek and Pleasant Run, a noted anomaly. Any recorded nitrate nitrogen value that is above 10 ppm exceeds the water quality standards set by the Federal and State governments. Cooks Creek does not fit the standard for nitrate nitrogen from January to July of 2003 and again in September and November. Muddy Creek, Pleasant Run and Long Glade Creek all show unacceptable values in January 2003. This value is high across all twelve watersheds and the cause is unknown.

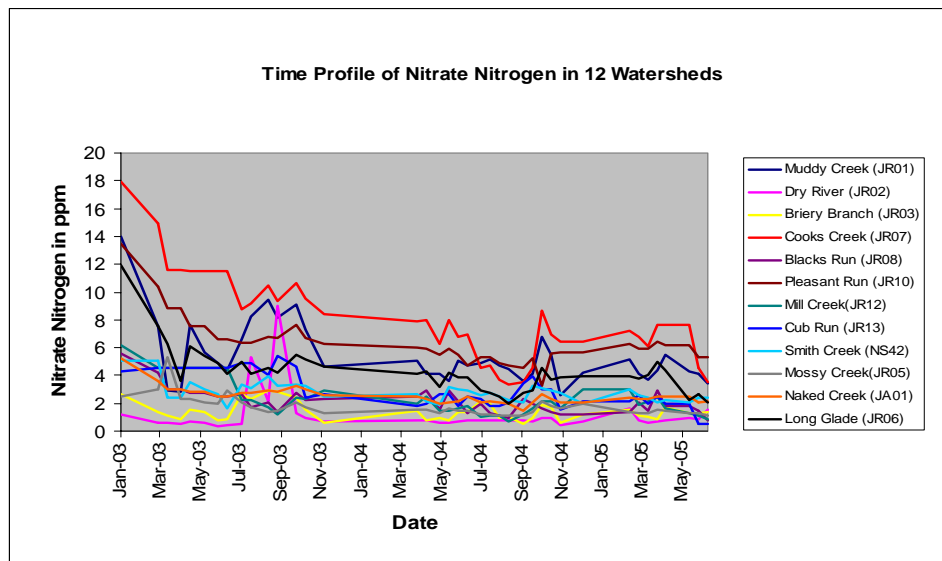


Figure 1: Watershed nitrate nitrogen nontent

Watershed areas with high poultry house density correlate to average high nitrate nitrogen content. The r-squared value of the correlation between the two variables is low ($r^2 = 0.393$), however the correlation is statistically significant ($p = 0.0292$). The graphs suggest an overall positive relationship between poultry house density and nitrogen levels. This relationship is somewhat obscured by the outliers Pleasant Run and Mossy Creek as mentioned earlier and when excluded from analysis there is greater statistical significance ($r^2 = 0.841$, $p = 0.0095$).

The second part of this study attempts to describe the seasonality of the nitrate nitrogen within a watershed. For the seasonal analysis, both outliers are excluded. All seasons have a high degree of statistical significance ($p = \text{less than } 0.0001$). The winter season, from December 1 to February 28, is legally restricted from poultry litter spreading and the data appears to be highly correlated. This correlation is due to the small amount of winter season samples (3) and the unusually high nitrate nitrogen values recorded in January 2003. This is an unrealistic result and limits comparison to other seasons. The other seasons have a lower correlation between the two variables, but are better descriptions of the true relationship between nitrate nitrogen content and poultry house density. The fall season (Sept-Nov) has the highest correlation of the data ($r\text{-squared} = 0.9381$) followed by the summer (0.93) and then the spring (0.7846). Fall's high correlation can be attributed to the high stream flow rates due to tropical storm events, the highest rates seen in the study. Hurricane Isabel occurred in September 2003, Hurricane Frances was in September of 2004 followed by Hurricane Ivan in October 2004. In agricultural areas, this amount of runoff can lead to heavy nutrient loads in streams if cropland or pastures were recently fertilized with poultry litter.

The nitrate nitrogen content measured in parts per million within a stream depends upon the stream flow (EPA STEPL, 2005). The third part of this study aims to describe this relationship. Each watershed's nitrate nitrogen content and the flow measured at Burketown are shown in Figure 2. It is identified that Nitrate nitrogen content measured at one date is representative of environmental conditions at two weeks previous to the testing date. The correlation between the measured nitrogen level and the daily 75th quartile stream flow from two weeks earlier is statistically significant when comparing all watersheds ($R \text{ squared} = 0.693485$, $p = 0.0001$).

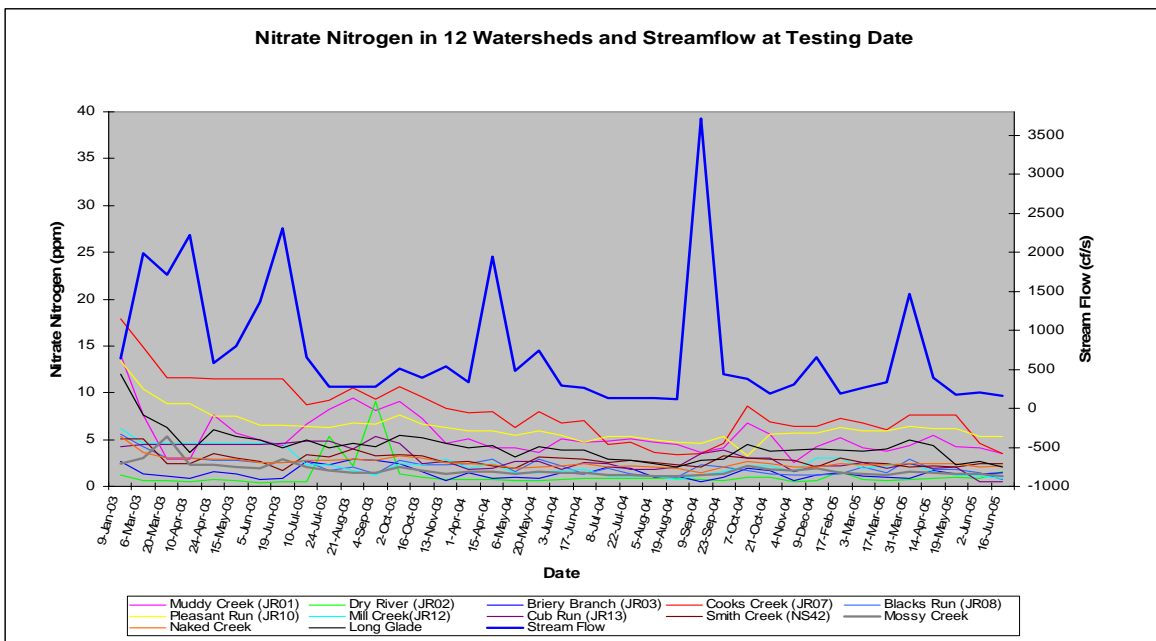


Figure 2: Watershed nitrate nitrogen content and stream flow

The analysis uses the 75th quartile flow in order to give a better description of the overall flow during each two-week period relating to the nitrate nitrogen values. A high one-time rainfall

event, namely the hurricanes, will inflate the mean value, thus the mean value is not used in this analysis. This time lag response shows that nitrate nitrogen content is a combination of surface runoff as well as subsurface or vadose zone water flow (Harter *et al*, 2005). Poultry litter that is applied to crop or pasture land is absorbed into the soil and percolates through the ground in the vadose zone. The vadose zone in the soil horizon is that part of the soil that is not continuously saturated with water, also called the unsaturated zone (Sellers, 1998). Vadose flow takes more time to reach a near body of water than surface runoff and this fact concurs with the results of this study. It also has the capacity to dilute pollutant concentrations from contaminants binding to other soil components or transforming into other compounds (Sellers, 1998). An analysis of each individual watershed's nitrate nitrogen content and two-week lagged stream flow values shows the intensity of the relationship. The watershed with the greatest intensity is Muddy Creek, followed by Pleasant Run, Cooks Creek and Long Glade Creek. Interestingly, Muddy Creek, Cooks Creek and Long Glade Creek all rank among the highest for poultry house density, the exception being data outlier Pleasant Run.

DISCUSSION

Poultry house density is statistically correlated to nitrate nitrogen content in the twelve watersheds studied in this report. The results show that watersheds with a low poultry house density also have low nitrate nitrogen content, this suggests that the high nitrate nitrogen content found in other watersheds is due to the poultry operations and not another factor. Specifically, Cooks Creek has the highest nitrate nitrogen content over the study period and also has the highest poultry house density within the twelve studied watersheds. Muddy Creek has the second highest poultry house density and also has a high level of nitrate nitrogen content. When excluding the two noted anomalies of Pleasant Run and Mossy Creek, the statistical significance of the correlation between the nitrate nitrogen content and poultry house density improves ($p = 0.0292$ to 0.0095) respectively. Therefore this study has high statistical significance. High poultry house density does in fact relate to high levels of nitrate nitrogen content within a watershed.

The seasonal analysis of the nitrate nitrogen content within a watershed also shows a high correlation with poultry house density. It was hypothesized that there should be higher levels of nitrate nitrogen content during approved spreading times as well as during seasons where the spreading of the poultry litter as a natural fertilizer will most benefit to the cropland. The degree of correlation between nitrate nitrogen content and poultry house density increases from the spring season to the fall. The fall season has the highest correlation with an r squared of 0.94 ($p < 0.0001$), meaning that the nitrate nitrogen average during this season for each watershed is represented in a linear fashion to the poultry house density for each watershed. The fall season is the time of the year when crops cease nitrogen uptake and surplus nitrate is free to move. Also, poultry litter spreading is common in the fall, especially on pastureland, as farmers attempt to lower stocks of litter in their storage facilities in advance of the non-spreading winter season. A fall application of poultry litter enables the litter to decompose over a greater time period and provides a greater nutrient release for following crops (Dick, 2006). Dick also describes that the fall season also has the greatest potential for nutrient leaching and denitrification, a nitrogen loss. This report shows that the seasons do play a role in the amount of nitrate nitrogen loading within

a watershed, however the results are inconclusive during the winter season due to the small sample size.

The results of this study identify priority watershed areas. A priority watershed signifies a strong statistically significant relationship between poultry house density and nitrate nitrogen content. This report shows the levels of nitrate nitrogen content in Cooks Creek exceeds the maximum allowable limits in terms of water quality and this is linked to poultry operations. Other priority areas identified by this study are Muddy Creek, Mossy Creek and Long Glade Creek. These four watershed areas all have the highest poultry house density and three have the highest nitrate nitrogen content. In terms of the results of this study, these priority areas signify that rehabilitation efforts or conservation programs would provide the most benefit if implemented in these areas to control nutrient loading. Legally, Muddy Creek is currently under a TMDL schedule to reduce nutrient loads and as found in this report Cooks Creek exceeds these TMDL standards as well. There are many water quality restoration efforts within Rockingham County, but few effectively control the nitrate nitrogen loading from poultry litter applications. As nitrogen is a controlled nutrient under the Chesapeake Bay Program 2000 and Rockingham County ranks highest in poultry production, nitrate nitrogen content within the area should be more strictly monitored and controlled in order to reduce the nutrient loading into the Bay.

This study addresses the issue of impaired water quality due to nutrient loading and shows that nitrate nitrogen content within watersheds is linked to the poultry house density. In order to protect environmental and human health future studies on this issue should be continued. This study does not factor in the nutrient content contributed by other farming practices, namely cattle ranching and chemical fertilizers. However, current Federal and State policies regarding poultry litter spreading is limited. The results of this study show that poultry operations have a direct impact on the water quality of the surrounding area. Further study on this impact should be completed and policy regarding this issue may need to be adjusted in compliance with Federal and State goals to improve water quality throughout Rockingham County and other agricultural areas of interest.

CONCLUSION

The twelve watershed areas within the Counties studied in this report show that poultry operations have a significant impact on the water quality. Specifically, poultry operations contribute nitrate nitrogen as a nutrient pollutant. Two watershed areas within this study area, Cooks Creek and Muddy Creek, show the highest correlation between poultry house density and impaired water quality due to nitrate nitrogen loading. Of these two, only Muddy Creek is currently State regulated for nitrate nitrogen content.

Water quality monitoring testing frequency is a major limiting factor of the conclusions in this report. Bimonthly water quality testing is appropriate for long term studies on the fluctuation of various water quality parameters but testing errors reduced the sample size making a bimonthly schedule inadequate for accurately determining poultry's contribution to the nitrate load in each stream. Increased testing frequency would provide more data for smaller spatial studies as well as increased accuracy for long term studies.

Poultry operation impacts on water quality are well documented (Edwards and Daniel, 1993, 1994; Sauer *et al.*, 1999; Sauer *et al.*, 2000; Spears, 2003; Weaver, 1998). Future studies of the impacts of poultry operations should include selecting an appropriate watershed, like a priority watershed identified in this report, and monitoring water quality daily to study the relationship in a more controlled fashion. This type of future study may help to lead to the adjustment of current water quality policies and regulation of poultry operations by providing specific and detailed information. Changing poultry regulations requires a carefully planned study that can point to specific practices requiring change, such as the timing and quantity of poultry litter spreading on different landscapes. We can now say with some confidence that there is a statistically significant relationship between poultry house density and nitrate nitrogen content within a watershed.

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Session III-A

Detecting Hydrophobic Pollutants in the Aquatic Environment

- Rapid and Sensitive Method for the Analysis of the Bioavailable Portion of Hydrophobic Pollutants
- PCB Monitoring in Virginia's Impaired Waterbodies Using an Ultra Sensitive Analytical Method
- PCB Bioaccumulation Factors in Potomac Estuary Fish
- Sampling for Emerging Contaminants in the South Branch of the Potomac River and Other West Virginia Streams 2004-2006

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RAPID AND SENSITIVE METHOD FOR THE ANALYSIS OF THE BIOAVAILABLE PORTION OF HYDROPHOBIC POLLUTANTS

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ABSTRACT

The combination of a semipermeable membrane device (SPMD) with fiber optics results in a remote sensing device that is capable of detecting low levels of hydrophobic pollutants in water. Fluorescence measurements are used for the direct observation of hydrophobic so-lute uptake into the SPMD from the surrounding medium. Since SPMD uptake mimics the passive accumulation of nonpolar pollutants by aquatic organisms, the sensor is a simple device that allows the detection of bioavailable fluorescent pollutants, such as the polyaromatic hydrocarbons (PAHs). Compared to conventional SPMDs, the sensor offers more rapid analysis and easy calibration to determine ambient pollution concentration. We will present the construction design of the SPMD sensors, their analytical characteristics for the analysis of representative PAHs, and investigate the mechanism of SPMD uptake and depuration. The sensor shows promise for remote sensing applications: one hour exposure times result in detection limits that are improved by two to three orders of magnitude compared to bare fiber optic sensors. Longer expo-sure times would result in still greater sensitivity. Moreover, the ability of the device to provide direct observations of uptake and depuration in real time is proven useful for investigations into the fundamental theory of SPMD technology.

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PCB MONITORING IN VIRGINIA'S PCB IMPAIRED WATERBODIES USING AN ULTRA-SENSITIVE ANALYTICAL METHOD

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KEY WORDS: fish impairment, PCB, congener, method 1668a, TMDL

ABSTRACT

Polychlorinated Biphenyl (PCB) contamination of fish has resulted in the Virginia Department of Health posting fish consumption advisories in numerous Virginia water bodies including rivers, lakes and estuaries. PCBs, which have been banned in the late 1970's, are very persistent in the environment and are generally considered a legacy pollutant tied up in sediments. In addition to the analysis of tissues, past studies focused exclusively on sediments where PCBs concentrate and were easy to quantitate using standard methods. New information generated through Total Maximum Daily Load (TMDL) source identification studies using a high resolution/low detection PCB analytical method on water samples suggests there are on-going sources of these chlorinated contaminants to Virginia's PCB impaired water bodies. The present studies focus on the Potomac and Roanoke Rivers. These findings should provide an accounting of major PCB inputs to the impaired waters with follow-up management actions that result in loading reductions.

INTRODUCTION

Polychlorinated biphenyls (PCBs) are a group of 209 discrete man-made chemicals that were initially synthesized in the late 1920's. Individual compounds are known as PCB congeners with a chemical make-up of a biphenyl ring and one to ten chlorine atoms, with the location and number of chlorine atoms determining the specific congener. PCBs can be further categorized into ten homolog groups based on the number of chlorine atoms. Congeners with greater levels of chlorination are more hydrophobic and readily bind to solids with a preference for the carbon based component (Erickson 1997). Chlorine content also affects resistance to environmental degradation and bioaccumulation. With their chemical make-up, PCBs were considered an industrial breakthrough with their greatest use in transformers and capacitors in the form of dielectric fluids (Erickson 1997). PCBs were manufactured in the United States under the brand name of Aroclor and typically contained mixtures of 110-120 congeners (Frame et al., 1996). Manufacturing of PCBs was banned in the late 1970's with their phase-out occurring over time. PCBs have been classified as a probable carcinogen.

PCB contamination of fish is widespread in Virginia's waters. The Virginia Department of Environmental Quality (VADEQ) annually monitors fish/shellfish tissue and sediment for the presence of contaminants including persistent toxins. Tissue and sediment samples are analyzed on a congener specific basis with all detectable analytes summed to yield total PCBs (tPCB).

Since toxicological information is not available for all PCB compounds, the Environmental Protection Agency (EPA) regulates this group of chlorinated compounds as tPCB which is defined by EPA as the sum of all congeners, isomers, homologs, or Aroclors. VADEQ assesses the tissue contaminant data at the water quality criterion (WQC) concentration of 54 ng/g (dry weight basis of tPCB) within the scope of the 305(b)/303(d) process. The Virginia Department of Health (VDH) evaluates the same data at a similar threshold level to evaluate human health risks and the need for fish consumption advisories or restrictions. In either case, when the regulatory threshold is exceeded and the waterbody is listed as impaired, Federal and State Regulations require development of a Total Maximum Daily Load (TMDL). A total of 973 river miles, 72,000 lake/reservoir acres, and 2,110 square miles of estuary have been listed as impaired in Virginia (VADEQ 2006).

Even though compiled data from the VADEQ fish monitoring program appear to be exhibiting a decline in fish tissue concentrations as the use of PCBs has diminished, the listing of PCB fish impairments has continued. For example, PCB concentrations in fish tissue data collected on the tidal Potomac estuary have decreased about 2x since the early 1990's while PCB concentrations in the Roanoke/Staunton River have decreased 2-3x in the same time-frame, but yet both waterbodies remain listed for PCBs. As such, a need exists to determine if the PCB concentrations observed in present day fish are a result of historical or on-going releases.

To fulfill this need, an initial step of the TMDL development process is to perform a source identification study. First, all available data are compiled and an assessment is made determining the types and spatial extent of missing data. A follow-up field study is then designed and implemented with the main objective to fill in data gaps. The data type most frequently missing is PCB water-based data and applies to ambient water and point/non-point sources. In the past, ambient water samples were collected and analyzed but the analytical results failed to detect PCBs at the EPA method 608 detection limits (40 CFR, Appendix A, Part 136, EPA method 608, 7-01-03 edition). The same is true for point sources where PCB compliance monitoring is required within NPDES permits using the same method.

The purpose of this study was to develop sample collection methods combined with a sensitive analytical approach for use within PCB impaired waterbodies in Virginia. To meet the needs of TMDL development, the approach must: 1) provide low level PCB data on possible sources, including NPDES permitted outfalls, and 2) provide ambient water data used to calibrate models and to assist in the simulation of the fate and transport of PCBs. EPA Method 1668A (EPA 1999) was used to analyze water based samples, including effluent from point sources and ambient river water, collected at locations identified within the first step of the TMDL source identification study. The method uses high resolution gas chromatography/high resolution mass spectrometry (HRGC/HRMS) which has the capability to measure the full spectrum of 209 PCB congeners at very low levels (5-10 pg/L range per congener).

METHODS

The study areas containing PCB fish impairments include the tidal Potomac River, with focus on Virginia's tributaries and embayments, and the Roanoke/Staunton River which are located in western/south central Virginia. Ultra-clean sampling procedures for the collection of effluent

were adopted from Delaware River Basin Commission (DRBC) with modification, and incorporated into a field sampling protocol (VADEQ 2005). Stakeholder input was considered in the development of these procedures. Most effluent samples were collected during dry flow condition which is defined as no rain exceeding 0.1 inch within 72h from when the samples were collected. American Sigma 900MAX Portable Samplers, or similar units, were used to collect 24h composite samples. New Teflon lined hose and flex tubing was solvent rinsed with Pesticide Grade Methanol. All ancillary equipment was washed with laboratory grade soap and solvent rinsed. Prior to effluent collection, rinsate blanks were collected by pumping PCB free deionized water obtained from the test laboratory through the pumping system. Ambient water samples were collected from shallow water by wading and from deep water either from a bridge or a boat. Sampling bottles were submerged to mid depth and pointed upstream in shallow water and weighted in deeper water and lowered to mid-depth. Samples were collected under base flow condition and when possible at high flow under a falling hydrograph. All PCB samples were collected directly into certified, pre-cleaned 2.3L bottles obtained from Fisher Scientific at a nominal volume of 2L. Sample preservation adhered to guidelines specified by EPA Region III (EPA 2004). Total suspended solids (TSS) and total organic carbon (TOC) samples were obtained concurrently with the PCB water samples.

The EPA performance based Method 1668A (EPA 1999) was used for PCB analysis. Data were generated using HRGC/HRMS from three different laboratories (U.S. EPA Fort Meade, Analytical Services and Quality Assurance Branch (ASQAB); Texas A&M, Geochemical and Environmental Research Group (GERG) in College Station, TX, and Battelle in Columbus, OH), although for consistency the GERG laboratory was exclusively used for effluent analysis for the Potomac River project. The HRGC/HRMS instrumentation was lab specific with use of either a SPB-Octyl or DB-5 column. Since GERG and Battelle were used for the Potomac ambient samples, there was a minor level of inconsistent reporting of congeners due to co-elution. Upon receipt at the laboratories, the water samples were extracted, purified (acid treatment) and analyzed following the protocols contained in EPA method 1668A. By doubling the sample volume and adding other laboratory specific modifications, reporting levels were typically in the ≤ 10 pg/L range on a PCB congener basis. All samples were analyzed as whole water with the end tPCB result consisting of the combined dissolved and particulate PCB fractions.

It is not unusual to observe low level PCB background concentrations in laboratory blanks with the use of an ultra-sensitive method. Consistent with DRBC approach, associated data with laboratory blanks and that exceeded 300 pg/L were not utilized. To account for the mean background (blank) tPCB concentration of 147 pg/L ($n = 10$) with the Potomac River effluent data-set, a statistically based method described by Ferrario et al. (1997) was used to adjust (subtract) the overlapping congeners. Estimated concentrations, which or data flagged with a "J", are congeners detected below the quantitation limit (QL) but above the detection limit (DL), were also included in the tPCB concentration. The Roanoke/Staunton data were not adjusted with the statistically based method due to the size limitation in the laboratory blanks ($n < 10$).

RESULTS

Results generated using highly sensitive method 1668A show present day inputs of PCBs. Table 1 includes a summary of tPCB data from municipal or federal facility based wastewater

treatment plants (WWTP) that discharge to the tidal Potomac River watershed and a mixture of industrial/municipal facilities that discharge to the Roanoke/Staunton River. The maximum observed concentration from a Virginia WWTP in the Potomac study was 2,447 pg/L which was less than that observed from Maryland (4,557 pg/L) or Washington D.C. (2,770 pg/L) facilities. Mean unadjusted effluent results for the Potomac estuary WWTPs are also found in Table 1. The highest level of tPCB observed in either study area was observed in effluent from a Textile Plant on the Roanoke/Staunton River at 60,372 pg/L. A second sample was collected at a later date and yielded a tPCB concentration of 6,888 pg/L.

Table 1. tPCB results from effluents collected as part of the Potomac River and Roanoke River TMDL source identification studies.

Summary Statistics	Potomac River TMDL Study			Roanoke River TMDL Study
	VA (11-facilities)	MD (4-facilities)	Wash. DC (1 facility)	Roanoke/Staunton River (5-Facilities)
n	18	8	5	6
Minimum tPCB (pg/L)	0	3	347	50
Maximum tPCB (pg/L)	2,447	4,557	2,770	60,372
Mean tPCB (pg/L)	431	1,042	1,305	11,813
Mean (Unadj) tPCB (pg/L)	492	1,153	1,409	No Applicable

Targeting the full list of 209 PCBs has revealed congeners that would not ordinarily be detected using Method 608 or other congener specific method with a shortened list of analytes. Figure 1 includes a list of congeners that contributed $\geq 1\%$ of the bulk mass to the tPCB concentration of the Textile Plant effluent on the Roanoke/Staunton River. Non-Aroclor PCB congeners 155 and 184 were detected in the effluent from a sample collected in January 2006 and again in August 2007 and account for 83% and 56% of the respective tPCB concentration. The data were generated from two different labs that perform the EPA method. PCB congeners 155 and 184 have also been detected several miles downstream in ambient water samples (PCB 155 - 5.09 pg/L to 246 pg/L; PCB 184 – 6.67 to 251 pg/L).

Total PCB data from ambient water samples were collected under low and high flow conditions from 24 stations located within urban/suburban/rural tributaries to the Potomac River (Table 2). The maximum tPCB concentration was much lower under low flow (1,277 pg/L) when compared to high flow (52,365 pg/L) at the same station. A strong relationship ($R^2 = 0.8817$) between tPCB concentration and Total Suspended Solids (TSS) in PCB contaminated Virginia urban tributaries is shown in Figure 2. The relationship between PCB and TOC was poor within these same urban tributaries ($R^2 = 0.1493$).

Ambient water data collected at five stations from the Roanoke/Staunton River study showed the same trend for tPCB concentration under low and high flow conditions (Table 2.) A maximum observed concentration under low flow was 262 pg/L and 1,317 pg/L at high flow. Since TSS

data were lacking at some stations, a relationship could not be established. PCB data were correlated against TOC and did show a good relationship ($R^2=0.6886$).

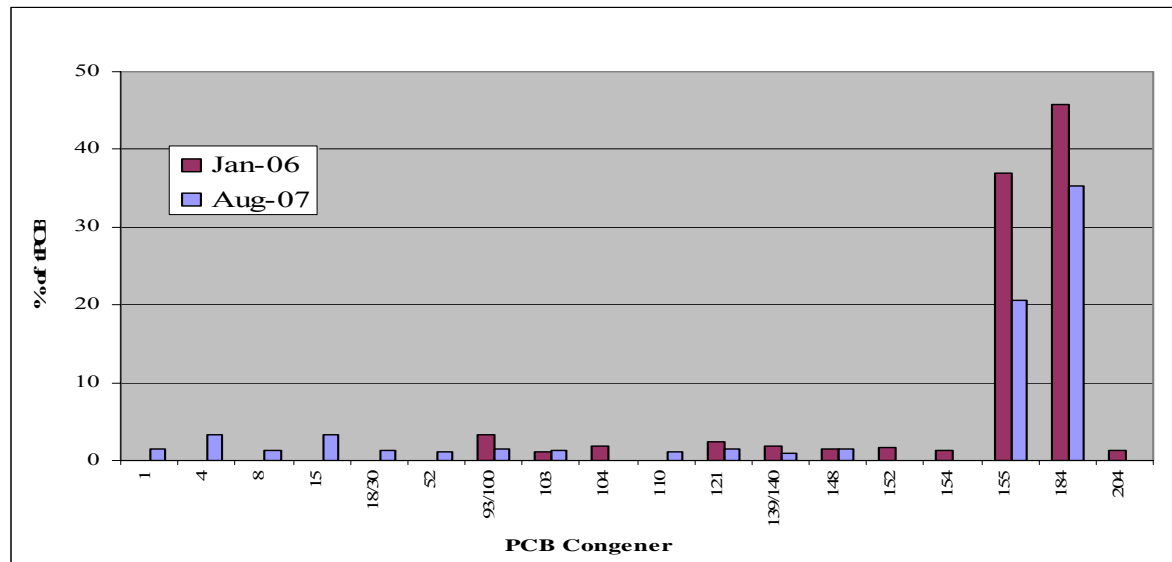


Figure 1. PCB congeners contributing to the bulk PCB mass in effluent collected from a Textile Plant on two occasions. Listed congeners make-up $\geq 1\%$ of tPCB concentration.

Table 2. Ambient tPCB data generated using EPA Method 1668A. Samples were collected under low/high flow conditions at specific stations across several tributaries (24 stations) on the Potomac River. The Roanoke study included five stations.

Summary Statistics	Potomac River TMDL Study			Roanoke River TMDL Study	
	Low Flow	Elevated Flow	High Flow	Low Flow	High Flow
n	23	10	22	5	4
Minimum tPCB (pg/L)	0	50	190	0	465
Maximum tPCB (pg/L)	1,277	1,295	52,365	262	1,317
Mean tPCB (pg/L)	371	339	3,278	86	912

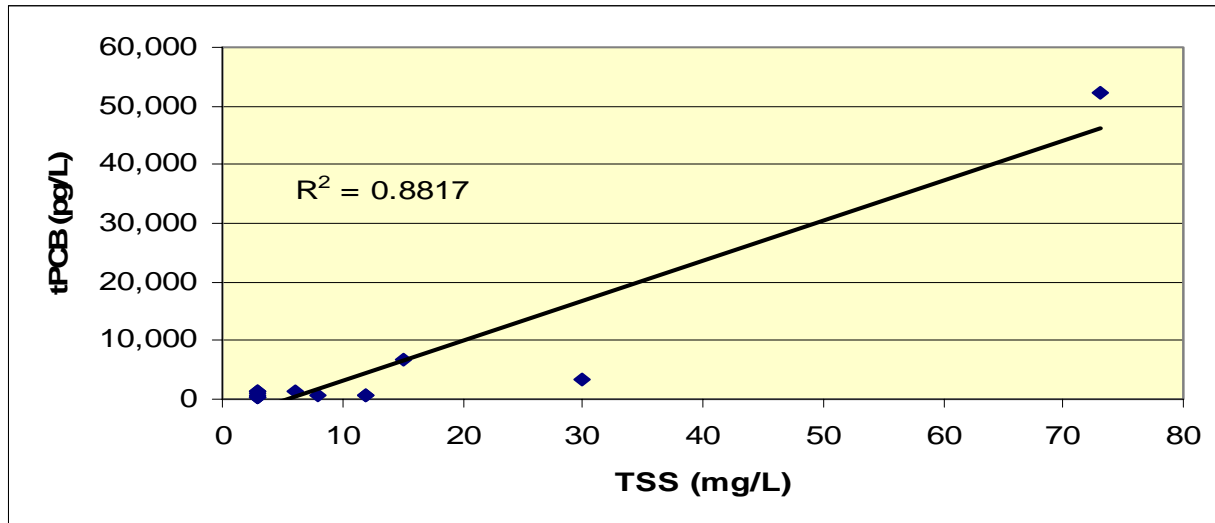


Figure 2. Relationship between tPCB and TSS for six Virginia urban tributaries to the Potomac River.

DISCUSSION

With the production of PCBs discontinued in the late 1970's, the common belief is all environmental releases have been associated with past activities (Erickson 1997). Consequently, PCBs have been termed a legacy issue and are viewed as being confined to sediments and therefore the main source of contamination to biota. With past results in Virginia's NPDES compliance monitoring program yielding non-detects for PCBs, the association of PCBs as a legacy contaminate has become well established. However, a study performed in New York/New Jersey Harbor (Durell et al. 1998) and a TMDL development study in Delaware Bay (DRBC) have shown there are present day, on-going sources of PCBs from permitted facilities. These studies utilized highly sensitive analytical methods that detected PCBs on a congener basis at nanogram per liter concentrations from municipal and industrial wastewater sources. It is easily understood why PCBs have not been detected in Virginia at current environmental levels as PCB compliance Method 608 has elevated detection levels (0.5 – 50 ug/L). Furthermore, method 608 only targets Aroclors, which according to Rushneck et al. (2004), are subjected to both environmental weathering and overlapping of commercial mixtures that can complicate the ability to identify them within environmental samples. Although WWTPs may be considered a loading source, they do not actually produce PCBs. Rather, they may serve as a pass-thru for contamination that may exist within their system from past spills, or may be introduced into the system during rain events from inflow and infiltration, for example.

EPA's PCB Method 1668A has allowed characterization of PCB sources in the Tidal Potomac Estuary. While the PCB results for the Potomac River WWTPs range from very low to exceeding the WQC, these concentrations were a magnitude or more lower than those observed in New York harbor or the Delaware River (Durrell et al. 1998, DRBC), which may not be surprising since the Washington D.C. metro area is not as heavily industrialized. In a few cases, the tPCB effluent concentrations were similar to background blanks. Since the PCB method is very capable of detecting low level PCBs, background contamination is not that unusual, but can be of concern if contributing to the final concentration. However, by using an analytical method

that provides congener specific data for quality assurance (i.e., laboratory blanks) and effluent samples, reported concentrations for each congener can readily be compared and adjusted using a statistical approach (Ferrario et al. 1997). The Potomac River results show that background PCB levels have a very small affect on the tPCB concentration, especially at elevated levels (e.g., >1,000 pg/L). Simply subtracting a laboratory blank tPCB concentration from an effluent sample tPCB concentration would be erroneous as the congeners contributing to the blank are not necessarily the same found in the effluent sample.

The results generated on the Roanoke/Staunton River provide additional evidence point sources are contributing PCB loading to aquatic systems. Unlike WWTPs, there are cases where industrial operations may inadvertently produce non-Aroclor PCBs (Erickson 1997) with the availability of carbon, chlorine, and high temperatures within certain processes (Panero et al. 2005). The PCB congeners (155 and 184) comprising the bulk PCB mass of the Textile Mill discharge are unusual as they are not Aroclor congeners, and would go undetected without the full congener scan. The validity of the congeners has been substantiated by the EPA Fort Meade (per. comm. with Stevie Wilding) as both met the method quality assurance requirements. The origin of these congeners should be explored further.

The approach used for ambient water collection with the Method 1668A analysis provided instream water data needed for TMDL model development. The HRGC/HRMS instrumentation coupled with method 1668A modifications and collection of 2-4L volume appears to be adequate to measure instream concentrations, even when concentrations are depressed under low flow. Of course when TSS and/or TOC concentrations are elevated, ambient water PCB concentrations would be expected to be at increased levels based on their affinity for solids and carbon. Data sets from both study areas showed this relationship in PCB contaminated areas. Based on this relationship, PCB loadings were predicted from all tributaries to the tidal Potomac River through regression analysis (Haywood et al. 2007).

CONCLUSIONS

Current inputs of PCBs are entering Virginia's waters and are contributing to fish tissue impairments. To account for the complete mass loading and instream transport, a scientifically sound approach is necessary to collect and analyze the full suite of 209 PCB congeners at low, environmentally relevant levels (pg/L). With appropriate performance based modifications, EPA method 1668A will provide data to meet the needs for TMDL development.

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PCB BIOACCUMULATION FACTORS IN POTOMAC ESTUARY FISH

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KEY WORDS: polychlorinated biphenyls, PCB, bioaccumulation, fish, Potomac

ABSTRACT

Species-specific bioaccumulation factors for polychlorinated biphenyls (PCBs) were developed for fish in the interstate Potomac River estuary, including tidal fresh waters. Bioaccumulation factors relating fish to water column (total BAFs) and sediment (SedBAFs) were calculated from recent PCB data collected by or for Virginia, Maryland and the District of Columbia agencies. Resident species exhibiting the strongest tendency to accumulate PCBs, as indicated by the high total BAFs and SedBAFs, were channel catfish, gizzard shad, pre-migratory striped bass, and common carp. When total BAFs and SedBAFs are normalized to fish lipid content and freely-dissolved water column PCBs or sediment organic carbon content, the species most susceptible (sensitive) to accumulating PCBs from their environment include pre-migratory striped bass, largemouth bass, and channel catfish, and possibly white catfish, bluegill sunfish, and pumpkinseed sunfish. PCB gradients occur in the different media with the highest PCB concentrations in the Washington metropolitan area. PCB concentrations in fish tissue, water column, and sediment appear to be approaching, or at, steady state in the tidal system. Virginia, Maryland and the District selected target species and used their BAFs (adjusted to a common condition) to establish water and sediment target concentrations for the tidal Potomac TMDL. The water quality standard for PCBs is about equal to the BAF-based water column PCB target in the District, but is 1.5 times higher in Maryland and 26.6 times higher in Virginia. Hence, fish failing the fish tissue threshold are found in waters that pass the water quality standard.

INTRODUCTION

Water column and surface sediment targets for PCB concentrations can be derived by dividing a jurisdiction's fish tissue screening threshold by some factor that represents the fish's ability to absorb and retain PCBs. US Environmental Protection Agency guidelines (EPA 2003) recommend developing a bioaccumulation factor (BAF) for persistent hydrophobic chemicals such as PCBs. This factor, called a "total" BAF, is the ratio of the PCB concentration in an organism's wet tissue to its concentration measured in waters where both the organism and its food and environment are exposed to PCBs. Another factor, the SedBAF, relates fish PCB concentration to sediments rather than water. Bioaccumulation factors depend on an assumption that PCBs in fish tissue, water, and sediments are relatively stable over months or years. Total BAFs and SedBAFs were developed and used to establish water column and sediment PCB targets for the tidal Potomac PCB Total Maximum Daily Load (TMDL) study (Haywood and Buchanan 2007). A fate and transport model (LimnoTech 2007) was run to determine PCB loading rates that result in the targets being achieved everywhere in the estuary. This paper

summarizes how the BAFs were developed and further examines the results.

METHODS

Total BAFs and SedBAFs were calculated for 23 and 21 species, respectively, from the available fish tissue (2000-2005), water column (2002-2006), and surface sediment (2000-2005) PCB data for the Potomac estuary. Samples were collected by or for the Virginia Department of Environmental Quality, Maryland Department of the Environment, and the District of Columbia Department of the Environment, and analyzed for PCBs by Chesapeake Biological Laboratory, Virginia Institute of Marine Science, or the US Fish & Wildlife Service (ICPRB 2007). Only fish samples collected in tidal waters were used in the analysis. Each fish sample (n = 204) was assigned a trophic level of predator (primarily piscivorous), planktivore (consumes primarily phyto- and/or zooplankton), or benthivore-generalist (opportunistic foragers consuming stream “drift” and/or benthic invertebrates), and a home range radius of 2, 5, or 10 miles, depending on species characteristics. Striped bass less than 560 mm were included because they have typically not begun to migrate (Setzler-Hamilton and Hall 1991).

Each fish sample was associated with all water column and sediment PCB data in the area within a radius equal to the home range radius of the species. Median values of water and sediment data collected in each Potomac PCB model segment were calculated; then the means of all values for model segments wholly or partly inside each home range radius were calculated. This approach weighs the available data more evenly across the home range than a grand average. Due to the hydrophobic nature and high partition coefficients of PCBs, EPA (2003) recommends using the average of multiple water or sediment samples taken in the fish’s habitat over sufficiently long time periods. Individual fish samples in the 2000-2005 period could be associated with as many as 46 water column or 81 sediment samples in the frequently sampled Washington DC metropolitan area. However, the median number of ambient water column and sediment samples available to calculate the factors was 2.5 and 2, respectively, for species with 2 mile home range radiuses, 6 and 3 for species with 5 mile home range radiuses, and 13 and 6 for species with 10 mile home range radiuses.

Total PCB (tPCB) concentrations in fish were used as reported. Significant inter-laboratory differences in recovery of PCB homologs 1 and 2 prevented the used of water and sediment tPCBs concentrations as reported. To use all available homolog 3-10 (PCB3+) data, water column and sediment tPCB concentrations were derived by dividing the reported PCB3+ concentrations by 0.92 (water column) or 0.90 (sediments).

Total BAFs were calculated for each fish sample using Equation 2-2 in EPA (2003):

$$\text{Total BAF} = \frac{[\text{tPCB}]_{\text{tissue}}}{[\text{tPCB}]_{\text{water}}} \quad (1)$$

where $[\text{tPCB}]_{\text{tissue}}$ is tPCB concentration in fish wet tissue (ng/kg) and $[\text{tPCB}]_{\text{water}}$ is the water column tPCB concentration in the fish’s home range area (ng/liter). Total BAFs for PCBs vary depending on the individual lipid concentrations of each fish and the water column concentration of freely-dissolved PCBs in the home range area. Total BAFs were normalized to each sample’s fish lipid content and the freely-dissolved PCB concentration, and the resulting baseline BAFs

were used to quantify each species' sensitivity to PCB bioaccumulation. The baseline BAF is calculated using Equation 2-3 in EPA (2003):

$$\text{Baseline BAF} = \frac{[\text{PCB}]_{\text{tissue}} / \% \text{lipid}}{[\text{PCB}]_{\text{water}} \cdot \% \text{fd}} = \left[\frac{\text{totalBAF}}{\% \text{fd}} - 1 \right] \cdot \frac{1}{\% \text{lipid}} \quad (2)$$

where %fd is the fraction of the total PCB concentration that is freely-dissolved in the water column in the fish's home range area, and %lipid is the fraction of tissue that is lipid in the fish sample. The fraction of freely-dissolved PCB is related to dissolved and particulate organic carbon concentrations in the water column, and was calculated with equation 4-6 in EPA 2003. Ambient values ranged from 9% - 50% in water column samples associated with the tidal fish tissue samples, with a median value of 29.2%. For the purpose of comparing species across different environmental conditions, the median of each species' baseline BAFs was normalized to a common condition, i.e., the median lipid content of the species and a freely-dissolved PCB concentration representative of the ecosystem:

$$\text{Adjusted total BAF} = ((\text{median baseline BAF} \cdot \text{median \%lipid}) + 1) \cdot 0.292 \quad (3)$$

The resulting adjusted total BAF for each species has no variability due to individual differences in lipid content or to spatial or temporal differences in freely-dissolved PCB concentrations.

The SedBAF, or sediment BAF, relates fish tissue and surface sediment PCBs and, like the total BAF, it reflects the fish's exposure to all relevant exposure routes. SedBAFs can be used to confirm the total BAF results or to predict total BAFs when water column concentrations of PCBs are low or variable and hence difficult to measure (EPA 2003). The SedBAF is calculated by dividing the fish tissue PCB concentration (ng/g wet weight) by the sediment PCB concentration (ng/g sediment dry weight). SedBAFs vary depending on individual lipid concentrations and the sediment fraction of organic carbon (%SOC) in the home range area. SedBAFs were normalized to each sample's fish lipid content and the area's %SOC (Equation 2-14 in EPA 2003). The resulting biota-sediment bioaccumulation factors, or BSAFs, serve as a second measure of each species' sensitivity to PCB bioaccumulation. An adjusted SedBAF for each species was derived by normalizing the median of the species' BSAFs to the median lipid content of the species and a %SOC of 2.85%, representing the ecosystem:

$$\text{Adjusted SedBAF} = \text{median BSAF} \cdot (\text{median \%lipid} / 0.0285) \quad (4)$$

RESULTS

Species-specific total BAFs, baseline BAFs, and adjusted total BAFs are presented in Table 1 and SedBAFs, biota-sediment BAFs (BSAFs), and adjusted SedBAFs are presented in Table 2. Total BAFs for fish species inhabiting tidal waters of the Potomac River are variable, with values in an individual species ranging as much as 40-fold (Figure 1). Species median values in liter per kg tissue wet weight range 34-fold from 16,200 (brown bullhead catfish) to 548,000 (gizzard shad). The baseline BAF values indicate pre-migratory striped bass, largemouth bass, and channel catfish, and possibly white catfish and the bluegill and pumpkinseed sunfishes are most sensitive to PCB bioaccumulation, or most readily absorb and maintain PCBs from their environment. The latter three species are less certain because sample sizes are small.

SedBAF values within each species vary proportionally as much as the total BAF values. Sediment PCB concentrations are generally believed to be less changeable over time than water column PCB concentrations, so the SedBAF variability is likely related to the spatial gradients apparent in sediment PCB concentrations. For example, the Anacostia sediments have distinct local hot spots adjacent to the Washington Navy Yard and below Watts Branch (Haywood and Buchanan 2007). Median SedBAF values, in g sediment dry weight per g wet weight tissue, ranged 32-fold from 0.22 (black crappie) to 7.08 (channel catfish). BSAFs indicate pre-migratory striped bass, largemouth bass, channel catfish, and common carp and possibly white catfish and bluegill sunfish most sensitive to PCB bioaccumulation (Table 2).

DISCUSSION

The variability evident in the individual total BAFs (Figure 1) and SedBAFs is expected for several reasons. Fish tissue PCB concentrations, the numerator in the BAF equation, vary depending on the lipid content and food habits of each individual fish. Benthic food pathways typically expose fish to higher PCBs than planktonic pathways. Seasonal changes in fish lipid contents and food habits will affect tissue concentrations. Concentrations of water and sediment PCBs, the denominators in the BAF equation, vary spatially in the Potomac estuary due to uneven external loadings. The highest concentrations are found in tidal freshwater around and just downstream of the Washington DC metropolitan area (Haywood and Buchanan 2007). Fish moving in their home ranges may experience different levels of PCBs depending on their geographic location. Small numbers of water and sediment samples used in calculating the denominator also adds to uncertainty in the BAF estimates. The non-normal distributions of the BAFs and SedBAFs suggest the median is a better measure of central tendency than the arithmetic mean. Until more samples are collected, most confidence should be placed in species BAFs and SedBAFs derived with larger sample numbers: channel catfish, white perch, largemouth bass, common carp, pre-migratory striped bass, and gizzard shad.

Bioaccumulation that approximates a steady state condition, where fish tissue and ambient PCB concentrations are relatively constant over the life time of the fish, increases the predictive value of the BAFs (EPA 2003). Steady states can occur before equilibrium is reached between external loads and ambient conditions, a process that will take an estimated 50+ years in the tidal Potomac (LimnoTech 2007). The tidal Potomac system may be close to, or at, steady state conditions. When PCB tissue concentrations for the six most sampled species (above) are grouped into three relatively homogenous regions, there are no consistent differences and few significant differences between 2000-2002 and 2003-2005 (Figure 2). The three regions are the Anacostia River, the tidal fresh (upper) Potomac and tributaries, and the oligohaline-mesohaline (middle-lower) Potomac and tributaries. Similarly, no consistent differences in sediment concentration are seen at the few river locations where 2000 data overlap 2003 and 2005 data. Finally, values of another indicator of steady state, the sediment-water quotient (Π_{socw}) to *n*-octanol water partition coefficient (K_{ow}) ratio (EPA 2003), support the assertion of steady state conditions. Medians of the ratios computed for the Anacostia, upper Potomac and middle-lower Potomac regions all fell between 2 and 10, values that indicate a steady state.

Table 1. Species-specific total BAFs (Eqn. 1), baseline BAFs (Eqn. 2), and adjusted total BAFs (baseline BAF normalized to each species median lipid fraction and a single, representative freely-dissolved PCBs fraction of 29.2%). * Species with highest baseline BAFs.

Trophic Level	Species		Total BAF (median)	Baseline BAF (median)	Lipid fraction (median)	Adjusted total BAF
		n	liter/ kg wet wt	liter/kg lipid		liter/ kg wet wt
planktivore	Gizzard Shad	12	548,000	9,790,000	0.296	845,000
benth/gen	Channel Catfish	37	306,000	* 20,000,000	0.058	341,000
predator	Striped Bass (Pre-Mig)	13	259,000	* 38,000,000	0.024	268,000
benth/gen	Common Carp	17	240,000	8,940,000	0.055	144,000
benth/gen	Pumpkinseed Sunfish	5	53,800	* 26,800,000	0.013	103,000
predator	White Perch	26	65,500	10,500,000	0.027	82,000
benth/gen	Yellow Perch	2	69,000	11,900,000	0.022	77,700
benth/gen	Mummichog	3	50,900	3,600,000	0.062	65,700
predator	Atlantic Croaker	4	51,500	843,000	0.266	65,500
predator	Largemouth Bass	19	91,300	* 26,200,000	0.008	61,500
benth/gen	White Catfish	3	112,000	* 26,100,000	0.008	58,700
planktivore	Bluegill Sunfish	3	38,600	* 27,900,000	0.007	57,100
predator	Black Crappie	1	98,900	5,750,000	0.032	54,000
predator	American Eel	9	66,600	1,540,000	0.090	40,400
predator	Blue Catfish	3	38,700	5,400,000	0.025	38,800
benth/gen	Brown Bullhead Catfish	5	16,200	5,940,000	0.022	38,500
planktivore	American Shad	3	54,700	1,250,000	0.104	38,100
benth/gen	Spot	9	23,500	814,000	0.138	32,700
predator	Bluefish	8	24,900	1,880,000	0.051	28,100
benth/gen	Green Sunfish	1	47,900	380,000	0.190	21,000
predator	Grey Trout (Weakfish)	1	18,400	370,000	0.186	20,100
benth/gen	Yellow Bullhead Catfish	2	22,500	2,680,000	0.024	18,500

Table 2. Species-specific SedBAFs (Eqn. 3), BSAFs (Eqn. 4), and adjusted SedBAFs (BSAF normalized to each species median lipid fraction and a single, representative sediment organic fraction of 2.85%). * Species with highest BSAFs.

Trophic Level	Species	n	SedBAF (median)	BSAF (median)	Lipid fraction (median)	Adjusted SedBAF
			g sed dry wt/ g wet wt	g SOC/ g lipid		g sed dry wt / g wet wt
benth/gen	Channel Catfish	39	7.08	* 3.68	0.058	7.52
benth/gen	Common Carp	13	4.44	* 3.04	0.055	5.87
predator	Striped Bass (Pre-Mig)	13	4.77	* 4.84	0.032	5.40
planktivore	Gizzard Shad	9	6.27	0.404	0.296	4.19
predator	American Eel	9	2.57	1.10	0.090	3.46
predator	White Perch	26	1.47	1.91	0.027	1.79
predator	Grey Trout (Weakfish)	1	1.64	0.254	0.186	1.66
planktivore	Bluegill Sunfish	2	1.34	* 5.36	0.007	1.32
predator	Largemouth Bass	17	1.59	* 4.57	0.008	1.29
predator	Blue Catfish	3	1.37	1.47	0.025	1.27
predator	Atlantic Croaker	4	1.36	0.124	0.266	1.16
benth/gen	Pumpkinseed Sunfish	5	1.37	* 2.30	0.013	1.06
predator	White Catfish	3	2.09	* 3.88	0.008	1.05
benth/gen	Brown Bullhead Catfish	4	0.846	1.28	0.022	1.00
benth/gen	Spot	9	0.675	0.172	0.138	0.830
benth/gen	Yellow Perch	2	0.683	1.02	0.022	0.801
planktivore	American Shad	3	1.07	0.204	0.104	0.747
benth/gen	Yellow Bullhead Catfish	2	0.560	0.864	0.024	0.716
predator	Bluefish	8	0.869	0.333	0.051	0.597
benth/gen	Mummichog	2	0.655	0.237	0.062	0.519
predator	Black Crappie	1	0.222	0.180	0.032	0.203

The baseline BAF which references water column PCBs and the BSAF which references sediment PCBs both characterize the sensitivity of a fish to PCB exposure from all pathways. The correlation coefficient between 181 paired values of baseline BAFs and BSAFs from all locations in the estuary was 0.67 ($p << 0.01$). This good agreement indicates BSAFs in the tidal Potomac could be used to predict total BAFs, as suggested by EPA (2003), when the distribution of PCBs in sediment and water is known. When species baseline BAFs and BSAFs are ranked and the ranks averaged, the most sensitive of the well sampled ($n > 10$) species in tidal waters is pre-migratory striped bass followed by largemouth bass, channel catfish, and common carp. Although less well sampled, bluegill sunfish, white catfish, and pumpkinseed appear to be sensitive species also. There is some evidence of biomagnification in the baseline BAF and

BSAF results: three of the seven most sensitive species are predators.

Four of the seven sensitive species—largemouth bass, white catfish, and bluegill and pumpkin sunfish—do not have high total BAF or SedBAF values (Tables 1 and 2). These species, all commonly found in tidal fresh and nontidal waters in the Potomac basin, have low lipid fractions ($\leq 1.3\%$) and will exhibit low PCB contamination on a wet weight basis. Conversely, gizzard shad with a moderate sensitivity to PCB accumulation but very high lipid content has the highest total BAF and second highest SedBAF. Common carp, channel catfish, and pre-migratory striped bass also have high BAFs. These latter species stand out as ones exhibiting the strongest responses to PCBs, a consequence of their particular combinations of lipid content, PCB exposure pathways, and sensitivity to PCB bioaccumulation.

Species comparisons across the variety of Potomac tidal waters are improved when total BAFs and SedBAFs are adjusted to a single condition with respect to water column freely-dissolved PCB concentration and sediment organic carbon content. This removes variability caused by different water-sediment PCB distributions. The noticeable differences between the adjusted and unadjusted BAFs of some species (Tables 1 and 2) are the result of dissimilar sediment-water PCB distributions at the tidal Potomac fish sampling sites. In the urban Anacostia River, PCB concentrations normalized to units of particulate organic carbon are about 2.9 times higher in the sediments than the water column. In the upper, tidal fresh Potomac River, carbon-normalized PCB concentrations in the water column and sediments are about equal. In the middle-lower tidal Potomac, water column PCBs are slightly higher than sediment PCBs and both decline with distance downstream. Adjusting the BAFs changes the rankings of the species, but the four species with the highest total BAFs and SedBAFs, i.e., channel catfish, gizzard shad, pre-migratory striped bass and common carp, retain the highest ranks.

The Virginia, Maryland, and District of Columbia agencies developing the tidal Potomac TMDL each selected a target fish species and used the species' adjusted total BAF and adjusted SedBAF to calculate target PCB concentrations for the water column and sediments, respectively, in their jurisdictional waters (Table 3). The target species were those sampled by the jurisdiction with the highest total BAF or SedBAF values. Maryland and the District did not select gizzard shad as the target species for their total BAFs because they currently do not measure PCBs in that species. PCB TMDL loads that lead to attainment of these PCB target concentrations will be protective of the target species and all species with lower BAFs. The current water quality standard for PCBs is about equal to the BAF-based water column PCB target in the District and 2.5 times higher in Maryland however it is 26.6 times higher in Virginia. As a result of this difference, approximately 53% of the Virginia tidal Potomac fish samples that violated the state's fish tissue threshold were found in waters that did not violate the state's current water quality standard.

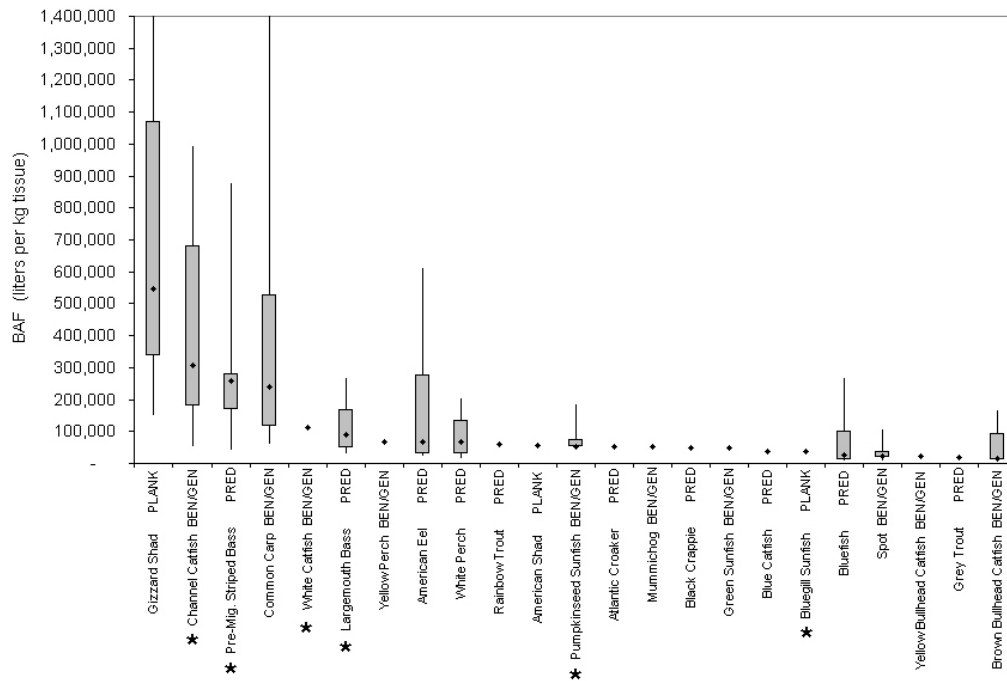


Figure 1. Total BAFs. Lines, 5th% - 95th%; boxes, 25th% - 75th%; dot, median.
 * Species with highest *baseline* BAFs are most sensitive PCB accumulation.

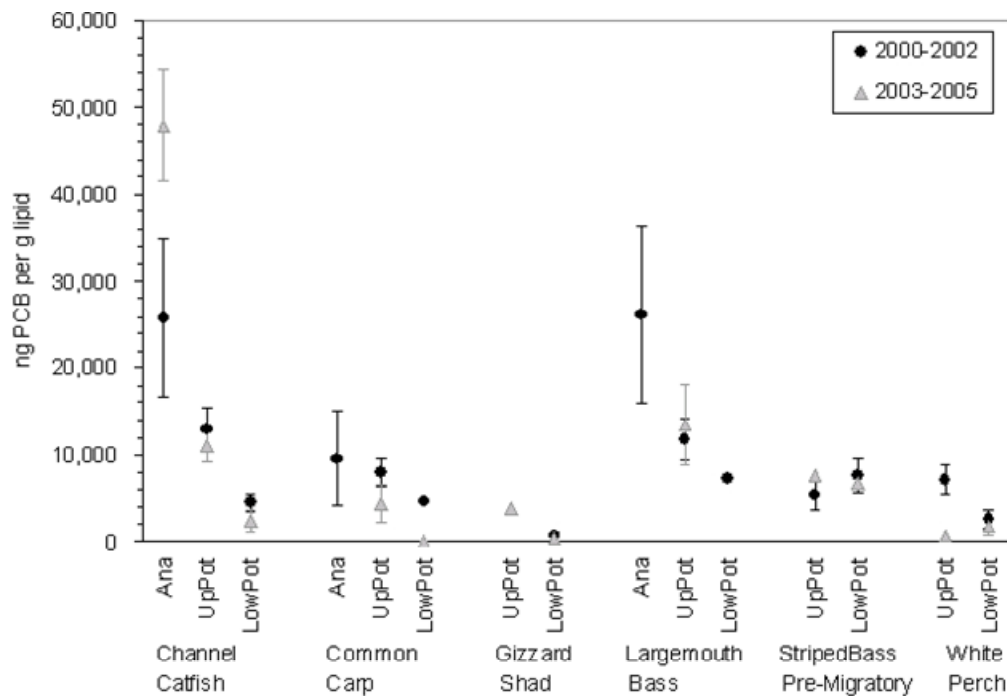


Figure 2. Comparison of 2000-2002 and 2003-2005 average PCB concentrations normalized to units of lipid for the six most frequently collected fish species in the tidal Anacostia R (Ana), the upper tidal Potomac R mainstem & tributaries (UpPot), and the middle and lower tidal Potomac R mainstem & tributaries (LowPot). Standard error is shown for sample $n > 2$.

Table 3. Jurisdiction current fish tissue thresholds and water quality standard criteria for PCBs, and BAF-based water column and sediment PCB targets.

Juris.	Current Fish Tissue Threshold, ng/g wet wt	Current Water Quality Standard Criteria, ng/liter	Calculated Water Column Target, ng/liter (Target Species)	Calculated Sediment Target, ng/g sed dry wt (Target Species)
DC	20	0.064	0.059 (channel catfish)	2.7 (channel catfish)
MD	88	0.64	0.26 (channel catfish)	12 (channel catfish)
VA	54	1.7	0.064 (gizzard shad)	7.2 (channel catfish)

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SAMPLING FOR EMERGING CONTAMINANTS IN THE SOUTH BRANCH OF THE POTOMAC RIVER AND OTHER WEST VIRGINIA STREAMS 2004-2006

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ABSTRACT

In 2003 and 2004 West Virginia DNR officials and USGS scientists examining fish kills and widespread incidences of fish lesions in the South Branch Potomac River, discovered many reproductive anomalies in smallmouth bass (*Micropterus dolomieu*). The anomalies were both widespread and pervasive within populations and included egg production in male fish. These reproductive anomalies, referred to as an intersex condition, have been found in up to 80% of the male smallmouth bass sampled at sites in the South Branch Potomac River Basin. Earlier studies by USGS researchers had found feminization of male common carp (*Cyprinus carpio*) in the Shenandoah River near Millville, W. Va., another tributary of the Potomac River. Abnormal development of reproductive tissues may result from disruptions of the endocrine system brought about by exposure to anthropogenic sources of hormones and synthetic organic compounds that mimic hormones. The USGS has used passive water-sampling devices to characterize the chemical environment at sites in the South Branch Potomac River Basin, the Cacapon River, and several comparative reference sites outside of the Potomac drainage. Analyses of passive sampler extracts indicated the presence of pesticides, flame retardants, and personal care product residues in stream water. Several of these compounds, some known or suspected as endocrine disrupting compounds (EDCs), were also found in blood plasma collected from fish with intersex conditions. Preliminary results of passive sampler extract bioassays indicate a gradient of estrogenic activity among sampled sites. Investigations are continuing and plans for future sampling will be presented.



Session III-B

Using Biological Assessments as Indicators of Stream Health

- Stream Monitoring Methods Suitable for Citizen Volunteers Working in the Coastal Plain and Lower Piedmont Regions of Virginia
- Correspondence of Benthic Macroinvertebrate Communities with a Water Chemistry Based Classification of Streams in a Mining Influenced Region
- Knowledge Is Power: Options for Improving Virginia's Approach to Benthic TMDL Development
- A Plan for Conducting an Aquatic Life use Attainability Analysis of a Central Appalachian Urbanized Stream

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STREAM MONITORING METHODS SUITABLE FOR CITIZEN VOLUNTEERS WORKING IN THE COASTAL PLAIN AND LOWER PIEDMONT REGIONS OF VIRGINIA

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KEY WORDS: biological monitoring, citizen monitoring, stream health

ABSTRACT

Citizen volunteers can play an important role in stream monitoring programs, but reliable methods to evaluate ecological condition are often lacking for specific ecoregions of the U.S.

We developed and validated a stream monitoring method suitable for the coastal plains and lower piedmont regions of Virginia. The Coastal Save Our Streams (SOS) method is based on four metrics (%Ephemeroptera, Plecoptera, and Trichoptera excluding Hydropsychidae; %Coleoptera; %Tolerant; and %Gomphidae) that are each ranked on a scale of 1-6 based on metric values obtained in field samples, producing a total SOS score ranging from 0 to 24 with higher values indicating that the stream is increasingly similar to undisturbed streams in the region. Scores from the Coastal SOS method were highly correlated with scores from professional assessments in 13 streams used to develop the method ($r=0.93$), and an additional 13 used to validate it ($r=0.91$). We confirmed that trained citizens could successfully apply the method in the field, including obtaining an unbiased sample of the benthic community and accurately classifying organisms into 21 taxonomic categories needed to use the method. Seasonal and annual variation in scores was relatively high, and so we recommend that citizens collect at least three seasons of data over two years before drawing conclusions about the ecological condition of any particular stream.

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**CORRESPONDENCE OF BENTHIC MACROINVERTEBRATE COMMUNITIES
WITH A WATER CHEMISTRY BASED CLASSIFICATION OF STREAMS IN A
MINING INFLUENCED REGION**

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ABSTRACT

We analyzed seasonal water samples from streams within the Cheat and Tygart Valley river basins, West Virginia, USA, in an attempt to classify streams based on water chemistry in this coal-mining region. We also sampled benthic macroinvertebrates at corresponding sites in the Cheat basin. Principal component analysis identified two important trends in water chemistry. The first was largely determined by mining related factors (elevated metals, sulfates, and conductivity) and the second was an alkalinity-hardness gradient. Cluster analysis grouped water samples into six dominant types that we described as Reference, Soft, Hard, Transitional, Moderate acid mine drainage (AMD), and Severe AMD. MANOVA and classification tree analysis confirmed that the types were statistically distinguishable on the basis of chemical constituents related to AMD and acid precipitation. Non-metric multidimensional scaling revealed that benthic macroinvertebrate communities broadly corresponded to, but were highly variable within and among, the dominant water quality types. Water chemistry variables that significantly distinguished water quality types also significantly correlated with macroinvertebrate communities. This research is the first to establish a statistically supported stream classification system based on water chemistry in mined watersheds. The results suggest that human related stressors in combination with geological control mechanisms are responsible for producing distinct water quality types. The correspondence between macroinvertebrate communities and water quality types suggests that communities may respond to remediation efforts designed to improve water quality, and this could be used to establish biological endpoints for restoration.

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KNOWLEDGE IS POWER: OPTIONS FOR IMPROVING VIRGINIA’S APPROACH TO BENTHIC TMDL DEVELOPMENT

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KEY WORDS: TMDL, benthic impairment, general standard

ABSTRACT

Virginia is by all measure a progressive TMDL state, with much to show for its efforts. However, Virginia faces complications in its approach to benthic TMDLs under the general standard. This paper provides a summary of the relevant regulatory authorities and recent benthic TMDL proceedings. It ends with options for improving Virginia’s approach.

INTRODUCTION

Buried in the water quality standards section of the Clean Water Act (“CWA”), and largely ignored for over two decades, the total maximum daily load (“TMDL”) program nonetheless promises to be one of the most challenging and powerful tools for water quality restoration nationwide. In contrast to many other states, Virginia is well ahead of the TMDL curve, and remains on track to meet all of the deadlines dictated by the consent decree in the American Canoe Association case.¹

This success aside, Virginia’s program is not without complications. Among the more obvious ones, Virginia must develop TMDLs for ubiquitous legacy pollutants, like PCBs and mercury, where the sources and causes of impairment may be diffuse and difficult to manage. Virginia also must develop TMDLs for a small but not insignificant number of water segments listed as impaired under the Commonwealth’s “general standard.”² This standard reads as follows:

State waters, including wetlands, shall be free from substances attributable to sewage, industrial waste, or other waste in concentrations, amounts, or combinations which contravene established standards or interfere directly or indirectly with designated uses of such water or which are inimical or harmful to human, animal, plant, or aquatic life.³

Virginia implements this general standard through a bio-assessment protocol known as RBP II. Using this protocol, Virginia assesses the health of the resident benthic macroinvertebrate community. Segments deemed to be unhealthy are then listed as “impaired” and slated for TMDL development.

By definition, a general standard listing means that (1) a relevant numeric water quality criterion is not in place (otherwise, the listing would be based on the criterion), or (2) the

pollutant/stressor causing the benthic impairment is unknown (such that a particular criterion, even if available, cannot be assigned). In either case, the missing link may dramatically complicate the TMDL development process. Although Virginia has succeeded in overcoming these complications in several recent proceedings, Virginia's approach is by no means perfect. This paper explores options for making it better.

BACKGROUND ON THE WATER QUALITY CONTINUUM

Congress enacted the CWA in 1972 with a goal of restoring and maintaining the chemical, physical, and biological integrity of our Nation's waters. In furtherance of those goals, each state must adopt, periodically review and revise water quality standards ("WQS") that consist of "the designated uses of the navigable waters involved and the water quality criteria for such waters based upon such uses."⁴

State WQS must "be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and also taking into consideration their use and value for navigation."⁵ These standards serve a dual role in establishing water quality goals for each waterbody and serving as the regulatory basis for water quality-based control requirements.⁶

In addition to setting WQS, states must assess the quality of their waters and identify those "for which the effluent limitations required by [CWA § 301(b)(1)(A) and (B)] are not stringent enough to implement any water quality standards applicable to such waters."⁷ States must submit their lists of these "impaired waters" to EPA for review and approval every two years.⁸

Since 2002, EPA has published and periodically updated guidance to states on integrating their lists with other assessment information. In this guidance, EPA recommends that states list their waters in one of the following five categories:

- Category 1: All designated uses are met
- Category 2: Some of the designated uses are met but there is insufficient data to determine if remaining designated uses are met
- Category 3: Insufficient data to determine whether any designated uses are met
- Category 4: Water is impaired or threatened but a TMDL is not needed because:
 - 4A: TMDL is already in place
 - 4B: Other pollution control requirements (e.g., best management practices) required by local, State or Federal authority are expected to address all water-pollutant combinations and attain all WQSs in a reasonable period of time
 - 4C: Impairment is not caused by a pollutant
- Category 5: Water is impaired or threatened and a TMDL is needed

The listing of a waterbody as impaired (i.e., on Category 5) triggers the need for a TMDL, which must be established "at a level necessary to implement the applicable water quality standards with seasonal variations and a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality."⁹ A TMDL essentially defines the assimilative capacity of a waterbody.

TMDLs must account for all sources and causes of “pollution,” which EPA defines as “[t]he man-made or man-induced alternation of the chemical, physical, biological, and radiological integrity of water.”¹⁰ States must periodically submit their TMDLs to EPA for review and approval. If a state fails to do so, or if EPA disapproves a submittal, then EPA must establish the required TMDL(s). Many states, including Virginia, are subject to court-ordered deadlines for developing TMDLs or deferring to EPA to do so.

Once in place, TMDLs dictate the limitations that must be assigned in permits for discharges from regulated point sources.

ATTAINABILITY ISSUES

In the 1960’s and 1970’s, many states designated uses that served the purposes of the CWA without regard to their attainability. In so doing, they rejected certain uses, like agriculture, that were appropriate, principal uses. These designations prompted situations where states were forced to impose overly stringent and costly treatment requirements, or simply ignore the underlying standards.¹¹

Recognizing that legitimate factors might prevent a use from being met, EPA proposed regulations in 1982 that would empower states to assess the attainability of a use as part of the standards adoption and revision process. EPA finalized the regulations a year later, and they remain binding today without variation from their original design 25 years ago.¹²

EPA’s regulations identify six scenarios where use attainment is not feasible and, in turn, the use may be removed or adjusted. Those scenarios arise when:

- (1) Naturally occurring pollutant concentrations prevent the attainment of the use;
- (2) Natural, ephemeral, intermittent or low flow conditions or water levels prevent the attainment of the use, unless these conditions may be compensated for by the discharge of sufficient volume of effluent discharges without violating State water conservation requirements to enable uses to be met;
- (3) Human caused conditions or sources of pollution prevent the attainment of the use and cannot be remedied or would cause more environmental damage to correct than to leave in place;
- (4) Dams, diversions or other types of hydrologic modifications preclude the attainment of the use, and it is not feasible to restore the water body to its original condition or to operate such modification in a way that would result in the attainment of the use;
- (5) Physical conditions related to the natural features of the water body, such as the lack of a proper substrate, cover, flow, depth, pools, riffles, and the like, unrelated to water quality, preclude attainment of aquatic life protection uses; or
- (6) Controls more stringent than those required by sections 301(b) and 306 of the Act would result in substantial and widespread economic and social impact.¹³

If a state believes that one or more of these scenarios may be present for a particular waterbody, then it should conduct a “use attainability analysis,” which EPA defines as “a structured scientific assessment of the factors affecting the attainment of the use which may include

physical, chemical, biological, and economic factors.”¹⁴ The results of the use attainability analysis may be applied to substantiate the removal or adjustment of a use, which would need to be vetted through a public rulemaking process and then submitted to EPA for review and approval.¹⁵

In defining the UAA mechanism, EPA imposed a bar against removing a designated use where: (a) it is an “existing use” (i.e., actually attained on or after November 28, 1975); or (b) may be attained by implementing technology- and water quality-based permit limits (for point sources) and cost-effective and reasonable best management practices (for nonpoint sources).¹⁶

Although the UAA mechanism is optional, states are required to revise their water quality standards whenever the standards specify uses less than those actually being attained.¹⁷ In other words, upgrades are mandatory, downgrades are optional.

In the past ten years, the water quality continuum has become considerably more complex. State assessment and listing decisions have resulted in the identification of approximately 60,000 impaired water segments. Court-ordered deadlines have forced states and/or EPA to develop TMDLs for those segments on an expedited basis. And growing sophistication in monitoring methods has made detection and quantification of pollutants achievable at exceedingly low levels. Each piece of the continuum has seen exponential progress. But the progress has not been synchronized.

In 2001, The National Research Council published a report styled, Assessing the TMDL Approach to Water Quality Management. Although the Council was tasked with assessing the scientific basis of EPA’s TMDL program, it focused considerable attention on the underlying WQS program. According to the Council:

Water quality standards are the benchmark for establishing whether a waterbody is impaired; if the standards are flawed (as many are), all subsequent steps in the TMDL process will be affected. Although there is a need to make designated use and criteria decisions on a waterbody and watershed-specific basis, most states have adopted highly general use designations commensurate with the federal statutory definitions. However, an appropriate water quality standard must be defined before a TMDL is developed. Within the framework of the Clean Water Act (CWA), there is an opportunity for such analysis, termed use attainability analysis (UAA).¹⁸

The Council examined various TMDL proceedings that were ineffective due to flaws in the underlying standards. For example, certain Louisiana TMDLs required zero discharge of pollutants but still could not attain the applicable standards. According to the Council, “a properly conducted UAA would have revealed the true problem -- naturally low dissolved oxygen concentrations -- before the time and money were spent to develop the TMDL.”¹⁹ The Council expressed concern over the “failure to widely employ UAA analysis,” pointing to “the absence of useful EPA guidelines” as a possible explanation. In its conclusions and recommendations, the Council urged EPA to issue new guidance on UAAs, addressing: (1) levels of detail required for UAAs for waters of different size and complexity; (2) broadened

socioeconomic evaluation and decision analysis guidelines for states to use during UAAs; and (3) the relative responsibilities and authorities of the states and EPA in making use designations for specific waters following a UAA analysis. The Council also recommended that UAAs “be considered for all waters before a TMDL plan is developed.”²⁰

EPA was slow to respond to the Council’s recommendations, but has recently issued several new guidance documents that are intended to promote the UAA process. Some states, including Virginia, have also answered the call with new laws and procedures. In 2006, the Virginia General Assembly amended the State Water Control Law to provide an explicit process for conducting use attainability analyses. The amendment reads as follows:

If an aggrieved party presents to the Board reasonable grounds indicating that the attainment of the designated use for a water is not feasible, then the Board, after public notice and at least 30 days provided for public comment, may allow the aggrieved party to conduct a use attainability analysis according to criteria established pursuant to the Clean Water Act and a schedule established by the Board. If applicable, the schedule shall also address whether TMDL development or implementation for the water should be delayed.²¹

RECENT BENTHIC TMDL EXPERIENCE

Virginia has developed several TMDLs for benthic impairments under the general standard, including Straight, Callahan and Russell Prater Creeks. In those proceedings, Virginia was stymied by several critical predicates to the actual TMDL. First, it had to determine the pollutants that were the most probable causes of the benthic impairment. Second, it had to establish in-stream pollutant targets, below which water quality conditions would actually achieve the general standard. Third, it had to identify the sources contributing pollutants above those targets. And finally, it had to equitably assign pollutant reductions to those sources (the sum of those reductions, coupled with a margin of safety, comprising the actual TMDL). Although each of these steps merited an independent public process, they were collapsed into one in order to meet Virginia’s consent decree deadlines. This limited the opportunity for informed public comment on the different decisional milestones in the process. By way of example, the decision on stressors led directly to the decision on targets, which in turn influenced the identification of sources and reductions. The public was given one opportunity to comment on this entire decision train. If, however, it had been given an opportunity to comment on each decision in series, then the public input may have influenced an entirely different outcome (e.g., one predicated on a different pollutant or an entirely different management tool, like a UAA).

In addition to these confounding predicates, Virginia was hampered by the limited scope of its analysis. To date, as a matter of practice, Virginia has only evaluated pollutant stressors during the TMDL process, even if non-pollutant stressors, like habitat modification, are apparent in the watershed. This practice makes it nearly impossible for Virginia to consider attainability issues as part of the process. For example, if stream channelization is a predominant non-pollutant stressor, which is exacerbated by overland sediment loading, then it may be appropriate to evaluate attainability before proceeding with a TMDL. However, under current practice, Virginia would proceed directly to a sediment TMDL, even if the resulting reductions from point

sources end up being more severe than necessary, or the eventual outcome of the TMDL remains inadequate to achieve the applicable WQS.

Despite its many hurdles, Virginia succeeded in developing TMDLs for Straight, Callahan and Russell Prater Creeks that were supported by local stakeholders and approved by EPA. These TMDLs are properly characterized as “phased TMDLs” under EPA parlance, in that they anticipate the need for additional data collection and analysis in order to validate or amend the underlying assumptions about stressor, target, sources and reductions.²²

Following the development of these TMDLs, Virginia also authorized the first-of-its-kind UAA for Straight Creek. This UAA, which is now underway, will provide a structured scientific platform for evaluating the impact of non-pollutant stressors, which was not considered during the TMDL process.

There are many reasons to consider these recent benthic TMDLs to be successes, among them, meeting relevant consent decree deadlines, obtaining EPA approval, overcoming stakeholder objections and thinking creatively to make progress in the face of uncertainty. However, more can be done to improve both process and results.

OPTIONS FOR IMPROVEMENT

The TMDLs for Straight, Callahan and Russell Prater Creeks compressed a number of critical decisions into one proceeding. As noted earlier, this was necessary in order to maintain pace with Virginia’s TMDL development schedule. However, Virginia was not compelled to jump into the TMDL process for these segments. Rather, given the data limitations and uncertainties associated with the underlying impairment listings, Virginia could have moved the segments from Category 5 to Category 3 or 4C in its integrated report. Doing so would have promoted a more step-wise process for gathering and assessing data to support further decision-making. Virginia still has this option for the group of waters currently listed in Category 5 for benthic impairment under the general standard.

Regardless of schedule or deadline, Virginia also needs to remain faithful to long-standing procedural safeguards. If, in the course of any particular proceeding, Virginia determines that a numeric water quality criterion is not available to address a particular stressor, then Virginia may not simply devise a numeric target on the spot. It must either use a translator procedure that has already been adopted as part of Virginia’s WQS, or amend the WQS to reflect a relevant numeric target. This, of course, cannot be accomplished on the spot, but it is precisely the procedure that Congress directed EPA and states to follow in the Clean Water Act.

Finally, Virginia should reconsider its practice of only evaluating pollutant stressors as part of the TMDL process. Nothing in the authorizing statutes or rules compels Virginia to ignore non-pollutant stressors. In fact, EPA’s UAA process is specifically designed to operate in concert with other water quality restoration efforts, like TMDLs, to ensure that those efforts are properly directed and effective. At a minimum, Virginia should do a screening analysis for attainability at the start of each benthic TMDL proceeding. The recent amendments to the State Water Control

Law are designed to facilitate this type of analysis, and, thereafter, to sequence further UAA efforts before, during or after TMDL development, as appropriate.

Virginia has a strong, collaborative and restoration-focused TMDL program. In the author's opinion, implementation of the options described above will make Virginia's program even stronger and more efficient.

Virginia TMDL Schedule, Consent Decree and Settlement Agreement in American Canoe Ass'n v. USEPA, June 21, 1999. See American Canoe Ass'n v. United States EPA, 54 F. Supp. 2d 621 (E.D. Va. 1999).

² Twenty of these are scheduled to be developed by May 2008, and another twenty-one by May 2010.

³ 9 VAC 25-260-20.A.

⁴ CWA § 303(a) and (c)(2)(A); 40 C.F.R. §§ 130.3, 131.2 and 131.3(i).

⁵ CWA § 303(c)(2)(A).

⁶ See 48 Fed. Reg. 51405 (November 8, 1983) (codifying EPA's water quality standards regulation).

⁷ CWA § 303(d)(1)(A).

⁸ 40 C.F.R. § 130.7(d).

⁹ CWA § 303(d)(1)(C).

¹⁰ 40 C.F.R. § 130.3(c).

¹¹ See 47 Fed. Reg. 49234 (October 29, 1982).

¹² See 48 Fed. Reg. 51400 (November 8, 1983) (codified at 40 C.F.R. § 131.10).

¹³ 40 C.F.R. § 131.10(g).

¹⁴ 40 C.F.R. § 131.3(g).

¹⁵ 40 C.F.R. § 131.10(e).

¹⁶ 40 C.F.R. § 131.10(h).

¹⁷ 40 C.F.R. § 131.10(i).

¹⁸ NRC Report at p. 90, accessible here: http://www.nap.edu/catalog/10146.html?onpi_newsdoc061501

¹⁹ NRC Report at p. 92.

²⁰ NRC Report at p. 94 (emphasis added).

²¹ Va. Code 62.1-44.19:7.E.

²² See Memorandum from Benita Best-Wong, Director, Assessment and Watershed Protection Division, EPA Office of Water, to Regional Water Division Directors, styled "Clarification Regarding 'Phased' Total Maximum Daily Loads" (August 2, 2006).

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A PLAN FOR CONDUCTING AN AQUATIC LIFE USE ATTAINABILITY ANALYSIS OF A CENTRAL APPALACHAIN URBANIZED STREAM

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KEY WORDS: use attainability analysis, designated use, water quality standards, benthic impairment

ABSTRACT

The U.S. Environmental Protection Agency (EPA) believes that it is important for States to have the right uses designated to each waterbody segment. States need to invest in putting in place more refined use designations along with differentiated criteria to protect those uses. A Use Attainability Analysis (UAA) is a structured scientific assessment of the factors affecting the attainment of the use which may include physical, chemical, biological, and economic factors. In July 2006 Virginia implemented legislation facilitating the conduct of a UAA. The Virginia Coalfields TMDL Group subsequently requested permission to conduct a UAA for the Aquatic Life Designated Use on Straight Creek in Lee County, Virginia. The Virginia State Water Control Board approved this request in March 2007. This paper presents the methods developed for the conduct of this UAA.

PREFACE

A Total Maximum Daily Load (TMDL) for Straight Creek (Lee County, VA) has been approved by Virginia Department of Environmental Quality (VA DEQ) and EPA. The Virginia Mining Issues Group (Group) proposed to conduct an aquatic life Use Attainability Analysis (UAA) to determine appropriate and achievable goals for Straight Creek. The Virginia State Water Control Board granted approval to proceed with development of a UAA in March 2007. This UAA approach was developed in cooperation with VA DEQ staff.

INTRODUCTION

A UAA is a structured scientific assessment of the factors affecting the attainment of the Designated Use. A designated use is defined as the appropriate water uses to be achieved and protected. Appropriate uses are identified by taking into consideration the use and value of the water body for public water supply, for protection of fish, shellfish, and wildlife, and for recreational, agricultural, industrial, and navigational purposes. States and Tribes examine the

suitability of a water body for the uses based on the physical, chemical, and biological characteristics of the water body, its geographical setting and scenic qualities, and economic considerations.

A UAA may include assessments of physical, chemical, biological, and economic factors. A UAA evaluates the reasons for use non-attainment, as well as provides a prescription for attaining the best use in a water body through remediation. The ultimate goal of any UAA is to determine the highest attainable use.

The UAA process integrates ecological data to arrive at a more thorough understanding of how the various forces in and around a particular stream interact. This UAA will use these data to develop a tool that will allow the biological condition of Straight Creek to be predicted based on other conditions in the watershed. The predictive tool will be used to determine the highest attainable aquatic life use based on the conditions expected and observed in Straight Creek after prescribed remediation.

Straight Creek is located in Lee County, Virginia and is a tributary of the Powell River/Upper Tennessee River system (Figure 1). Land uses in the watershed consist of forest, residential and mining. This watershed has a long history of timber harvesting, mining and residential influences.

Straight Creek's aquatic life was determined to be impaired (general standard, benthic macroinvertebrate) and it was included on the 1996 Virginia 303 (d) list of impaired waters. A Total Maximum Daily Load (TMDL) study identified sedimentation and Total Dissolved Solids (TDS) as the most probable stressors affecting benthic health.

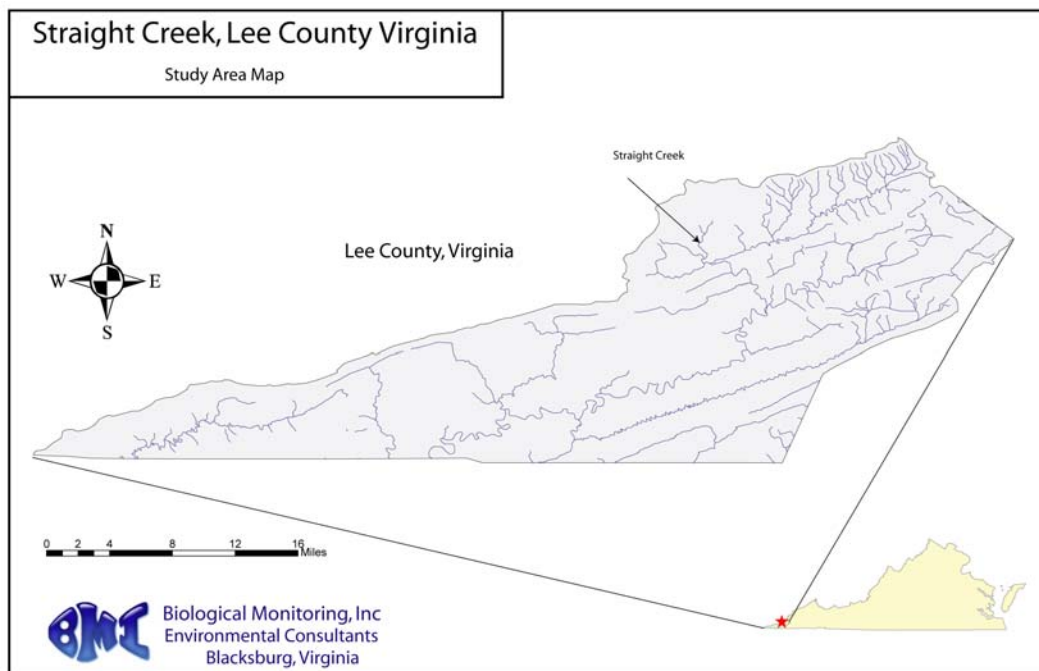


Figure 1. Location map.

The EPA *Technical Support Manual: Water Body Surveys and Assessments for Conducting Use Attainability Analyses* contains technical guidance to assist States in implementing Water Quality Standards Regulations (EPA, 1983). In addition, the EPA *Water Quality Standards Handbook* (EPA, 1994) provides background information and a framework for the conduct of a UAA. Over the past several years, the EPA has been conducting regional UAA workshops and now has a web site dedicated to the UAA process (<http://www.epa.gov/waterscience/standards/uaa/info.htm>).

The EPA recognizes that consideration of the suitability of a water body for attaining a given use is an integral part of the water quality standards review and revision process. EPA UAA guidance is intended to assist States in answering three central questions. These questions have been incorporated into this UAA study as the objectives. These questions are:

1. Is the Designated Aquatic Life Use an Existing Use?
2. What is preventing the Designated Aquatic Life Use Attainment?
3. What is the Highest Attainable Aquatic Life Use after remediation?

Methods for answering these three questions are presented herein.

METHODS

The first phase of this UAA will be to determine whether Straight Creek's designated aquatic life use is an existing use. An existing use is defined as a use actually attained in the water body on or after November 28, 1975, whether or not it is included in the water quality standards (40 CFR 131.3). Available information may be examined from historical records, ecological data, and scientific literature. The information gathered will be integrated so as to infer biological conditions on or after November 28, 1975.

The second phase of the study will involve development of a tool to predict biological condition based on stressors/pressures. This predictive tool will involve three distinct steps. These three steps include:

1. Determine stressors/pressures preventing use attainment;
2. Determine post-remediation stressor/pressure level;
3. Predict highest attainable aquatic life use based on steps 1 & 2.

The Straight Creek TMDL report identified two water quality parameters as the "most probable stressors" affecting impairment of the designated aquatic life use. This UAA will not be limited solely to water quality stressors. Other factors affect the assemblage of aquatic organisms in streams. Five such ecologic factors are discussed in EPA's Tiered Aquatic Life Use (TALU) documents. These five factors are:

- a) Water Quality
- b) Habitat Condition
- c) Flow Regime
- d) Energy Source
- e) Biotic Interaction

Step one of this tool, unlike theoretical models, will be based on actual collected data. The first task will be to collect data from outside Straight Creek, and develop a gradient of

stressors/pressures versus biological condition. Since a gradient of stressors/pressures versus biological condition must be developed, the data that are to be collected must represent a broad range of conditions. Data must be collected from streams having a broad range of both the stressors/pressures and a broad range of biological conditions.

Then, data from Straight Creek (and perhaps others) will be compared to these gradients so as to validate the tool. The tool needs to accurately predict the measured responses (stressors/pressures and biological condition). This approach is similar to the concept of a Biological Condition Gradient (BCG) and/or the Human Disturbance Gradient (HDG) of a TALU as described by Davies and Jackson (EPA, 2005).

The difference between the original and this UAA’s application of the BCG conceptual framework is in the selection of biological tiers. The original concept involves selection of biological attributes (narrative and numeric) for up to six tiers of biological condition (Figure 2). The tiers are determined by a consensus of experts such as regional biologists. Given the limited resources and narrow geographic scope of this UAA, it will instead rely on the narrative and numeric criteria for four tiers already developed and presented in the Virginia Stream Condition Index (VSCI) validation report (VA DEQ, 2006).

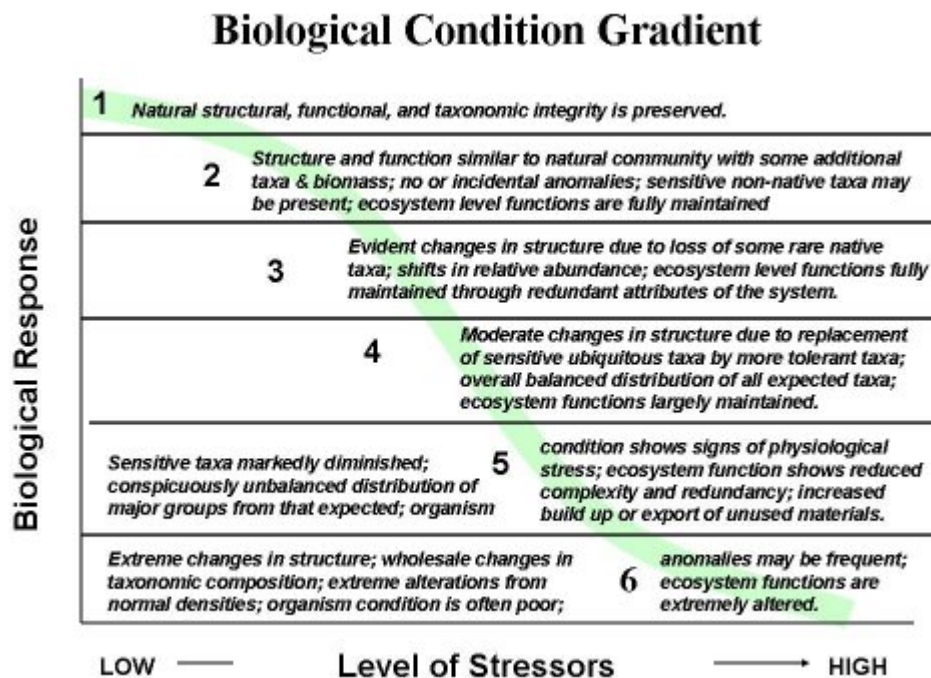


Figure 2. Conceptual biological condition gradient.

The predictive tool envisioned for this UAA will examine many streams ranging from very good to very bad biological and disturbance levels. This examination is necessary to develop a tool with application over a broad range of conditions. Next, each of the streams will be classified into one of the VSCI tiers. Then, those same streams will be classified into one of several tiers of stressor/pressure intensity. For example, one stressor gradient could be low, medium, and high RBP habitat quality. Finally, statistical analyses (e.g., discriminant analysis, etc.) will be

performed to determine whether stressor tiers can discriminate between VSCI tiers. The relationship could look something like Figure 3. This procedure would be repeated with each candidate stressor/pressure, creating multiple candidate relationships. Techniques such as the weight of evidence approach and EPA risk assessment methods may then be applied to determine the strongest and most appropriate relationships that will comprise the final predictive tool.

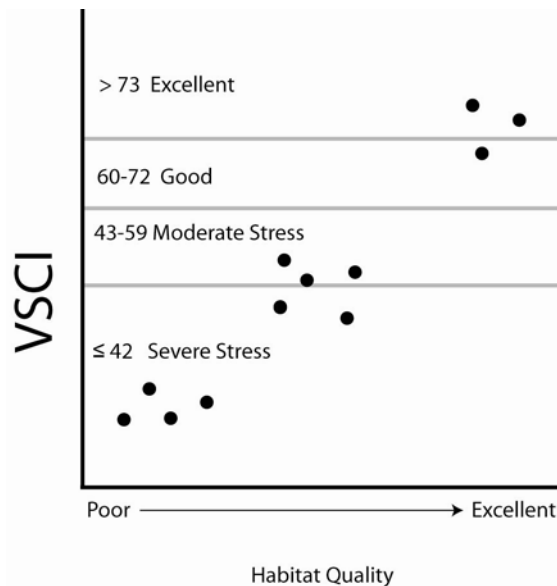


Figure 3. Conceptual biological condition gradient using VSCI.

The gradient of stressors/pressures and biological condition developed will be calibrated to Straight Creek. Calibration will ensure that the available stressor/pressure data for Straight Creek is predictive of its respective biological condition. Validation will use the same approach, but with a subset of Straight Creek data that is withheld from the development and calibration phase.

Step 1 of the predictive tool will be applied to Straight Creek to determine the stressors/pressures preventing aquatic life use attainment. The validated relationship from Step 1 will specify the stressors/pressures that are predictive of Straight Creek’s current biological condition. These stressors/pressures will be the factors preventing aquatic life use attainment.

Step 2 will involve development of a post-remediation model based on empirical data to predict the stressor/pressure level following remediation efforts. In order to determine feasible remediation efforts, the UAA study will be integrated (shared data, modeling, etc.) with the Straight Creek TMDL Implementation Plan (IP) and the discharge permitting process. Other concepts for determining feasible remediation efforts may include Risk Assessments and Environmental Impact Statement (EIS) framework (e.g., Alternatives Analysis).

The post-remediation model will be validated by monitoring the effects of the phased remediation efforts. In this manner, appropriate goals for improvements in Straight Creek and its tributaries may be achieved.

Once post-remediation watershed conditions have been estimated (Step 2), the highest attainable use can be determined (Step 3). The output of the post-remediation model (Step 2) will be applied to the relationship between stressors/pressures and biological condition (Step 1). This process will specify the highest attainable aquatic life use and the criteria to protect that use.

DISCUSSION

A key concept in assigning designated uses is "attainability," or the ability to achieve water quality goals under a given set of natural, human-caused, and economic conditions. Appropriate and defensible water quality standards are essential for achieving the Clean Water Act goals of maintaining and restoring water quality - and getting WQS right starts with getting designated uses right (EPA, 2006). The overall success of pollution control efforts depends on several factors, including a reliable set of underlying designated uses in water quality standards. Setting attainable water quality goals is important in stimulating action to improve water quality. Setting attainable uses advances actions to improve water quality.

Many waterbodies have had designated uses refined as a result of informative and compelling demonstrations provided by UAAs. A review of many UAA case studies reveals the breadth and variety of UAAs (EPA, 2006). In some cases, such as the one for Chesapeake Bay, the UAA is complex and resource-intensive. However, there are many effective UAAs that are much simpler, especially those with limited geographic scope. While many states have conducted UAAs, Virginia and EPA Region 3 have limited experience with them. However, Virginia DEQ has been proactive in their efforts to define the most appropriate designated use for Straight Creek.

A UAA is an investigative procedure to define the highest attainable use and criteria to protect the use. It is not a mechanism for use change. Rather it is simply a scientific tool that can be used to inform a separate, formal rulemaking process should a use change be deemed appropriate. Therefore, no presumption of use modification is made during the UAA process.

The findings of this UAA study could be used in support of one of several scenarios, four of which have been identified for Straight Creek. First, it is conceivable that the designated aquatic life use (DALU) and its criterion be retained. Alternatively, the DALU could be retained with a new site-specific criterion proposed. Third, a sub-category of the DALU and a new criterion could be proposed. Finally, although unlikely, a proposal to remove the entire DALU could be made.

To date, a formal UAA study plan been prepared cooperatively with VA DEQ and submitted for public comment. Once a final study plan has been negotiated, the study will proceed with continued collaboration with VA DEQ staff.

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Session III-C

Measuring Nutrients in Lakes

- Visualizing Water Quality with GIS
- Adapting Methods for Nitrogen Speciation Measurements Without Producing Toxic Waste
- Developing Nutrient Criteria for Lakes and Reservoirs: Virginia Impoundments
- Does Smith Mountain Lake and Claytor Lake Meet the New VDEQ Lake Criteria?

* * * * *

VISUALIZING WATER QUALITY WITH GIS

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KEY WORDS: water quality, Smith Mountain Lake, GIS, lakes

ABSTRACT

No lake is homogeneous. Therefore, representing water quality as a single average of readings from across the body of water does not give an accurate picture of the many varied processes occurring. Geographic Information Systems (GIS) can provide a means of visually representing the spatial variation of water quality. GIS can create maps which reflect the gradation present in a lake especially a reservoir which is more river-like in the upper reaches. It can also be used to indicate “hot spots” or areas of concern to both citizens as well as regulatory officials for use in policy formation and establishing best management practices. It is also possible to combine the visualization of the water quality with the land use patterns and/or soil erosion characteristics which is very useful in the development of a management plan to protect water quality. This presentation focuses on the use of GIS in representing nutrient characteristics collected from Smith Mountain Lake in a 20 year project between Ferrum College and the Smith Mountain Lake Association in conjunction with Appalachian Power Company and the Virginia Department of Environmental Quality. Sample maps including the variables of total phosphorus, chlorophyll-*a*, water clarity and *E. coli* populations from Smith Mountain Lake and the questions and challenges of the spatial representation of water quality data will be presented and discussed.

INTRODUCTION

For over twenty years, Ferrum College and the Smith Mountain Lake Association, in conjunction with Appalachian Power Company and the Virginia Department of Environmental Quality, have

collected data on the nutrient characteristics of Smith Mountain Lake. Using a combination of citizen monitors from the Smith Mountain Lake community and student interns from Ferrum College, water samples are collected from multiple locations on a weekly basis during the summer. These samples are then analyzed for total phosphorus levels, chlorophyll-*a* levels, water clarity and *E. coli* populations. To reflect the gradation present in the Smith Mountain Lake samples, maps generated by a Geographic Information System (GIS) were added to the annual report beginning in 2006. In addition, an executive summary including maps and initial descriptions of the data was produced in 2007. The summary provides preliminary results to the citizen monitors and the Smith Mountain Lake community prior to the more lengthy and detailed formal report.

The use of GIS in water quality monitoring is not new. Several states, including California and Idaho, as well as the EPA, are using GIS to collect, store and analyze water quality data (Kerney, Viers, et al, 1998; ESRI, 2007.) The 1972 Clean Water Act biennial reporting requirements' 2006 IRG includes guidelines for GIS frameworks (EPA, 2005). The use of GIS in the Smith Mountain Lake water quality project is in its initial stages and as such is currently focused on data representation and not more powerful spatial analysis.

METHODS

To create high-quality maps that serve as a vehicle of information there are five basic steps (Price, 2008):

1. Determine the objectives of the map
2. Decide on the data layers to be included
3. Plan a layout
4. Choose colors and symbols
5. Create the map

The objectives for the Smith Mountain Lake maps included highlighting “hot spots” or areas of concern to citizens and demonstrating the variation of the nutrient characteristics across the lake. Averages for the four variables at each of the sampling locations were used in the maps. GIS allows for sophisticated analysis of data sets, but for the initial use of mapping in the reports averages were deemed the most appropriate information to communicate. The layouts of the maps were kept consistent except for the bacteria map (Figure 1.) which was presented in a portrait rather than a landscape view in order to fit into the report format. The symbols, colors and scales were chosen very purposefully.

The bacteria map was created using simple point symbols. Each point on the map represents the bacteria count at a different sampling site. The other maps were generated using a technique called Inverse Distance Weighting (IDW). IDW takes the data for a given point and extrapolates a set distance from that location. For the total phosphorous, chlorophyll-*a* and secchi depths this method allows the viewer to gain a more complete picture of the lake's status. The bacteria readings from samples can vary dramatically from one location in close proximity to another in the lake given patterns of animal and human activity and thus the IDW is not an appropriate method for representing bacteria counts.

In the bacteria map, a color gradient of green to yellow to red was used to reflect the danger associated with higher levels of bacteria counts. The scale for this map follows guidelines established by the Virginia Department of Environmental Quality for levels of bacteria that constitute a health hazard. The chlorophyll-*a* levels (Figure 2) were represented with a green color gradient ranging from a light green for excellent levels to dark green for the worst levels. This reflects criteria established by Reckhow and Chapra (1983). The total phosphorus levels map (Figure 3) has a color ramp from yellow to green to blue representing excellent to worst levels as well and a scale also based upon Rackhow and Chapra. The water clarity map (Figure 4) reports the secchi depth readings from the various sampling locations. It has a color gradient that uses brown to symbolize the murkiness of the water where secchi disk readings were low and blues to indicate clearer water and higher secchi disk readings.

RESULTS

Each of the four maps is included below along with a description of the results. The bacteria populations, specifically *Escherichia coli* (*E. coli*), in Smith Mountain Lake in 2007 were the lowest ever measured at the lake. None of the sample dates and sites recorded had *E. coli* values that exceeded the Virginia Health Department standards as evidenced by the lack of red points on the map. The comparison of marinas and non marinas showed no significant difference in *E. coli* values in 2007, which may be the result of the Smith Mountain Lake boat pump-out service, the boaters and marinas' owners' vigilance and the lack of runoff into the lake in 2007.

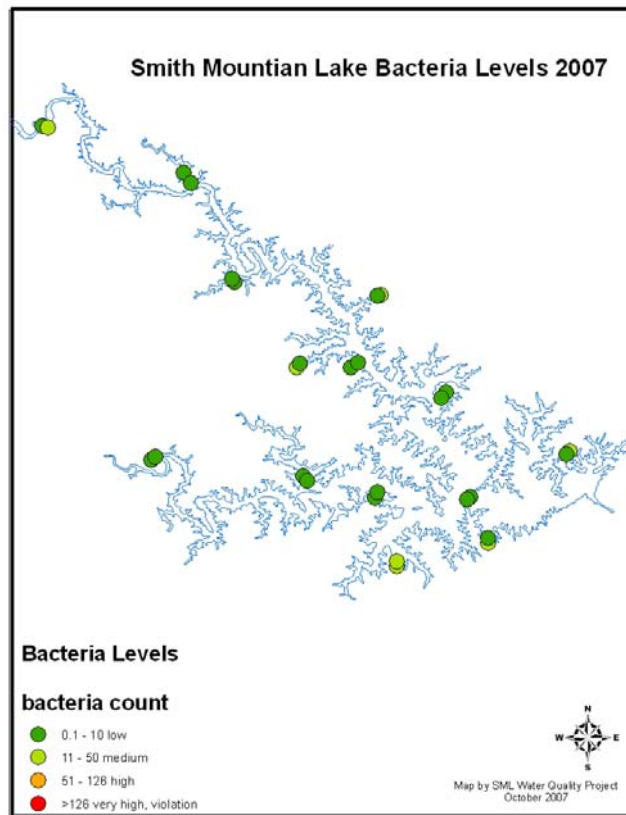


Figure 1. Smith Mountain Lake bacteria levels 2007.

Large portions of the lake have Chlorophyll-*a* levels less than 5 ppb as seen by the predominance of light green shading on the map. This means that on average there were fewer algae in Smith Mountain Lake in 2007 than there were in 2006. With less rainfall in 2007, fewer nutrients came into the lake from runoff resulting in less algal growth.

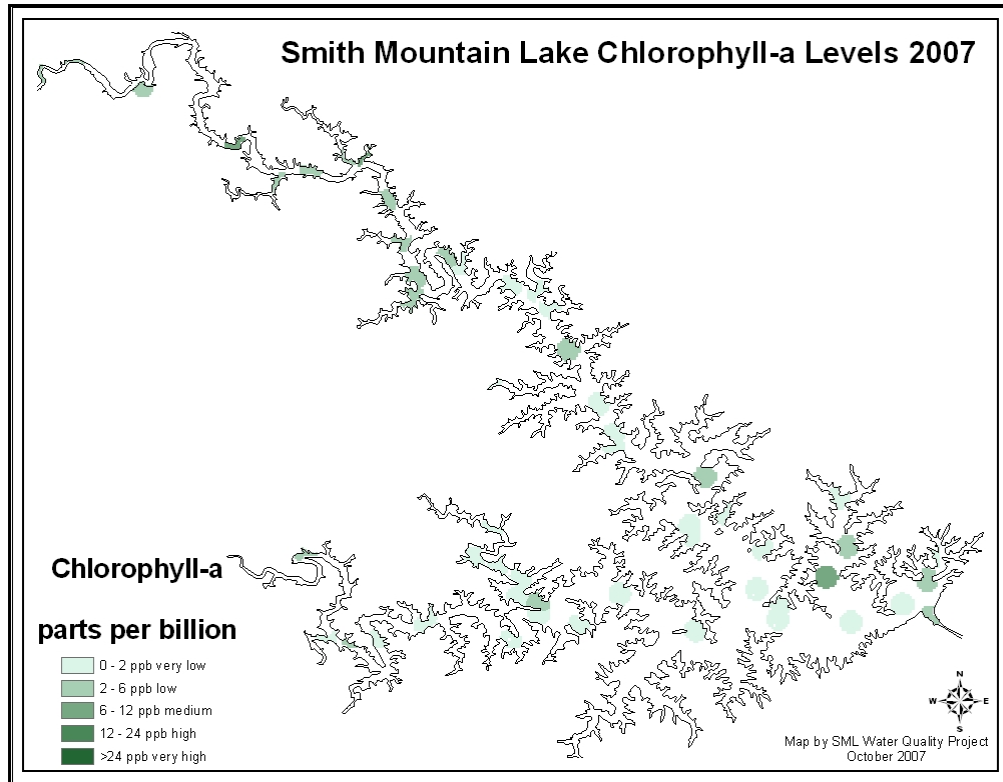


Figure 2. Smith Mountain Lake chlorophyll-a levels 2007.

Total phosphorus levels in the lake were in the Oligotrophic levels (yellows and light greens in the map) closer to the dam; while they were consistently in the Eutrophic range the further upstream. This is because nutrient-rich waters enter the lake from the Blackwater and Roanoke Rivers. As the water slows down, suspended clay particles settle into the lake bottom as sediment. Phosphorus is strongly associated with the clay particles and is deposited into the sediment. As a result, phosphorus levels close to the dam are lower than those in the rivers.

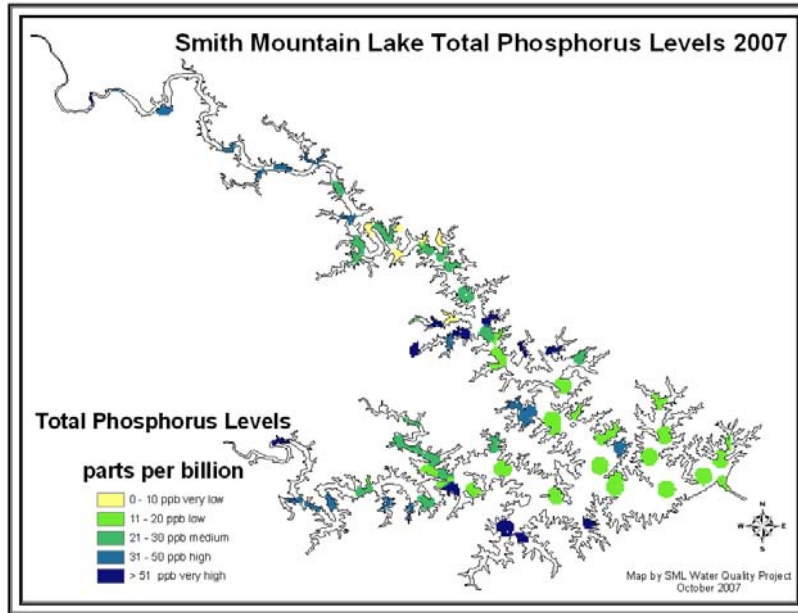


Figure 3. Smith Mountain Lake total phosphorus levels 2007.

Readings from summer 2007 clearly show in the map below the decrease in secchi depth as you move away from the dam. This is no surprise, since water enters the lake with a large amount of suspended solids and dissolved nutrients that encourage algal growth. More suspended solids and more algae means the clarity of the water is diminished and Secchi depths are smaller. Closer to the dam, the suspended solids have been deposited with the sediment along with nutrients, so there is less algae. Fewer suspended solids and less algae means clearer water and larger Secchi depths. This is readily apparent in Figure 4 as the browns are concentrated up river from the dam and in the tributaries.

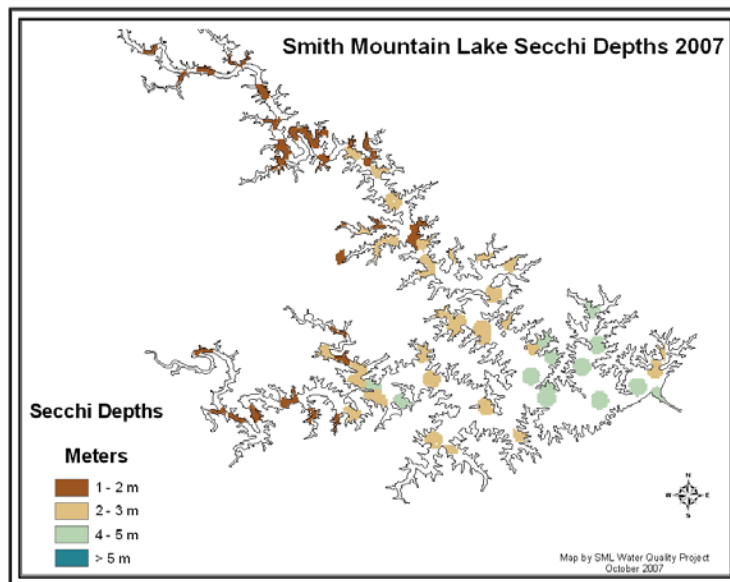


Figure 4. Smith Mountain Lake Secchi depths 2007.

CONCLUSIONS

The four maps above are able to convey at a glance the variation in nutrient characteristics in Smith Mountain Lake. The pattern of improving water quality closer to Smith Mountain Lake Dam is readily apparent as exhibited by all four nutrient characteristics. Initial response from the citizen monitors and the Smith Mountain Lake Association was very positive. The maps served as an effective vehicle for conveying very complex and crucial information to concerned citizens and public organizations.

ACKNOWLEDGEMENTS

The authors would like to acknowledge Doug Gillespie, Rachel Brown and Drew Howell for their work in cleaning the data, running the initial inverted distance weights and creating the draft maps.

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* * * * *

ADAPTING METHODS FOR NITROGEN SPECIATION MEASUREMENTS WITHOUT PRODUCING TOXIC WASTE

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ABSTRACT

The Ferrum College Water Quality Program stopped analyzing Smith Mountain Lake and Claytor Lake for total nitrogen several years ago. One important reason for this cessation was the production of toxic cadmium waste by using the cadmium reduction standard method for total nitrogen colorimetric analysis. However, nitrogen dynamics play an important role in the trophic status of a lake and revisions to the Virginia water quality regulations will include nitrate as a criteria pollutant for lakes.

Over the past two years, we have adapted and tested procedures for the measurement of total nitrogen speciation in these lakes. This adapted method enables the analysis of nitrate nitrogen, ammonium ion nitrogen, and organic nitrogen in freshwater. Fluorescence is used to measure ammonium ion concentration and second derivative UV spectroscopy is used to measure nitrate nitrogen and total nitrogen as nitrate (after persulfate digestion). Organic nitrogen is determined by mass balance.

The results of preliminary studies using these methods will be presented and an estimate of the precision and accuracy of the measurements will be discussed using preliminary data from 2007 in Smith Mountain Lake and Claytor Lake.

* * * * *

DEVELOPING NUTRIENT CRITERIA FOR LAKES AND RESERVOIRS: VIRGINIA IMPOUNDMENTS

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ABSTRACT

Under the Clean Water Act, criteria are components of water quality standards that are intended to protect water bodies' designated uses. The U.S. EPA has directed states and tribes to develop scientifically defensible criteria that will protect lakes and reservoirs from nutrient (N, P) overenrichment. Most of Virginia's lakes and reservoirs are constructed impoundments managed for multipurpose uses. Virginia's nutrient criteria are intended to protect these impoundments' capabilities to support aquatic life and fishery recreation. Recreational fish communities were considered as bioindicators of impoundments' capabilities to support these uses. Because impoundments' planktonic algae levels are influenced by water-column nutrient levels and have a more direct influence on higher-order fish species than do nutrients themselves, the criteria applied in the majority of the state's impoundments are expressed as near-surface chlorophyll-a levels. Alternative criteria, expressed as total phosphorous (TP), have been established for application in those impoundments that are treated with algicides. This presentation will describe the process and logic that were used in establishing Virginia's nutrient criteria for constructed impoundments.

* * * * *

DOES SMITH MOUNTAIN LAKE AND CLAYTOR LAKE MEET THE NEW VDEQ LAKE CRITERIA?

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KEY WORDS: lakes, nutrients, water quality regulations, phosphorus, chlorophyll-*a*

ABSTRACT

Virginia has recently developed “Criteria to Protect the Designated Uses of Lakes and Reservoirs from the Impacts of Nutrients” by proposed amendments to Water Quality Standards. The amendments propose to add new numerical and narrative criteria to protect designated uses of man-made lakes and reservoirs, such as dissolved oxygen concentration in the epilimnion during times of thermal stratification and nutrient criteria as controlled by section 9 VAC 25-260-187. Smith Mountain Lake and Claytor Lake have been two of the Virginia lakes designated as “nutrient enriched waters” but this designation may change when the new criteria are implemented. The trophic status of both of these lakes have been studied for many years (SML for 21 years and CL for 12 years) with the measurement of total phosphorus, chlorophyll-*a* and water clarity. Since 2005, depth profiles of dissolved oxygen and other chemical parameters have been measured on Smith Mountain Lake. These values will be compared to the new criteria being implemented in Virginia. A comparison over the years as to the change in the predicted trophic status of both lakes will be presented. The Carlson Trophic State Index for both lakes will also be compared to the potential change in nutrient enriched status of Smith Mountain Lake and Claytor Lake. The 2006 Trophic Status Index for Smith Mountain Lake which is a large pump storage lake, ranged from 38.2 – 62.9, while the Claytor Lake (a smaller run of the river reservoir) Trophic Status Range in 2006 was 51.5 – 58.7.

INTRODUCTION

The Smith Mountain Lake Volunteer Water Quality Monitoring Program was initiated in 1987 and has functioned each year since. The Smith Mountain Lake Association and scientists from Ferrum College cooperatively administer the program. The mission of the program is to monitor water quality in Smith Mountain Lake and to encourage active participation of the lake community in protecting the resource.

The trophic status of Smith Mountain Lake is monitored by measuring three parameters: total phosphorus as an indicator of nutrient enrichment, chlorophyll-*a* as an indicator of algal biomass, and Secchi depth as an indicator of water clarity. A new type of monitoring which is an indicator of trophic status was begun in 2005 and continued in 2006 and 2007 which measured depth profiles of dissolved oxygen, temperature, pH and conductivity.

The trophic status of a lake indicates the degree of nutrient enrichment and the resulting suitability of that lake for various uses. The process of eutrophication is nutrient enrichment of a body of water resulting in a significant increase in aquatic plant life (algae and aquatic plants). Phosphorus is most often the nutrient that limits algal production when concentration is low and

attempts have been made to relate the trophic status of a lake to concentration of phosphorus. In other words the concentration of phosphorus controls the algal population. (USEPA, 1974) One index used commonly is Carlson's Trophic State Index (TSI) which combines the three trophic status parameters mentioned above, total phosphorus, chlorophyll-*a* and Secchi depth into one index. The index ranges from 1-100 with the higher the index the more eutrophic the lake is classified, with 50 being approximately mesotrophic (Carlson, 1977).

Virginia has recently developed "Criteria to Protect the Designated Uses of Lakes and Reservoirs from the Impacts of Nutrients" by proposed amendments to Water Quality Standards. The amendments propose to add new numerical and narrative criteria to protect designated uses of man-made lakes and reservoirs, such as dissolved oxygen concentration in the epilimnion during times of thermal stratification and nutrient criteria as controlled by section 9 VAC 25-260-187. (DEQ, 2005). Smith Mountain Lake and Claytor Lake have been two of the Virginia lakes designated as "nutrient enriched waters" but this designation may change when the new criteria are implemented.

METHODS

Detailed descriptions of the methods of sample collection, preservation and analyses, and quality control/quality assurance procedures can be found in the VEE Report (Johnson and Thomas, 1990), in the *Volunteer Methods Manual* (Thomas and Johnson, 2006), and in the *Ferrum College Water Quality Procedures Manual* (Johnson, Thomas and Love, 2006). Samples are collected by volunteers every two weeks from the first week in June to the second week in August on Smith Mountain Lake and Claytor Lake.

The water quality parameters measured include water clarity (turbidity), measured as Secchi disk depth, total phosphorus, measured spectrophotometrically ($\lambda = 880 \text{ nm}$) after persulfate digestion; and chlorophyll-*a*, determined using the acetone extraction method and measured fluorometrically. (APHA, 1999). Quality control and quality assurance procedures evaluate sample collection and storage by the volunteers as well as laboratory procedures and are described in a later section in the yearly Smith Mountain Lake Reports. (Johnson & Thomas 1990, 1993, 1995, 1997, and 1999 & Thomas & Johnson 1992, 1994, 1996, 1998, 2000, 2001-2006)

Dissolved oxygen measurements were made on Smith Mountain Lake using an In-Situ Profiler™ (which includes a multi-sensor probe with 200 feet of cable), the parameters measured through a single profiling event are depth, temperature, dissolved oxygen concentration (DO), pH, and conductivity to the bottom (APHA, 1999). Measurements were made from a stationary boat data recorded on the Rugged Reader™ at five sites throughout Smith Mountain Lake six times from the first week in June to the second week in August.

RESULTS

The total phosphorus has gradually increased in Smith Mountain Lake for most of the 21 years measured, whereas chlorophyll-*a* and Secchi depth (a measure of water clarity) fluctuated over the years as presented for the last 5 years in Table 1.

Table 1. Smith Mountain Lake trophic status parameters from 2003-2007.

Year	Total Phosphorus	Chlorophyll- <i>a</i>	Secchi Depth
2007	34.5 ppb	4.3 ppb	2.3 m
2006	32.2 ppb	7.5 ppb	2.1 m
2005	28.9 ppb	6.2 ppb	2.1 m
2004	26.7 ppb	3.7 ppb	2.2 m
2003	54.4 ppb	8.7 ppb	1.9 m

The total phosphorus, chlorophyll-*a* and Secchi depth has fluctuated over the years in Claytor Lake for most of the 12 years measured as presented for the last 5 years in Table 2.

Table 2. Claytor Lake trophic status parameters 2002-2006.

Year	Total Phosphorus	Chlorophyll- <i>a</i>	Secchi Depth
2006	38.5 ppb	10.3 ppb	1.5 m
2005	36.9 ppb	9.0 ppb	1.6 m
2004	39.0 ppb	4.2 ppb	1.5 m
2003	51.2 ppb	10.3 ppb	1.3 m
2002	59.4 ppb	8.8 ppb	1.6 m

The Virginia Department of Environmental Quality (DEQ) has been studying the need for water quality criteria for lakes and reservoirs for a number of years and the following Table 3 is the recommended criteria based on eco-regions and water temperature designation.

The advantage to the Virginia Department of Environmental Quality is that the adoption of these criteria will continue to meet the phased obligations to USEPA of the Virginia's nutrient criteria development plan and to develop nutrient criteria appropriate for Virginia waters instead of USEPA and the commonwealth of Virginia accepting the default national criteria (DEQ, 2005). The other advantage to Virginia is that the adoption of these criteria will help protect the public water supplies and recreational lakes listed in these proposed amendments from the effects of nutrient enrichment.

Smith Mountain Lake and Claytor Lake are both considered "Cool water" reservoirs based on average water temperature. Smith Mountain Lake is considered in Eco-region 9 and Claytor Lake is considered in Eco-region 11. The Smith Mountain Lake criteria level would indicate a total phosphorus level of 30 ppb ($\mu\text{g/L}$) and a chlorophyll-*a* level of 25 ppb ($\mu\text{g/L}$) and Claytor Lake's criteria level would be a total phosphorus level of 20 ppb ($\mu\text{g/L}$) and a chlorophyll-*a* level of 25 ppb ($\mu\text{g/L}$) as listed in Table 3.

Table 3. Candidate criteria to accommodate fishery recreation and protect aquatic life, as recommended by Academic Advisory Committee (AAC) of the Department of Environmental Quality (DEQ) Division of Water Quality Programs' Ad Hoc Advisory Committee on Proposed Lake Nutrient Criteria January 2005 report and July 2005 Addendum One to the January Report. (Zipper et al, 2005).

Fishery Type	Warm-water	Cool-water	Cold-water (trout)	Managed / Fertilized	Warm-water	Cool-water	Cold-water (trout)	Managed / Fertilized
Eco-region	----- chl-a ($\mu\text{g/L}$) ^a -----				----- TP ($\mu\text{g/L}$) ^b -----			
11	35	25	10		40	20	10	
9	35	25		60	40	30		40
14	60	25			40	20		

^a Chl-a are 90th percentile values representative of the April – October period.

^b TP are the median values representative of the April – October period

If the Carlson's Trophic State Index (TSI_C) is considered the criteria to define the trophic status of Smith Mountain Lake and Claytor Lake then Smith Mountain Lake is considered mesotrophic in 2006 and Claytor Lake has a slightly higher trophic status and slightly more eutrophic than Smith Mountain Lake as the TSI-C values indicate in Table 4.

Table 4. Combined Trophic State Index (TSI-C) for Smith Mountain Lake and Claytor Lake, 2002 – 2006.

Smith Mountain Lake		Claytor Lake	
Year	TSI-C	Year	TSI-C
2006	50.3	2006	54.8
2005	48.9	2005	53.9
2004	46.4	2004	51.6
2003	53.6	2003	57.0
2002	46.2	2002	56.1

Another criterion suggested for lake water quality criteria is the dissolved oxygen levels in the epilimnion of the lakes. The epilimnion of Smith Mountain Lake based on a general accepted standard of that water column above the thermocline. Thermocline is the part of the water column that changes temperature degrees (Celsius) one degree or greater per meter of depth. According to the DEQ recommendation the Smith Mountain Lake epilimnion, which is in the "mountainous zones waters" (IV) should meet the criterion of a minimum of 4 ppm and a daily average of 5 ppm. At the two sites presented in the next Figures 1 and 2, PR19 and PM2 the thermocline is approximately at 3 meters on the August 14 sample date. The following figures indicate that the dissolved oxygen does not drop below 4 ppm until a lower depth. This

information indicates that Smith Mountain Lake meets this criterion on this date, late in the summer when stratification is most pronounced. Profile data are not available for Claytor Lake.

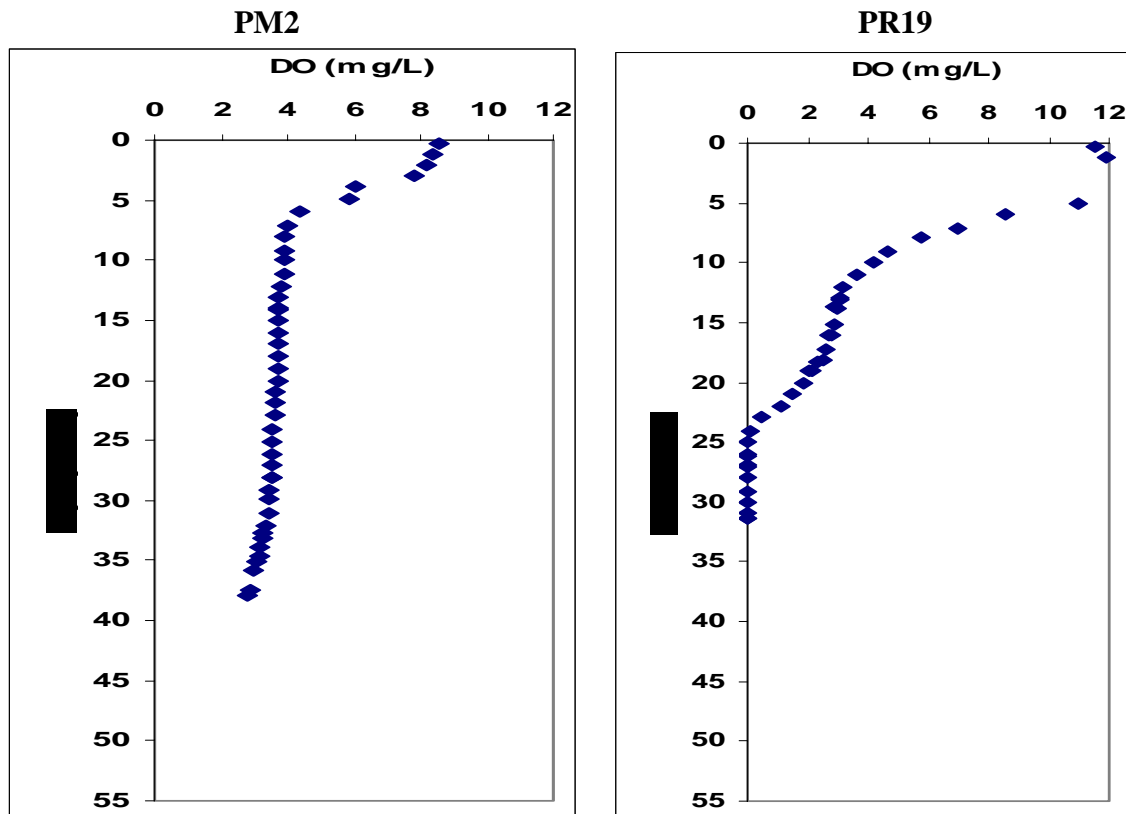


Figure 1 & 2. Smith Mountain Lake dissolved oxygen profiles on August 14, 2007 on the Roanoke River 19 miles from the dam (PR19) and at the main basin of Smith Mountain Lake 2 miles from the dam (PM2).

Other profile parameters recommended by VDEQ are pH with a range of 6-9 and temperature maximum allowable of 31 degrees Celsius. These two parameters were not evaluated in this paper.

DISCUSSION

In a comparison of the Smith Mountain Lake values to the recommended Virginia DEQ criteria for nutrients in cool waters and eco-region 9, Smith Mountain Lake exceeds the total phosphorus levels allowed in 2006 and 2007 and meets the criterion for chlorophyll-*a* for all years by a wide margin.

Claytor Lake values when compared to the Virginia DEQ criteria for cool waters in eco-region 11 exceeded the total phosphorus criterion all twelve years however the chlorophyll-*a* levels were lower than the criteria all years by a wide margin,

The Virginia Department of Environmental Quality recommends the following criteria for Smith Mountain Lake and Claytor Lake as shown in Table 5.

Table 5. Standard criteria recommended for Smith Mountain Lake and Claytor Lake by the Virginia Department of Environmental Quality. 2005.
SURFACE WATER STANDARDS WITH GENERAL, STATEWIDE APPLICATION-
PART I. 9 VAC 25-260-5.

Man-made Lake or Reservoir Name	Location	Chlorophyll <i>a</i> (µg/L)	Total Phosphorus (µg/L)
Claytor Lake	Pulaski County	25	20
Smith Mountain Lake	Bedford County	25	30

The other two criteria described in the results, trophic status index (TSI) and dissolved oxygen in the epilimnion also indicates that for both lakes the trophic status appears to be mesotrophic which generally is acceptable trophic status

CONCLUSIONS

As listed below the Virginia Department of Environmental Quality has recommended that Smith Mountain Lake and Claytor Lake be removed from the listed designation of “nutrient enriched waters” of Virginia. Apparently the chlorophyll-*a* criterion is more significant in designations than are the total phosphorus criterion.

“These two lakes, Smith Mountain Lake and Claytor Lake are repealed from the list of nutrient enriched water since the new method of controlling nutrients in these and other man-made lakes and reservoirs will be from implementation of the criteria set forth in 9 VAC 25-260-187. According to previous regulation these two lakes were classified as “nutrient enriched lakes” (9VAC 25-40 et seq.)” (DEQ, 2005)

The other descriptive characteristics of trophic status, trophic state index (TSI) and dissolved oxygen in the epilimnion also indicate the acceptable water quality of both Smith Mountain Lake for the TSI value and dissolved oxygen and Claytor Lake for the TSI value.

ACKNOWLEDGEMENTS

The colleagues who made this research possible were David M. Johnson, Jason Powell and Carol Love all from the Ferrum College Water Quality Program. The many students who have worked for the Water Quality program for 21 years have contributed greatly to the success of the Ferrum College Water Quality Program. Thanks go out to all of our volunteer monitors at Smith Mountain Lake and Claytor Lake who made this program possible for all the years of this program with their dedication and support. The Smith Mountain Lake Association (SMLA) provided political and financial support. We would like to acknowledge the support and time of Stan Smith, the liaison with the SMLA. The Friends of Claytor Lake (FOCL), Dean Jackson and Litt Furr have made the 12 years of research and sample collection at Claytor Lake possible.

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Session IV-A

Building Better Water Models

- WCMS – The Watershed Characterization and Modeling System: An Overview of Capabilities and Development
- Development of Enhanced Water Resources Datasets and Capabilities for West Virginia
- Cumulative Hydrologic Impact Assessments of Surface Coal Mining in West Virginia Using WCMS-HSPF
- Development of a WCMS Groundwater Modeling Capability for Coal Mine Hydrologic Impact Assessment

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WCMS – THE WATERSHED CHARACTERIZATION AND MODELING SYSTEM: AN OVERVIEW OF CAPABILITIES AND DEVELOPMENT

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ABSTRACT

The Natural Resource Analysis Center (NRAC) at WVU developed the Watershed Characterization and Modeling System (WCMS) to add advanced GIS tools and data to support watershed analysis. Since initial tools were developed in 1991, the major influence on WCMS development has been the primary issue affecting West Virginia watersheds – the cumulative impacts of coal mining. Support for WCMS development has included WVU resources and grants and contracts with state and federal agencies and non-profit organizations. WCMS was developed as an ESRI ArcInfo application in the early 1990's. Rewritten in ArcView, by 1999 WCMS was a tool used by permit writers in the WV Department of Environmental Protection (WVDEP) Divisions of Water and Waste Management and Mining and Reclamation. Interactions with WVDEP staff led to modifications and enhancements. WCMS is now an

ArcGIS extension that appears as a toolbar in ArcMap. Additional tools have been added to support hydrologic and water quality modeling through links to the Hydrological Simulation Program – FORTRAN (HSPF). Ongoing research seeks to model the impacts of underground mines, higher resolution spatial data, and advancements in NHD. Development continues. This paper provides an overview of WCMS, outlines steps in development, and discusses the application of WCMS to understanding the hydrology of relatively large watersheds. WVDEP staff currently use the WCMS for permitting activities. Permit writers use the software extensively to delineate watersheds and to get basic flow, loading, and area information for cumulative hydrologic impact assessments (CHIA).

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DEVELOPMENT OF ENHANCED WATER RESOURCES DATASETS AND CAPABILITIES FOR WEST VIRGINIA

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KEY WORDS: water resources, West Virginia, NHD, GIS

ABSTRACT

Several related efforts are currently underway in West Virginia to create comprehensive water resources datasets and associated capabilities for the state in Geographic Information System (GIS) format. The National Hydrography Dataset (NHD) has been completed at the 1:24,000 map scale for the entire state, and key enhancements have since been made to this dataset, allowing for increased utility. We have also completed stream segment-level watershed delineation for all stream segments in the 1:24,000 NHD in WV. The spatial framework provided by these datasets allows for detailed watershed modeling and characterization throughout the state, including quantification of: land use/land cover in the contributing watershed of any stream segment, cumulative watershed statistics, and distance to features of interest along streams, such as mines. These capabilities have many practical applications in cumulative impact analysis and ecological modeling. In addition to existing water resource datasets, we are also in the process of developing even more highly detailed maps of surface water resources at the local map scale of 1:4800 following the NHD data model. This local scale NHD is being developed from hydrologic features captured from detailed aerial photography, supplemented by recently completed statewide 3 meter Digital Elevation Models. Mapping streams at the local scale will allow more accurate capture of stream drainage alterations due to construction, mining, and other activities.

INTRODUCTION

Accurate, detailed digital spatial datasets of surface water features such as streams, rivers, lakes, ponds, etc. are extremely important in water resources management activities at the local, state, and regional levels. Mapped surface water features are used as important base map layers for

map making, as a means of tracking various datasets associated with stream or lake features (such as water quality monitoring sites), and in ecological and hydrological modeling efforts. For the past several years, the Natural Resource Analysis Center (NRAC) at West Virginia University has been highly involved in several inter-related efforts to coordinate water resources data development for the state of West Virginia, including the development of the high resolution NHD, the application of a statewide stream coding system to the high resolution NHD, the development of stream segment-level watersheds to correspond with the high resolution NHD, the development of additional tools and methods for modeling stream processes using these datasets, and finally the current development of more detailed local scale NHD data for the state.

The National Hydrography Dataset (NHD) is a comprehensive digital spatial dataset that maps and describes the nation's surface water features, including streams, rivers, lakes, swamps, and constructed waterways (USGS 2007). High resolution (1:24,000 map scale) NHD data are now available for the entire country, as well as the previously completed medium resolution (1:100,000 scale) datasets. The NHD data format was developed in order to combine previously existing USGS and EPA spatial datasets for surface water features. The common format of the NHD at various spatial scales is designed to encourage cooperation and exchange of data between users at the federal, state, and local levels. NHD data support a variety of applications, including map making, referencing existing datasets to locations within the NHD data, modeling water flow, and general hydrologic data maintenance.

Within the state of West Virginia, the high resolution NHD datasets for each of the state's 32 USGS sub-basins (major watersheds, or USGS 8-digit hydrologic cataloging units) were completed in cooperation with NRAC, USGS, USDA Forest Service, and the West Virginia Department of Environmental Protection (WVDEP). Since the completion of the high resolution NHD, several related efforts have been initiated. The purpose of this paper is to describe each of these related efforts in enhancing stream datasets in WV, and their utility to water resources management. Related articles are currently under review for publication, and further details may be obtained from the authors.

METHODS

Application of WV statewide stream coding system

The first enhancement to statewide stream datasets in West Virginia was the application of an existing statewide stream coding system to all streams mapped in the high resolution NHD for West Virginia. The WVDEP's Division of Water and Waste Management (as well as other state agencies) use an alternate, alphanumeric stream coding system to uniquely identify all streams in state records and databases. For example, the stream code "KC-26-B" refers to the 2nd tributary (B) to the 26th tributary (26) to the Coal River (KC) in West Virginia.

Several points differentiate these stream codes from the system already in place within the NHD data structure (for details on the NHD data structure, please refer to USGS 2007). These codes offer a unique reference ID for each stream as a whole, as opposed to the NHD stream coding which may or may not uniquely identify a stream as a whole. The system also applies a stream

name to every stream (even where NHD gives no name), and locates unnamed streams along mainstem named streams using a river mile based reference system. The coding system enables WVDEP to maintain a link between legacy datasets using the stream coding system and the spatial representation of each stream in the NHD. WV state stream codes were also applied to braided segments of larger rivers and streams, while the NHD does not attach names to these features.

Code transfers were largely automated, with several data integrity checks, such as verification that all NHD segments with the same NHD Reachcode received the same WV state stream code/name. If integrity checks were violated, this generally indicates an error in the NHD stream names (GNIS_NAME) or other attributes. All potential errors in the NHD are recorded for future reference as the NHD stewardship updating process gets underway in WV.

Development of stream segment-level watersheds

Stream segment-level watersheds are watershed drainage areas delineated for individual stream segments. Stream segments are defined as portions of stream linework between stream confluences or junctions. Each segment-level watershed area is labeled with stream segment identifiers and watershed labels, allowing users to reference any spatial dataset to an individual stream segment. Segment-level watersheds were delineated for each of the major river basins (8-digit cataloging hydrologic units) in WV (Figure 1), including the out of state portions of each drainage.

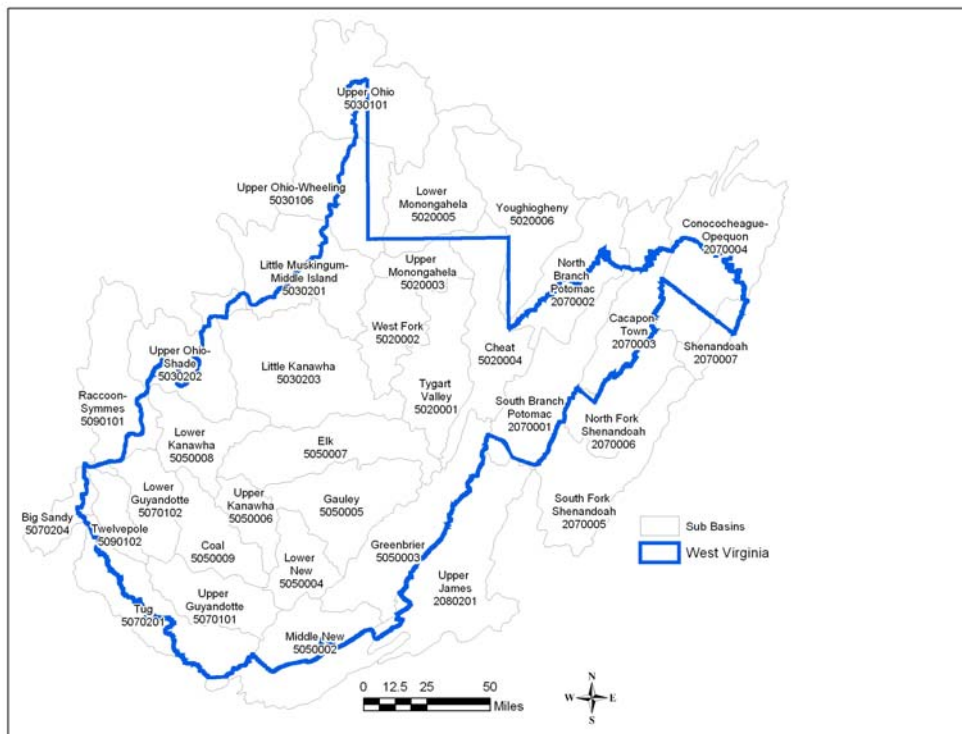


Figure 1. USGS sub-basins in West Virginia.

Segment-level watershed delineation required source datasets for stream linework and elevation. The stream linework was obtained from the 1:24,000 scale high resolution National

Hydrography Dataset (NHD) for all 8 digit hydrologic units in WV. Streams depicted in the NHD datasets are essentially all streams found in the 1:24,000 Digital Line Graph (DLG) hydrography dataset, ultimately based on the USGS topographic maps. Elevation data was obtained from the one arc-second (30 meter) National Elevation Dataset (NED) (USGS 2006). The elevation dataset was hydrologically corrected (for procedures, refer to Strager et al., 2007). Streams and stream centerlines from the NHD drainage route features were used to determine downstream pour points for each stream segment. Pour points were used in an automated routine to determine the drainage area for each individual segment. Some manual editing was required following delineation to produce a final statewide map of segment-level watersheds. Once watersheds were created, an automated procedure was followed to label or attribute each watershed with a unique code, as well as other 6, 8, and 11 digit watershed codes. Following delineation and attribution, a database table of watershed flow relationships was created in order to facilitate stream network-based and cumulative modeling applications.

Segment-level watersheds are now complete for the entire state at the 1:24,000 scale. We also completed 1:100,000 scale watersheds in the same manner, and a similar product called the NHDPlus is now available through a USGS contractor (Horizon Systems) at the 100,000 scale (Horizon Systems 2007).

Modeling stream processes with segment-level watersheds

Additional modeling capabilities have been developed that allow NRAC to use the segment-level watersheds in various modeling efforts designed to examine the influence of landscape factors on instream water quality and biota. We have developed tools that allow us to cumulative assess any spatially mapped features that occur in segment-level watersheds. For example, we have calculated cumulative area upstream for any stream segment, cumulative upstream area of any other mapped features (such as mining permits), and minimum distance upstream to any other mapped features of interest that may influence instream water quality (such as mines or discharges) (Figure 2). These tools are currently in use in the Cheat and Tygart river basins in WV to assess stream conditions as they relate to past and present mining activities throughout these river basins (Strager et al. 2007a).

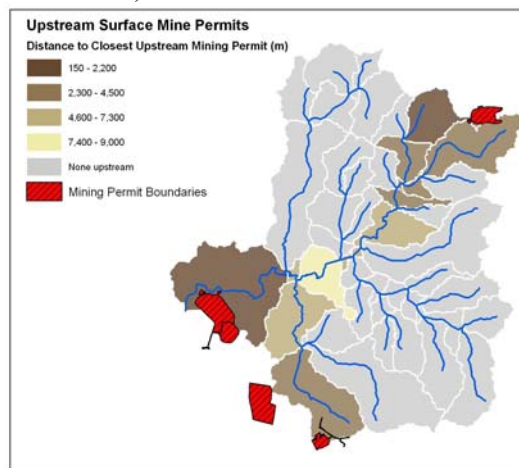


Figure 2. Example of result of cumulative modeling framework: Each segment level watershed is shown by the distance upstream to the nearest upstream mining permit polygon.

Local scale NHD data development

In addition to the high resolution (1:24,000 scale) NHD, efforts are currently underway to develop a local resolution (1:4800 scale) NHD in WV. The local NHD in West Virginia will provide a highly detailed cartographic base map of rivers, streams, lakes and ponds that will correspond with West Virginia's Statewide Addressing and Mapping Board (WVSAMB) aerial photography from the spring of 2003 (WVSAMB 2005). The local scale datasets are expected to be invaluable in efforts requiring detailed water feature maps, such as emergency and floodplain management, water quality permitting, and many other projects at the local or municipality level. The local NHD in West Virginia will provide additional detail on changes to stream drainage due to activities such as mining or construction that has not been previously captured in the high resolution NHD. For example, according to EPA estimates, up to 1,208 miles of streams (over 2% of all stream miles) in the multi-state mountaintop removal/valley fill study region (including large portions of southern West Virginia) have been lost or directly impacted by filling and mining activities (EPA 2003). These stream alterations are not mapped in the current best available stream dataset for West Virginia (high resolution NHD) but would potentially be identified and more accurately mapped by the local NHD.

The primary source of stream geometry for the local NHD is hydrologic features mapped using stereo pair orthophotos by the WVSAMB project contractors. WVSAMB contractors mapped all streams with visible water as either single-lined or double-lined streams with banks, depending on width. Visible lakes and ponds meeting minimum size requirements were also mapped, as were swamps and marshes. Intermittent stream features were not mapped, and hydrographic features were not named.

NRAC and the WV GIS Technical Center are currently converting WVSAMB stream linework into NHD format using techniques and procedures developed by USGS for two pilot sub-basins in West Virginia: the Gauley (05050005) and Upper Guyandotte (0507010).

RESULTS

Application of WV statewide stream coding system

Through the state stream coding process, we determined that about half of the streams in West Virginia (by length) do not have a recorded name. At the 1:24,000 map scale, there are 55,418 miles of mapped streams in WV. Of these, there are 8,761 unique stream names found. The remaining segments (n=43,839) are unnamed. By length, 28,267 miles (51%) of stream lines in the state had an official USGS stream name, while 27,151 miles (49%) of stream lines had no official USGS name. Official USGS stream names are those recorded in the Geographical Names Information System (GNIS) database.

Development of stream segment-level watersheds

A total of 166,000 segment-level watersheds were delineated for the area shown in Figure 1. The largest Segment-Level Watersheds are generally found in the Greenbrier River sub-basin

(05050003) due to the karst topography and sub-surface stream drainage. The average area of a segment-level watershed is 164.1 acres.

Local scale NHD data development

Local scale NHD data development is currently underway for two pilot watersheds. A comparison of stream mileage is shown in Table 1.

Table 1. Comparison of stream mileage between local and high resolution datasets for the Gauley River sub-basin.

Dataset	Miles	Segments
Gauley - Local resolution (1:4800)	4,714	36,411
Gauley - High resolution (1:24,000)	3,063	7,357

Overall we anticipate that the local scale NHD product will have greatly increased accuracy in stream feature geometry mapping, due to the improved and more up to date data sources used in its creation, as compared to the high resolution NHD (which was based on the USGS Digital Line Graph stream “blue lines” on the paper topographic maps). The local NHD will more accurately reflect recent changes in stream location and length, incorporating recent changes due to mining, road construction (particularly along interstate and highway corridors), and other forms of development (See example in Figure 3 of an area affected by construction of a mining related valley fill).

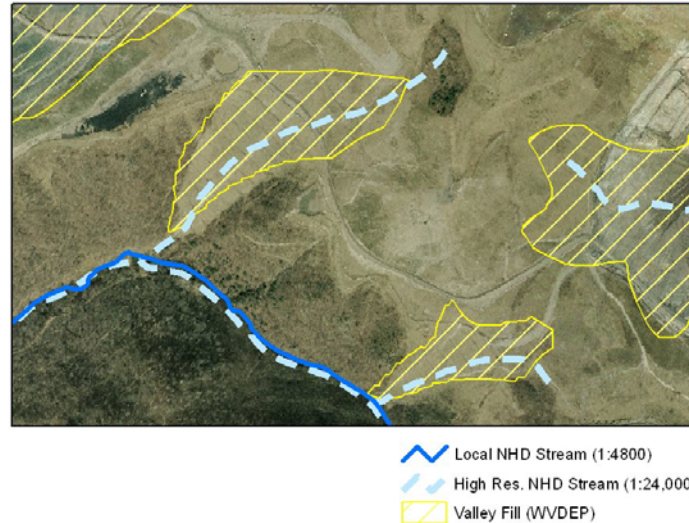


Figure 3. Comparison of stream feature geometry for local and high resolution NHD datasets for a portion of the Upper Guyandotte sub-basin affected by valley fill construction due to mining. WVSAMB aerial photo date is 2003.

ACKNOWLEDGMENTS

The WVDEP, USDA Forest Service, and USGS all participated in supporting the creation of the 1:24,000 NHD for the state of West Virginia. The stream coding and stream segment-level

watershed work was made possible with support and active participation and interest from the WVDEP. Dr. Todd Petty and Jen Barker-Fulton contributed to the development of applications of the stream segment-level watersheds to an ecological assessment of the Cheat and Tygart River basins in WV. NRAC is partnering with the WV GIS Technical Center and USGS in the local resolution NHD project (including Kurt Donaldson, Evan Fedorko, Kevin Kuhn, and Craig Neidig).

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CUMULATIVE HYDROLOGIC IMPACT ASSESSMENTS OF SURFACE COAL MINING IN WEST VIRGINIA USING WCMS-HSPF

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KEY WORDS: HSPF, coal mining, wcms, watershed modeling

ABSTRACT

A Cumulative Hydrologic Impact Assessment (CHIA) is required for all proposed coal mine permits. The Natural Resource Analysis Center (NRAC) at West Virginia University, with support from the West Virginia Department of Environmental Protection (WVDEP) and the U.S. Office of Surface Mining, Reclamation and Enforcement (OSMRE), has developed a suite of tools based on ERSI ArcGIS software to assist in this process. The watershed model Hydrologic Simulation Program-Fortran (HSPF) has been added to NRAC's Watershed Characterization and Modeling System (WCMS) to predict changes in water quality and quantity caused by surface mining. A joint calibration approach was adopted using historical stream flow records from five calibration watersheds and four additional verification watersheds throughout West Virginia. This resulted in one parameter set representative of the entire trend station region. A link between the NRCS Curve Number (CN) and HSPF parameters was developed based on an established empirical method. Segmentation of the study site watersheds is based on the 1:24,000 NHD stream maps. A table defining the flow connectivity between each segment-level watershed (SLW) was also created. The input control file for HSPF is automatically created using the SLW physical attributes and flow connectivity table. This results in a faster, more efficient method of creating HSPF input files than traditional raster-based methods.

INTRODUCTION

In cooperation with the West Virginia Department of Environmental Protection (WVDEP) and the U.S. Office of Surface Mining, West Virginia University's Natural Resource Analysis Center (NRAC) has developed a computer modeling system to assist in the cumulative hydrologic impact assessment (CHIA) of proposed surface coal mine sites. The system is an extension to ERSI's ArcGIS 9.2 software. The hydrologic assessments are required by the federal Surface Mining Control and Reclamation Act (SMCRA) and include the effects of mining on water quality and quantity. The WVDEP identified 235 trend station watersheds (TSW) throughout the coal mining region in West Virginia as potential study sites. These TSW's are defined by water quality sampling points at their outlets.

The Watershed Characterization and Modeling System (WCMS) is a suite of GIS-based tools developed at NRAC that has been used by WVDEP personnel for a number of years (Strager 2005). The tools extract specific watershed characteristics such as drainage area and flow path and can estimate mean discharge and 7Q10 low flows at any point on any stream within WV. The National Hydrography Dataset (NHD) has been completed at the 1:24,000 scale for the entire state along with the associated watersheds for each stream segment, or the segment-level watersheds (SLW). This dataset acts as the basic hydrologic modeling unit for WCMS-HSPF other analyses within WCMS.

The Hydrologic Simulation Program-Fortran (HSPF) is a complex, nonlinear, continuous model designed to simulate surface and subsurface water quantity and quality processes occurring in a watershed (Bicknell, et al, 2001). Its origins can be traced to the Stanford Watershed Model which was developed in the 1970's. Today, HSPF is supported by the EPA. The minimum inputs to the model are potential evapotranspiration and a precipitation time series. The precipitation data was supplied by Zedx Inc. in the form of hourly estimates between 1948-2002 on a 5 km spaced grid covering WV and portions of surrounding states. Because of its agency support and acceptance in the scientific community, HSPF was incorporated into WCMS to supplement CHIA analyses through the simulation of stream flow quality and quantity before and after proposed surface coal mining operations. A module to simulate mine acid drainage in-stream chemistry (AcidpH) was developed at NRAC and added to HSPF. The AcidpH chemical constituents are Aluminum, Conductivity, Iron, Manganese, Acidity, and Hydrogen Ion. In addition, several tools have been developed in WCMS to create the necessary pre and post-mine site input files for the hydrologic model. Several user interfaces have also been developed allowing the user to simulate mine site impacts in a series of simple, easy-to-use steps.

MODEL CALIBRATION

The first step in integrating the hydrologic model into WCMS was to calibrate HSPF to the trend station watersheds. Because of the large number of watersheds and limited availability of historic stream flow records necessary for calibration, a joint approach was adopted using five calibration watersheds and four additional verification watersheds throughout the trend station area (Figure 1).

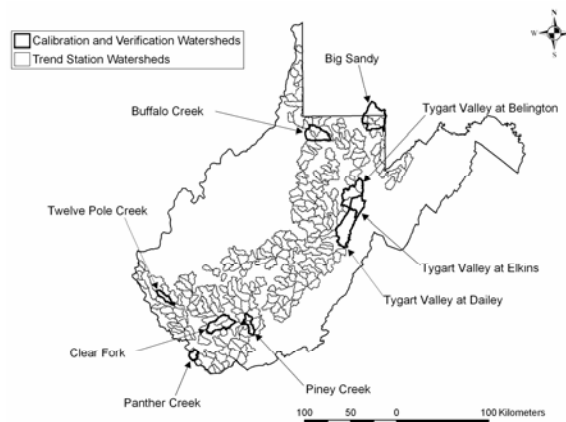


Figure 1. Trend station watersheds and calibration and verification watersheds.

Two software packages HSPEXP (1994) and PEST (Dougherty 2002) were used to determine the optimal HSPF parameter set over the trend station region for 11 land-use/cover categories. These categories are based on the 1993 GAP data (Strager and Yuill 2002) and include Barren, Pasture/Grassland, Row Crop/Agriculture, Shrubland, Urban/Developed, Wetland, Mined, Forest-Mild Slope, Forest-Moderate Slope, and Forest-Steep Slope land-use classes. The resultant mean error for the 10 year simulation period is less than 12% for calibration watersheds and less than 15 % for four of the verification watersheds. Since the NRCS Curve Number (CN) is typically the only piece of hydrologic information available for proposed mine sites, a link between the CN and HSPF parameters was developed based on an established empirical procedure (Lamont et al., In Press). A storage-outflow relationship for the proposed mine site retention pond was developed using information from the mine permit including study site drainage area, selected storm frequency, and the NPDES discharge.

WCMS-HSPF CHIA MODELING PROCEDURE

The first step in the WCMS-HSPF CHIA modeling procedure is to define the study area. This is accomplished by choosing the “Select Sheds” tool from the WCMS-HSPF toolbar and clicking on the segment-level watershed where the cumulative mine affects will be simulated (Figure 2).

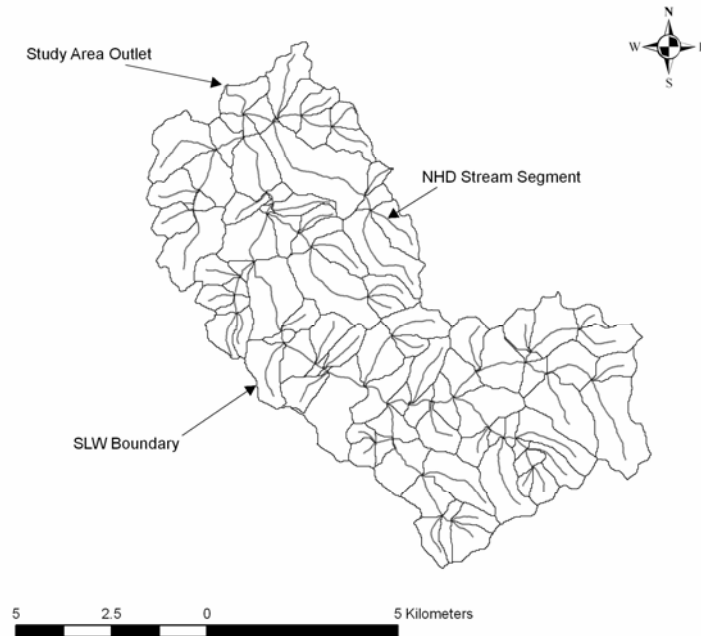


Figure 2. Example WCMS-HSPF pre-mine study area.

Each SLW in the study area layer contains several pre-calculated topographic and topologic attributes such as mean slope, cumulative drainage area, minimum and maximum elevation, the area of each land-use class, and flow connectivity variables, which are necessary for HSPF simulations. Because each SLW is processed individually by HSPF, the number of SLW's in the study area cannot exceed 500. An algorithm was developed at NRAC to aggregate the SLW's based on cumulative drainage area in the event of large study areas.

The second step is to add the proposed mine site polygon layer to the map and re-calculate the area of the land-use categories. This is done by selecting the “LULC” tool and designating the pre-mine study area layer, the pre-mine land-use/cover layer, the proposed mine site polygon, and by graphically selecting the SLW that will receive the mine site drainage.

The final step in the WCMS-HSPF CHIA modeling process is to input the proposed mine site permit information including the AcidpH constituent concentrations, drainage area, CN, storm frequency, NPDES outflow, and output report options in the user input form and run the HSPF simulation (Figure 3).

Figure 3. WCMS-HSPF user input form.

Once the simulation is complete, the pre and post-mine scenario comparison is displayed in an output report. This report can be customized to display information in graphic and tabular formats according to the user’s preference.

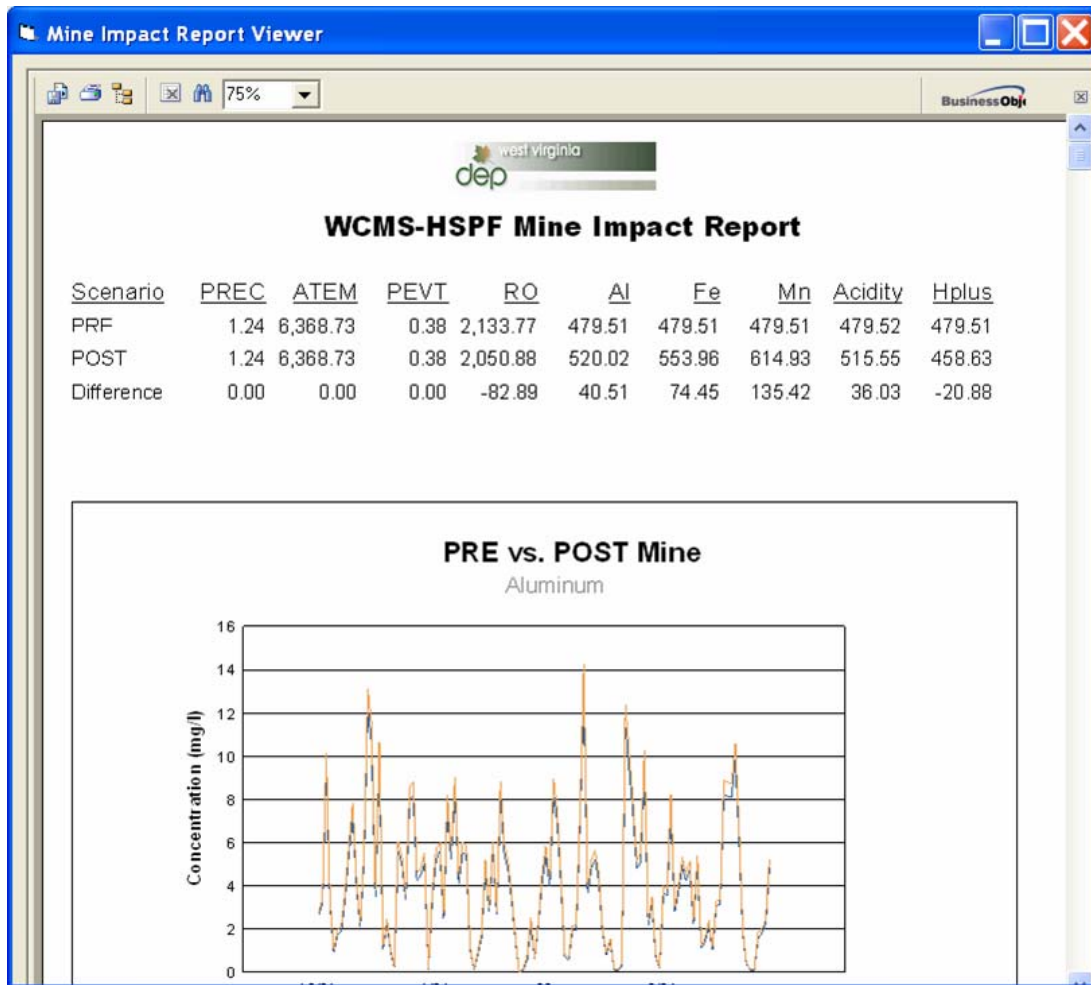


Figure 4. Output report display.

DISCUSSION

The WCMS-HSPF tool was developed specifically for WVDEP personnel to assist in the cumulative hydrologic impact assessments that are necessary for proposed surface coal mine permits. The model has been calibrated to a specific set of land-use/cover classes resulting in one parameter set for the entire trend station region of West Virginia. It is meant to be applied only as a theoretical comparative tool for pre and post-mine site scenarios in support of the CHIA process, not as a stand-alone modeling tool. However, through re-calibration, the model can be adapted to other land-use/cover layers as well as other geographical locations.

The use of the segment-level watersheds as a basic hydrologic modeling unit drastically decreases the run time of the model when compared to raster-based models. The majority of the necessary attributes can be pre-calculated for each SLW and stored in a table allowing the model to function almost entirely on vector-based operations. Following this approach, a SLW database can be developed that could potentially support other modeling applications.

This tool is currently in the development phase. Tasks to be completed include the calibration of the AcidpH module and the completion the CN-HSPF parameter relationship study.

ACKNOWLEDGMENTS

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DEVELOPMENT OF A WCMS GROUNDWATER MODELING CAPABILITY FOR COAL MINE HYDROLOGIC IMPACT ASSESSMENT

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KEY WORDS: groundwater, modeling, mining, WCMS, HSPF, RGRM

ABSTRACT

The impact of surface and underground mining on water resources is of continuing and growing concern. Specific issues include the effect of mining on aquifer and stream flow, inter-basin water transfers, acid mine drainage and other water quality issues. WCMS (Watershed Characterization and Modeling System) is a combination of software and data developed by the Natural Resource Analysis Center (NRAC) at West Virginia University (WVU). WCMS adds advanced geographic information systems tools to ESRI ArcGIS software. The EPA HSPF (Hydrologic Simulation Program – Fortran) watershed model, a recent addition to WCMS, will be joined by a new groundwater modeling tool named the Regional Groundwater Recharge Model (RGRM). RGRM will be calibrated jointly with HSPF using the recessionary portion of stream flow data. RGRM is designed to be a simplified 2-dimensional model that can be extended to a quasi-3-dimensional capability if needed. The groundwater flow domain subdivision is accomplished by Delaunay triangulation of the WCMS segment-level watershed (SLW) centroid point dataset. SLW's are 1:24,000 scale NHD (National Hydrography Dataset) stream network subwatersheds averaging approximately 160 acres in area. Delaunay triangulation automatically defines Voronoi polygons with sides that determine groundwater flow cross sections, yet allows variable resolution for mine site representation by the addition of local control points. The more capable USGS MODFLOW groundwater model can be imbedded within the regional structure of RGRM for high resolution modeling of underground mine impacts where desired.

INTRODUCTION

The Natural Resources Analysis Center (NRAC) Watershed Characterization and Modeling System (WCMS) is an analysis tool widely used within the West Virginia Department of Environmental Protection (WVDEP) for a variety of applications related to water resource management (Fletcher, et al., 2004). Recently, WCMS capabilities were expanded with development of a new toolbar to support the application of the U.S. Environmental Protection Agency HSPF (Hydrologic Simulation Program-Fortran, Bicknell, et al., 2001) watershed

hydrology and surface water quality model, which is a component of the EPA BASINS watershed management system (USEPA, 2001). WCMS provides specific GIS interface functionality that substantially automates the application of HSPF to mine-impacted watersheds, significantly reducing the complexity of watershed modeling analyses needed to evaluate new coal mine permit applications (Eli, et al., 2004, 2005). Additionally, a new model component has been developed to include the impacts of the proposed surface mine site hydrology, including the drainage plan design and NPDS (National Pollutant Discharge Elimination System) outflow points (Lamont, et al., 2005). Currently, WCMS-HSPF has been calibrated for use on 235 trend station watersheds within West Virginia by jointly calibrating five representative watersheds that share similar surface land use/vegetative cover, soils, topography, and geology with the trend station watersheds. Trend station watersheds have been defined by WVDEP at stream locations where regular water quality monitoring is conducted.

WCMS-HSPF models the surface rainfall-runoff components of the hydrologic cycle. The subsurface is not explicitly modeled in HSPF, and therefore, the geologic structure and hydraulic characteristics of the subsurface, including the impacts of underground mines on the local and regional groundwater system, are not included. Conversely, the USGS MODFLOW groundwater model (Harbaugh, 2005) has been used to model the impacts of underground mining on groundwater systems (Sahu, 2003; Capo, 2004). MODFLOW is universally accepted as a standard for groundwater modeling as is HSPF for surface water modeling. The idea of combining the two to form a complete water basin modeling system is not new. The Integrated Hydrologic Model (IHM) is a combined HSPF-MODFLOW water basin modeling system developed for use in Florida where shallow water table aquifers predominate and hydraulic gradients are low (Ross, et al., 2004). Although this latter integrated system is impractical for use in conjunction with WCMS in West Virginia due to differences in ground water regimes and data structures, joint use of HSPF and MODFLOW is a validated concept.

Development of a Regional Groundwater Recharge Model (RGRM) that is compatible with WCMS-HSPF and the underlying SLW (segment-level watershed) data structure of WCMS has been initiated to provide a more parsimonious solution to the groundwater modeling need in the coal mining areas of West Virginia. The development plan for the RGRM component supports the optional use MODFLOW as the primary underground mine site hydrologic model, imbedded within the much larger regional groundwater system modeled by RGRM. Since the RGRM component is to be run jointly with HSPF, and is also to be pre-calibrated to all trend station watersheds, it may be more accurately described as an extended HSPF capability rather than a separate groundwater modeling system. It will provide a broad-brush, approximate picture of the trend station watershed groundwater system opposed to that potentially provided by the more detailed modeling capabilities of MODFLOW.

RGRM CONCEPTUAL MODEL

The WCMS GIS database will be expanded to include new spatial attribute data required for support of RGRM. WCMS maintains a statewide database of 1:24,000 scale NHD stream segments (reaches), and the associated DEM (30 m resolution, re-sampled to 20 m). Other existing data layers include land use/cover, surface drainage information, and the SLW polygons associated with each NHD stream segment. The NHD stream networks and their associated

SLW's are illustrated in Figure 1 for the Twelve Pole Creek watershed. The SLW's are the highest resolution HRUs (Hydrologic Response Units) consistent with the NHD network, and are to be used as the primary level of spatial subdivision within the RGRM component.

Delaunay triangulation of the NHD SLW area centroids, as shown in Figure 2, establishes a network of connected points (nodes) from which Voronoi polygons (MATLAB, Mathworks, 2005) are constructed (Figure 3). The Delaunay triangulation establishes the spatial connectivity of the aquifer layer across the entire water basin being simulated via the triangulated irregular network (TIN) interconnections between nodes. The horizontal boundary of the control volume for maintaining the conservation of mass in each SLW aquifer layer corresponds to the actual boundary of that SLW and not the Voronoi polygon boundary. The Voronoi polygon sides are used to compute the directional flow cross section areas while the Delaunay triangle sides (links) are used to establish flow direction connectivity and to compute hydraulic gradients between adjacent SLWs. The connecting network of links are assumed to convey groundwater much in the same way as pipes conveying water in a pipe flow network. The pipe network analogy allows a quasi-2-dimensional model to be constructed using a one dimensional energy equation formulation, which significantly simplifies the solution for the SLW piezometric heads. It should be noted that the flow in the imaginary pipe network does not reflect the actual groundwater flow vector map. The groundwater flow vectors are generated in a post-processing step using the 2-dimensional piezometric head solution. Initially, a single layer model is to be developed and tested with the flow assumed to be predominantly horizontal. Therefore a single equivalent horizontal hydraulic conductivity value is used for each flow direction and the aquifer is assumed to be unconfined with horizontal flow behavior according to the Dupuit assumption (Bear and Verruijt, 1987)

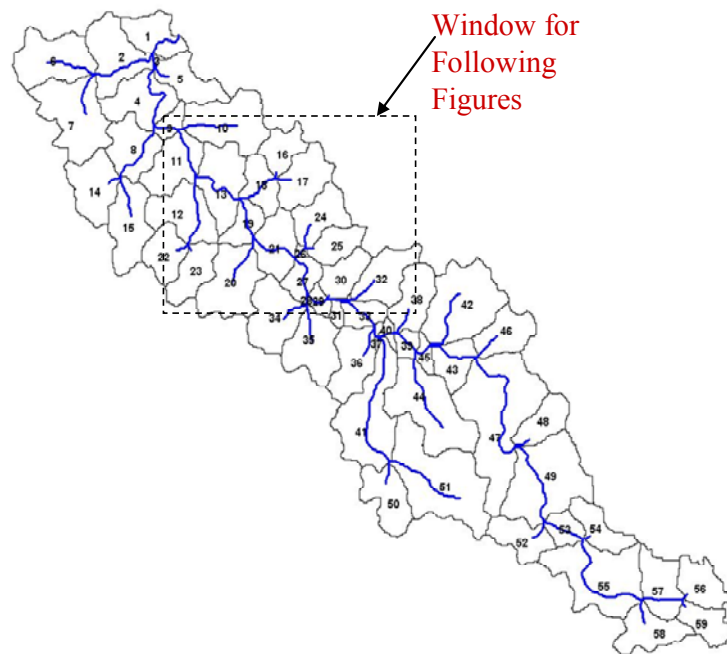


Figure 1. Twelve Pole Creek NHD stream network and associated WCMS segment-level watersheds.

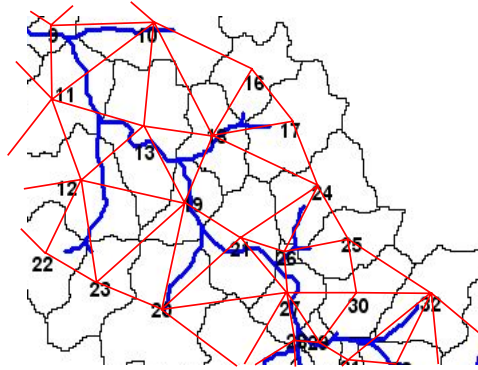


Figure 2. Delaunay triangulation using the SLW centroids.

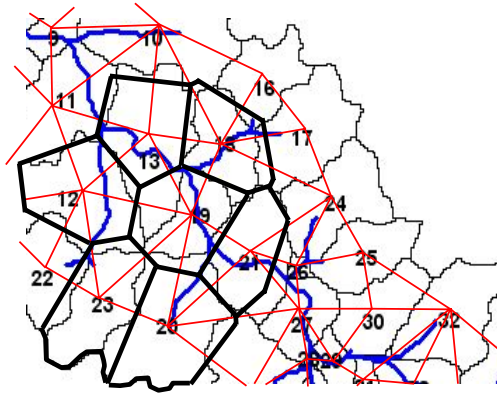


Figure 3. Voronoi polygons constructed surrounding segment-level watershed 19.

SINGLE AQUIFER LAYER FORMULATION

The continuity equation is written for each connecting node (SLW centroid) in the Delaunay triangulation (Equation 1).

$$\sum_j Q_{ij} - Q_i^* + I_i A_i^* = \frac{dS_i}{dt} = \frac{d(h_i A_i^* \eta_i)}{dt} = A_i^* \eta_i \frac{dh_i}{dt}$$

where:

$$Q_{ij} = \text{inflow to node } i \text{ from adjacent node } j \left[\frac{m^3}{d} \right]$$

$$Q_i^* = \text{outflow to SLW } i \text{ stream segment} \left[\frac{m^3}{d} \right]$$

$$I_i = \text{recharge rate in SLW } i \left[\frac{m}{d} \right]$$

$$A_i^* = \text{SLW } i \text{ planform area} \left[m^2 \right]$$

$$S_i = h_i A_i^* \eta_i = \text{SLW } i \text{ groundwater storage volume} \left[m^3 \right]$$

$$h_i = \text{aquifer saturated thickness in SLW } i \left[m \right]$$

$$\eta_i = \text{drainable porosity in SLW } i$$

$$t = \text{time} \left[d \right]$$

(1)

It should be noted that the summation sign convention in Equation 1 assumes that inflows to node i are positive and outflows are negative. Additionally, it is assumed that the drainable porosity η_i is equal to the specific yield as it is customarily defined in water table aquifers. The energy equation is written along each link (triangle side) of the Delaunay triangulation (Equation 2).

$$Q_{ij} = K_{ij} A_{ij} \frac{d\phi_{ij}}{dl_{ij}}$$

where:

$$Q_{ij} = \text{flow from node } j \text{ to node } i \left[\frac{m^3}{d} \right]$$

$$K_{ij} = \text{hydraulic conductivity along link } ij \left[\frac{m}{d} \right] \quad (2)$$

$$A_{ij} = \text{flow cross section area along link } ij \left[m^2 \right]$$

$$\phi_{ij} = \text{piezometric level along link } ij \left[m \right]$$

$$\frac{d\phi_{ij}}{dl_{ij}} = \text{piezometric gradient from node } i \text{ to node } j \text{ along link } ij$$

$$l_{ij} = \text{link } ij \text{ length } \left[m \right]$$

The sign convention for the direction of flow Q_{ij} remains the same as in Equation 1. The flow cross section area A_{ij} is computed as a product of the Voronoi polygon side length L_{ij} bisecting link ij and the mean aquifer saturated thickness $(h_i + h_j)/2$ at that point. A special form of the energy equation is used to compute the outflow from SLW i to its stream segment Q_i^* (Equation 3).

$$Q_i^* = K_i^* A_i^* \frac{d\phi_i}{dl_i^*} \approx K_i^* \sqrt{A_i^*} (\phi_i - \phi_i^*)$$

where:

$$\frac{d\phi_i}{dl_i^*} \text{ is approximated by } \frac{(\phi_i^* - \phi_i)}{\sqrt{A_i^*}}$$

and where

$$\phi_i^* = \text{elevation of the outflow end of stream segment } i \left[m \right] \quad (3)$$

$$\phi_i = \text{piezometric level in SLW } i \left[m \right]$$

$$K_i^* = \text{the effective outflow hydraulic conductivity within SLW } i \left[\frac{m}{d} \right]$$

$$\sqrt{A_i^*} = \text{the effective flow length from SLW } i \text{ to its outlet point } \left[m \right]$$

the remaining variables are defined as above

A forward in time finite difference equation can now be written for Equation 1 for each node i in the network (Equation 4).

$$\frac{\left(\sum_j Q_{ij} - Q_i^* + I_i A_i^*\right)^t + \left(\sum_j Q_{ij} - Q_i^* + I_i A_i^*\right)^{t+\Delta t}}{2} = A_i^* \eta_i \frac{(h_i)^{t+\Delta t} - (h_i)^t}{\Delta t} \quad (4)$$

Equation 2 can also be expressed in finite difference form along each link ij in the network (Equation 5).

$$Q_{ij} = K_{ij} A_{ij} \frac{(\phi_j - \phi_i)}{l_{ij}}$$

where:

$$A_{ij} \approx \frac{(h_i + h_j)^t}{2} L_{ij} \quad (5)$$

L_{ij} = Voronoi polygon side length bisecting link ij [m]

In Equation 5 it is important to note that A_{ij} is defined at time step t to avoid creating a nonlinear finite difference equation in the piezometric head ϕ . Substituting Equation 5 into Equation 4, and assuming that zero elevation datum corresponds to the aquifer bottom (then $h = \phi$), Equation 6 results.

$$\left[-\sum_j \left(\frac{K_{ij} (A_{ij})^t}{2l_{ij}} \right) - \frac{K_i^* \sqrt{A_i^*}}{2} - \frac{A_i^* \eta_i}{\Delta t} \right] (\phi_i)^{t+\Delta t} + \sum_j \left(\frac{K_{ij} (A_{ij})^t (\phi_j)^{t+\Delta t}}{2l_{ij}} \right) =$$

$$\left[\sum_j \left(\frac{K_{ij} (A_{ij})^t}{2l_{ij}} \right) + \frac{K_i^* \sqrt{A_i^*}}{2} - \frac{A_i^* \eta_i}{\Delta t} \right] (\phi_i)^t - \sum_j \left(\frac{K_{ij} (A_{ij})^t (\phi_j)^t}{2l_{ij}} \right) - K_i^* \sqrt{A_i^*} \phi_i^* - \frac{(I_i^t + I_i^{t+\Delta t})}{2} A_i^* \quad (6)$$

The piezometric heads ϕ at time step $t + \Delta t$ on the left hand side of Equation 6 are to be calculated as a function of the known quantities on the right side. Equation 6 is written for each node in the network yielding N equations in N unknown piezometric heads. Solution of the system of N equations is accomplished using matrix methods at each time step in the simulation. Following determination of the piezometric heads, connecting link flows and node outflows (to the stream segments) can be computed using Equations 3 and 5 for each time step.

CONCLUSIONS AND FUTURE DIRECTIONS

The HSPF model is continuous simulation watershed model that includes all of the principal components of surface and subsurface hydrologic processes (Bicknell, et al, 2001). It focuses on the surface and soil water processes, but includes a simple groundwater component to permit closure of the surface water hydrologic balance via outflow to the stream reach, or by allocation to inactive groundwater storage. The groundwater component is a simple reservoir-recessionary flow model that is associated with each land use – land cover category for which a single HSPF PERLND (pervious land segment) component is defined. Therefore, the HSPF groundwater component operates independently within each individual PERLND and does not reflect any

connectivity with the other groundwater components in adjacent PERLND's, and bears no resemblance to the reality of spatial interconnectivity of groundwater systems found in a typical drainage basin. RGRM is intended to correct this shortcoming within HSPF. The inflow to the groundwater reservoir in HSPF can be output as a time series for the simulation period and applied as input to the RGRM model as the recharge rate I_i . After running RGRM for the same simulation time period, the corrected groundwater outflow time series then replaces that generated by HSPF. Some limited recalibration of HSPF may be required to adjust the recharge rate to reflect the correct mass balance for the simulated watershed. This correction should better reflect the spatial distribution of dry weather recessionary stream flows within the watershed stream network.

Currently, the RGRM single aquifer layer modeling concept presented above is being developed and tested. RGRM's principal task is to improve simulation of spatially and temporally distributed groundwater recharge rates, outflows to streams, and interception of groundwater by existing and proposed underground coal mines. This capability is intended to support the assessment of mining impacts on the groundwater and surface water mass balance within the affected drainage basin. RGRM is being designed to be strongly mass conservative, and is intended to be calibrated jointly with HSPF by comparing the groundwater component supplied to streamflow to that obtained from stream gage data (or available statistical low flow data). The primary hydraulic parameters to be used in calibration consist of aquifer hydraulic conductivity and specific yield. Future work includes the addition of a multiple aquifer layer option and the development of user interface tools for addition of mine cavities and fracture flow paths for simulation of mine subsidence impacts on stream flow.

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Session IV-B

Heavy Metals: An Ecological Conundrum

- Ulcerative Shell Disease (USD) and Its Possible Relationship to the Bioaccumulation of Lead (Pb) in Aquatic Turtles in an Urban Lake
- Fish Tissue Mercury Status and Trends for North Fork Holston River
- Comparative Ultrasound Assisted Leaching of Selenium and Arsenic from Coal Mining Associated Rocks and Sediments

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ULCERATIVE SHELL DISEASE (USD) AND ITS POSSIBLE RELATIONSHIP TO THE BIOACCUMULATION OF LEAD (Pb) IN AQUATIC TURTLES IN AN URBAN LAKE

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KEY WORDS: bioaccumulation, lead, turtles, shell disease

ABSTRACT

Turtles are a visible component of the biodiversity urban lakes which receive large quantities of terrestrial runoff that includes heavy metals. Turtles have been determined to accumulate heavy metals, but little is known regarding the effects on the health of turtles. Bioaccumulation of lead has been implicated in the suppression of the immune system and impairment of calcium uptake and skeletal calcification in other vertebrates. This study surveys the occurrence and severity of Ulcerative Shell Disease (USD) of aquatic turtles in an urban lake and investigates whether bioaccumulation of lead (Pb) is related to its occurrence. USD is a chronic condition of the bone and scutes of turtles, affecting both the carapace and plastron. *Trachemys scripta* (Sliders) and *Psuedemys rubriventris* (Cooters) were captured over a two-year period. Incidence and severity of USD were calculated, and small sections of shell were removed for lead analysis. The occurrence of USD was high in both species and higher in females than males. Cooters had significantly higher lead concentrations than Sliders. Levels were below reported no observable effects level (NOEL) for Sliders, yet above levels shown to interrupt some physiological activities in other species. Linear regression revealed no positive relationships between lead concentration and USD severity for either species. We conclude that there is not a direct relationship between lead concentrations and USD in these turtles.

INTRODUCTION

Ulcerative Shell Disease (USD) is a contagious and chronic disease afflicting both the carapace and plastron (Wallach 1975). While this disease does not usually prove to be directly lethal to the infected turtle, secondary infections may lead to mortality. A specific pathogen responsible for USD has not been determined; pathogenic infection may be manifested due to impairment of the immune system, with toxic or immunosuppressive pollutants providing an opening for opportunistic, pathologic agents (Lovich *et al.* 1996; Garner *et al.* 1997).

Metals are the second most common pollutant for lakes, ponds and reservoirs (US EPA 2000). In urban areas, the most significant contributor to the deposition of heavy metals is runoff (Pitt 1995). Lead, which tends to accumulate in soils and sediment, can remain in the environment for a long period of time (Davies 1994, Menzer 1991). In lakes, concentrations of aqueous Pb are positively correlated with high DOC, high total nitrogen, shallow secchi disk depths, minimal oxygen concentrations and nutrient enrichment (Chen *et al.* 2000). In recent studies of Lake Maury, an urban lake in southeastern Virginia and the location for this study, lead levels in sediments ranged from 22.3 - 119.2 ppm (Brown 2005), while the concentration of lead in the water was less than 5 ppb (Wainwright 2007 personal communication).

Bioaccumulation of lead brings about a multitude of effects in human and animal skeletal systems (Pounds *et al.* 1991). Bioaccumulation of Pb has been implicated in the suppression of the immune system (Borošková *et al.* 1994, Krocova *et al.* 2000). Effects also include disruption of bone development, formation and resorption, as well as interference with the function of bone cell function hormones. Additionally, accumulation of lead is associated with decreases in bone mineral density and bone volume, as well as interference with osteoblast-like cell activity (Gruber *et al.* 1997, Schirmacher *et al.* 1998, Wiemann *et al.* 1999, Bagchi and Preuss 2005).

Studies with *Chelydra serpentina*, *Trachemys scripta* and *Terrapene carolina* have shown these species accumulate significant levels of lead (Pb) (Beresford *et al.* 1981, Overmann and Krajicek 1995, Golet and Haines 2000, Nagle *et al.* 2001). Lead accumulates in the calcareous tissue of reptiles, including turtle shell; the large mass of calcified tissue in the carapace may serve as a sink for Pb, perhaps reducing the amount accumulating in soft tissues. (Hinton and Scott 1990 Overmann and Krajicek 1995) In Slider hatchlings, lead levels of 100 ppm has been shown negatively to affect survival, growth, and righting behavior (Burger *et al.* 1998).

This study will focus on the genera *Trachemys* (Sliders) and *Pseudemys* (Cooters) within the family Emydidae. Slider and Cooter species are medium to large-sized turtles distributed from New England to Brazil, and into the West Indies (Carr 1995). These species are typically pond inhabitants, but are also found in small streams and along lake edges. Slider turtles are habitat generalists including those contaminated with radioactive wastes and chemical pollutants (Gibbons 1990).

METHODS

Lake Maury in southeastern Virginia is a 68 ha urban lake in the Mariners' Museum Park, located inside the city limits of Newport News, VA. The watershed is comprised of park woodlands and recreational fields, residential and commercial zones, and major thoroughfares. The lake also serves as a stormwater detention basin.

Turtles were captured using dip nets and baited hoop nets. Captured turtles were weighed, measured, sexed, and morphological data recorded. Severity of USD was determined by calculating the percent of damage to the plastron. Plastrons of all specimens were diagrammed on a standard size plastron outline, placing a grid overlay onto the diagram, counting the squares containing evidence of disease, and calculating the percentage of damaged area. Shell samples were collected by cutting a small triangle of shell from the left and right, fourth or fifth marginal pleurals with a small hacksaw. Samples were placed into individual clean and sealed HDPE sample tubes and maintained at -5 °C until prepared for analysis.

Outside shell lamina was removed to the greatest extent possible with a Teflon scraper. Shell samples were placed in 3% hydrogen peroxide (Fisher Scientific certified) in sealed HDPE tubes and agitated in a sonic water bath (Bransonic 321) for 15 minutes, rinsed twice with distilled water, and dried at 75°C for 24 hours. Dry masses for each sample was obtained. Samples were wet-digested in 0.6 mL concentrated (67%) trace metal grade nitric acid in sealed HDPE sample vials in a 60° C sonic water bath for 30 min. After cooling for 1 hour, samples were brought up to 4 mL with distilled water to give a final concentration of 10% nitric acid.

Lead (Pb) concentrations were analyzed using graphite furnace atomic adsorption spectroscopy (Varian SpectraAA 220 Fast Sequential, Varian Australia Pty Ltd, Mulgrave Australia) with a Varian Graphite Tube Analyzer 110. A Varian notched-partition graphite tube with a pyrolytically coated forked platform was used for all analyses (Brodie 1985). A 40 ppb bulk solution was made through serial dilution of a 1000-ppm standard solution (High Purity standards) with distilled water. Standard curves were completed prior to every ten samples being analyzed with 0, 8, 20, 32, and 40 ppb solutions in triplicate sets. A palladium nitrate (800 ppm in 2% HNO₃) matrix modifier was co-injected with all samples and controls to stabilize analyte during desolvation and ashing steps (Rothery 1988, Subramanian *et al.* 1993).

RESULTS

Over half of the captured *T. scripta* had evidence of previous or active USD (51.5%), greater than that of *P. rubriventris* (35.7%). This difference was not statistically significant (Chi-square binomial comparative trial, $\chi^2 = 0.125$, $\chi^2_{0.05,1} = 3.841$, $P > 0.25$). Table 1 shows the severity of USD for each species.

Lead (Pb) was found in the bone of the carapace of both species. The mean Pb concentration in *P. rubriventris* (16.2 ppm) was higher than found in *T. scripta* (5.8 ppm). This difference was significantly different (Single factor ANOVA, $F = 5.54$, $F_{0.05(2)1,40} = 4.08$, $P = 0.025$). Lead concentrations were significantly higher in diseased *P. rubriventris* (21.9 ppm) than diseased *T. scripta* (5.6 ppm) (Single factor ANOVA, $F = 4.34$, $F_{0.05(2)1,21} = 4.32$, $p = 0.050$). However,

mean concentrations of Pb in healthy *P. rubriventris* and *T. scripta* were not statistically different, 10.5 and 5.9 ppm respectively (Single factor ANOVA, $F = 2.00$, $F_{0.05,(2)1,17} = 4.45$, $p = 0.175$). The greatest concentration of Pb found in *P. rubriventris* carapace was 83.1 ppm, while the greatest concentration in *T. scripta* was 15.2 ppm.

Table 1. Severity of USD of diseased *Trachemys scripta* and *Pseudemys rubriventris*. No male *P. rubriventris* with evidence of USD were captured. The methods for calculation of severity of damage are described in Methods.

Species (sex)	n	Mean USD severity (%)
<i>T. scripta</i> (both)	50	12.9%
<i>P. rubriventris</i> (both)	10	16.4%
<i>T. scripta</i> (female)	41	15.0%
<i>T. scripta</i> (male)	9	3.1%
<i>P. rubriventris</i> (female)	10	16.4%

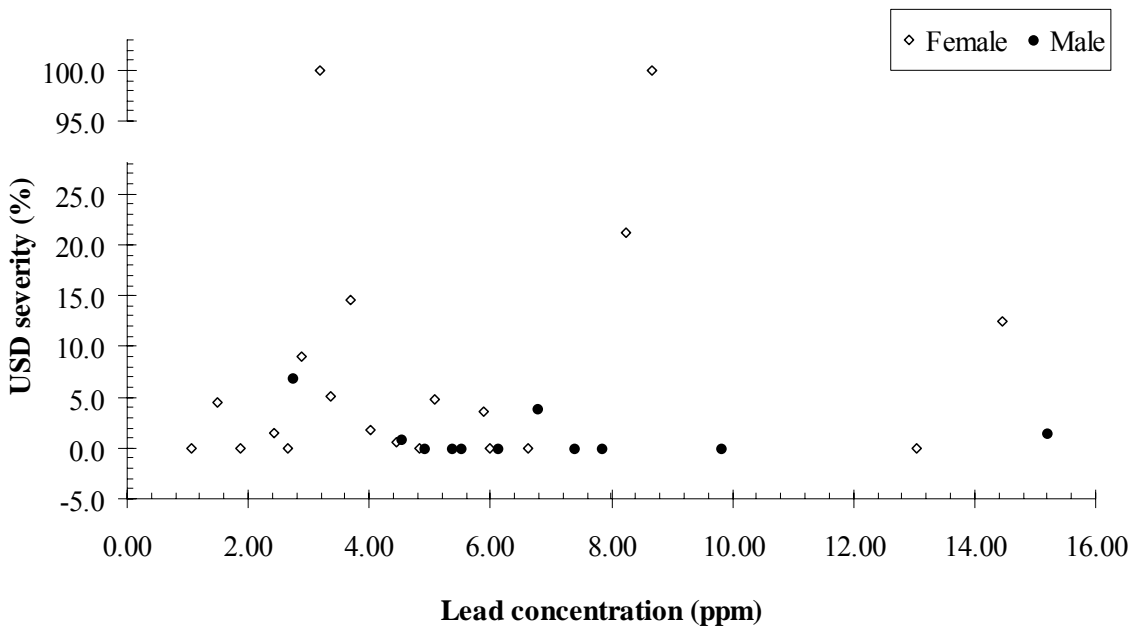


Figure 1. Comparison of severity of USD and concentration of lead in the bone of the carapace of *Trachemys scripta* (female $n = 21$, male $n = 11$). Severity is based on the percent of the plastron damaged by USD as described in Methods. Data for severity of disease were arcsin transformed for statistical analysis (Zar 1999).

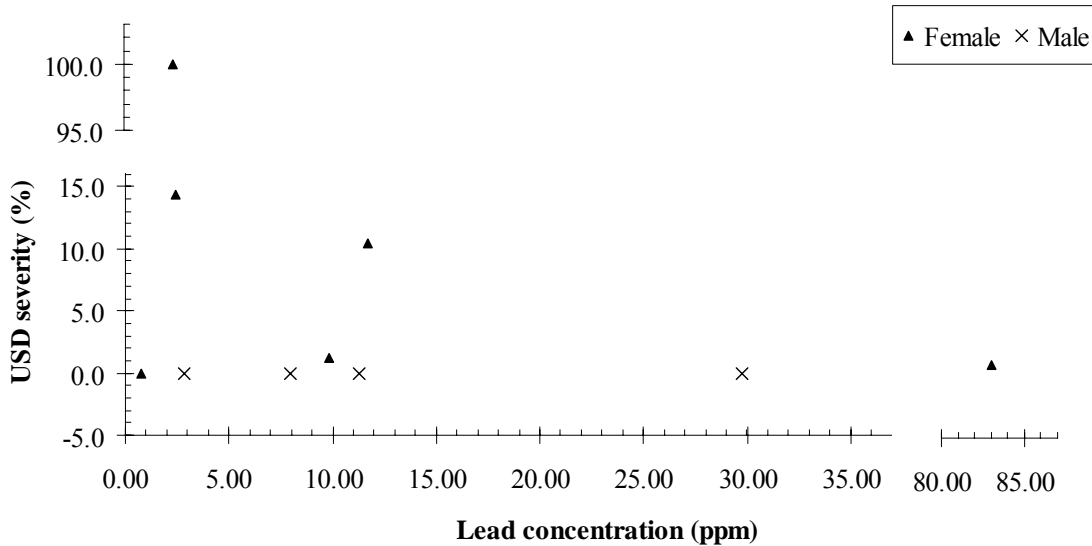


Figure 2. Comparison of severity of USD and concentration of lead in the bone of the carapace of *Pseudemys rubriventris* (female n = 6, male n = 4). Severity is based on the percent of the plastron damaged by USD as described in the Methods. Data for severity of disease were arcsin transformed for statistical analysis (Zar 1999).

The relationship between severity of USD and the concentration of Pb in the bone of the carapace was not significant in female *T. scripta* turtles (Linear regression $r^2 = 0.016$, $P = 0.58$) (Fig 1). Likewise, no relation exists in male *T. scripta* (Linear regression $r^2 = 0.013$, $P = 0.74$). Three turtles with the greatest concentrations of Pb, from 13.0 – 15.2 ppm, only had severity of USD ranging between 0.0 – 12.5%. Likewise, the two turtles with 100% USD damaged plastrons had relatively low Pb concentrations, 3.2 and 8.8 ppm.

The relationship between severity of USD and the concentration of lead in the bone of the carapace of *P. rubriventris* was also not significant (Linear regression $r^2 = 0.097$ $P = 0.55$) (Fig 2). The turtle with the greatest concentration of Pb, 83.06 ppm, had less than 1% of the plastron damaged by USD. Conversely, the turtle with 100% USD damaged plastron had a Pb concentration less than 2.5 ppm.

DISCUSSION

Both species had mean shell bone Pb concentrations less than 17 ppm (Fig 19). *P. rubriventris* did have a significantly higher mean concentration of Pb than *T. scripta*, 16.2 and 5.8 ppm respectively. Feeding habits may play a role in lead concentration differences. Demnicki (2007) found that *P. rubriventris* in Lake Maury were 100% herbivorous, while *T. scripta* was somewhat omnivorous; 67% of the volume found in female *P. rubriventris* consisted of vascular plant material and 30% of algae. For *T. scripta*, 78% of the volume in female diet consisted of vascular plant material, with no evidence of the consumption of algae. Several species of green algae accumulate large concentrations of Pb; the species *Ankistrodesmus falcatus* has been found to accumulate close to 20,000 ppm (Moore 1991, Wong *et al.* 1987). Demnicki (2007) did not

find any evidence of algae consumption by male *P. rubriventris*. While more carnivorous turtles should be a greater risk for bioaccumulation due to their higher trophic level (Keller *et al.* 2006), the magnitude at which algae hyperaccumulates lead may surpass the magnification of Pb with increasing trophic levels.

Our results indicate that there is no correlation between lead concentration and USD. While *P. rubriventris* had significantly greater lead concentrations, the frequency and severity of USD between *T. scripta* and *P. rubriventris* was not significantly different. *P. rubriventris* with USD had significantly higher concentrations of lead than did diseased *T. scripta*; concentrations were not significantly different in healthy turtles of each species. This may suggest that *P. rubriventris* may suffer less consequence from lead accumulation than *T. scripta*. Yet, linear regression revealed no relationship between concentration of lead and severity of USD for either species (Figs 1 and 2). The turtle shell may serve as a sink for bioaccumulated lead, reducing concentrations found in visceral tissues. Sakie *et al.* (2000) found that the carapace accounts for 15% of Loggerhead and Green Turtle mass, and accumulates nearly all the body's lead burden.

Other environmental contaminants should be investigated. Cadmium and mercury have been detected in visceral tissues of several marine turtle species (Caurant *et al.* 1999, Sakai *et al.* 2000, Storelli and Marcotrigiano 2003). These metals have several toxic effects including suppression of the immune system (Chowdhury and Chandra 1987, Zelikoff 1993, Krocova *et al.* 2000, Keller *et al.* 2006). Likewise, bioaccumulation of polychlorinated biphenyls (PCB) should be studied. PCBs are the leading cause of lake, pond and reservoir impairment in Virginia (VA DEQ 2002). Studies on the toxicological effects of PCB are numerous; this class of chemicals has been implicated as endocrine disrupters, carcinogens, and immunosuppressors (Stone 1992).

The majority of diseases found in wild populations of reptiles are usually a consequence of environmental stressors, such as pollution or habitat destruction, which decrease an individual's resistance (Gibbons *et al.* 2000). All of the factors associated with ulcerative shell disease in freshwater turtles are known environmental problems of urban lakes and other aquatic systems. The presence and severity of USD in freshwater turtles could be an easily quantified indicator of ecosystem health.

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FISH TISSUE MERCURY STATUS AND TRENDS FOR NORTH FOLK HOLSTON RIVER, VA

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KEY WORDS: bioaccumulation, fish tissue, mercury, status and trends, TMDL

ABSTRACT

Mercury contamination of the fish tissue in North Fork Holston River prior to 1972 led to a ban on fish consumption by the Virginia Health Department. The ban extends about 84 miles from Saltville to the Virginia/Tennessee state line and prohibits the consumption of any fish. A total maximum daily load (TMDL) is scheduled for 2010. Long term trends since 1982 document substantial reductions in fish tissue mercury levels from target species, northern hogsucker, rock bass and redbreast sunfish. Despite extensive remediation efforts to the river and upland sites over two decades, recent status and trends show mercury levels are no longer declining and show an increasing trend from 2000 to 2005. Fish tissue mercury concentrations remain high (above 0.5 mg/kg) for the target species with a positive correlation found between annual mean fish tissue mercury concentrations and annual flows since 1997. The northern hogsucker, rock bass and smallmouth bass (added to the monitoring program in 2003) displayed bioaccumulation of mercury with age while sunfish showed no significant difference between younger and older fish. The next step will include source assessments and mercury loadings to the system. This will be done in support of developing a TMDL for mercury as part of a consent decree.

INTRODUCTION

In 1974, Virginia Department of Health (VDH) issued an advisory prohibiting consumption of any fish caught from North Fork Holston River (NFHR) from Saltville ~84 miles to the Virginia/Tennessee state line due to mercury contamination. The source of contamination was due to historic releases from an Olin manufacturing facility in Saltville (VA) between 1929 and 1950. While the majority of the mercury catalyst was captured and reused, losses from the system to the river occurred over those 21 years of operation by the facility. Initial studies indicated that fish mercury concentrations would slowly decrease without any remedial action; however, no discernable declines in fish tissue levels were observed in the 29 years since mercury contamination in the river was first discovered. Following several decades of discharge, contaminated sediments were excavated from the river bottom near the facility in 1982. This action removed what was believed to be the largest source of mercury. Additional restoration was completed around 1994 to prevent further release of mercury to the river. It included construction and operation of a treatment plant to process effluent from the 75 acre Pond 5. In 1997 a cap was placed on Pond 5 and cover added to Pond 6.

Based on these actions, river wide annual mean contaminations of mercury in local fish populations were studied. Olin (2005) reported a declining trend in fish tissue mercury from 1982 to 2005. Their study was river wide with all age classes combined for each species. The primary objective of the present study was to conduct additional assessment of mercury contamination in fish collected from the river below RM 81. It compared the declining trend over three periods -1982 to 2005, 1997 to 2005 to 2000 to 2005 with an emphasis on the latter since it best reflects the most recent water quality assessment period.

METHODS

The annual surveys of fish tissue (filet) mercury concentrations (mg/kg) for the NFHR have been monitored since 1982. This survey determines the concentration of mercury in three target species: northern hogsucker (*Hypentelium nigricans*), rock bass (*Ambloplites reupestris*) and redbreast sunfish (*Legomis auritus*) from selected locations in the river. These stations include monitoring both upstream and downstream of the Saltville Waste Disposal Site. Two upstream locations are river mile (RM) 98 and 84 with nine downstream sites (RM 77 to 8) (Olin 2005). The disposal site and ponds (5 and 6) are located between RM 84 and 81.

Since filet mercury concentrations were generally selected from size ranges of the target fish (northern hogsucker 23 to 32 cm, rock bass 15 to 25 cm, and redbreast sunfish 12 to 20 cm total length), size distributions were tested for normality. Smallmouth bass (*Micropterus dolomieu*) was added to the target species list in 2003. The relative status of each station by RM was calculated from all data collected during that period. While nonlinear trends may have improved the overall analysis on a few time series, linear trend slopes were used for consistency in the mercury concentration time series plots. The analysis was divided into current status with trend analysis over three periods (1982-2005, 1997 – 2005, and 2000 – 2005). No trends were developed for smallmouth bass since just two years of data were available. Trend analyses were conducted on data collected at eleven stations above and below the Saltville facility beginning at RM 98 and ending at RM 8. An association between mercury fish tissue levels and flow was also tested. There was no attempt to flow-adjusted concentration trends at this stage of the study.

A one-way ANOVA was used to test bioaccumulation of mercury by fish age. Only fish collected between RM 77 and 22 were used in the analysis from 2004 and 2005. The null hypothesis (H_0) assumed there was no difference in mercury concentrations between age classes while the alternate hypothesis (H_A) tests the assumption that older fish contain higher levels of mercury in their tissue than younger age classes.

RESULTS

The current status of mercury levels in fish tissue from the NFHR were plotted from the 2004-05 monitoring period (Figure 1). Despite size selection for filet tissue analysis, fish sizes were normally distributed. All three species were observed to have fish tissue mercury below 0.5 mg/kg above RM 77. At and below RM 77, mercury levels increased with averages above 0.7 mg/kg. Both northern hogsucker and rock bass displayed little change in tissue mercury levels between RM 77 and 22. A second peak in sunfish and smallmouth bass was noted around RM 61. By RM 8, all four species showed declining mercury fish tissue levels. Smallmouth bass

displayed the largest range in mercury levels along this section of study area with highest mercury levels around RM 77 and decreasing downstream.

The historical mercury trends of mercury in fish tissue for three target species from 1982 to 2005 is shown below for RM 77 to 22 (Figure 2). All three species displayed a declining (negative) trend over the twenty-three years study period (northern hogsucker $R^2=0.69$, rock bass $R^2=0.65$, and sunfish $R^2=0.56$). No trends were observed for any of the species between 1997 and 2005; however, increasing (positive) trends were observed for all three species. Strongest trends were reported for both the northern hogsucker ($R^2=0.80$) and rock bass ($R^2=0.65$) with a weaker trend noted in sunfish ($R^2=0.54$) since 2000 (Figure 3).

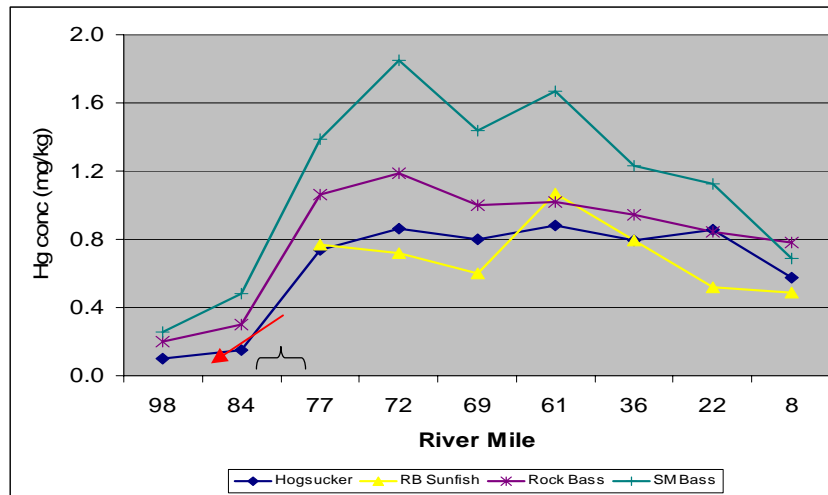


Figure 1. NF Holston River fish tissue mercury concentrations (mg/kg) by river mile for four fish species (Saltville facility and ponds 5 and 6 located between RM 84 and 81, red arrow) during 2004-05.

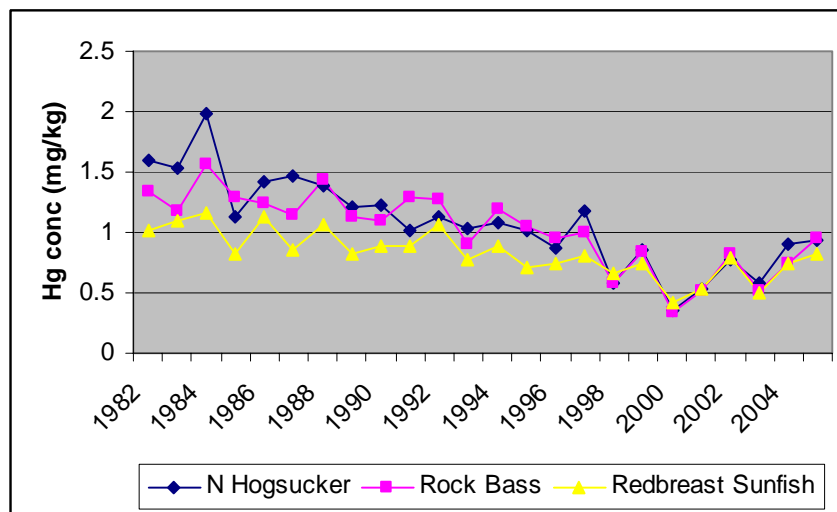


Figure 2. Declining trends in mercury fish tissue for three target species from 1982 to 2005 (RM 77 to 22).

No association was found between flow and fish tissue concentrations for any given year over the study period. However, there was some correlation between flow and fish tissue levels the following year. For example, the 1997 flow was correlated to 1998 fish tissue levels. This association was strongest since 1997 with rock bass displaying the strongest correlation followed by the sunfish and northern hogsucker (Figure 4; Table 1) (note, smallmouth bass were not considered since they were not sampled until 2003).

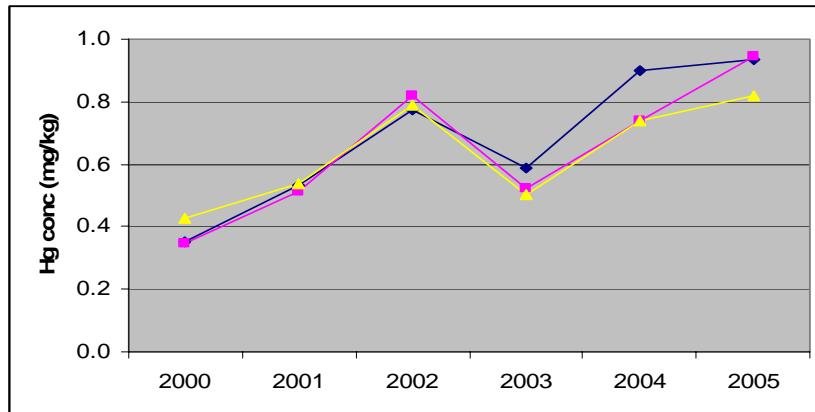


Figure 3. Increasing trends in mercury fish tissue for three target species from 2000 to 2005 for RM 77-22.

Table 1. Correlation coefficient (r) between flow and fish tissue mercury levels for three target fish species from 1982 – 2005.

Parameter	No Lag 1982-2005	Yr Lag 1982-2005	Yr Lag 1997-2005
Northern Hogsucker	-0.03	0.04	0.45
Redbreast Sunfish	0.03	0.30	0.73
Rock Bass	-0.01	0.23	0.82

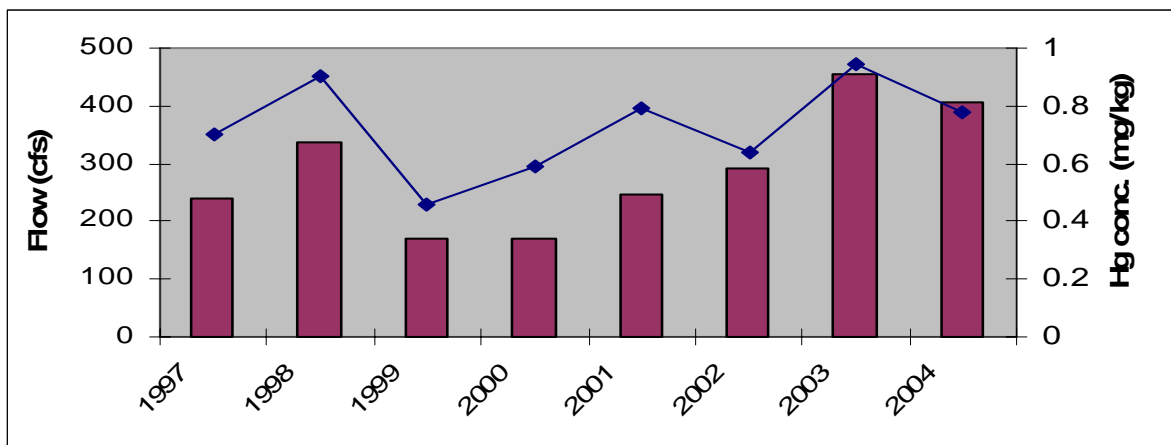


Figure 4. Rock bass fish tissue mercury concentrations (line) (mg/kg) (1998 to 2005) and flow (bar) lagged by year (1997 – 2004) (1997 flow corresponds to 1998 fish tissue mercury concentrations).

Three of the four species demonstrated bioaccumulation of mercury with increasing age (size) (Table 2). Based on the most recent data, northern hogsucker showed older fish (2+ years old) maintain body burdens above 0.5 mg/kg (Virginia's criterion for fish consumption advisories) as did the rock bass. All age classes of the sunfish and smallmouth bass were above 0.5 mg/kg mercury.

Table 2. One way ANOVA for fish tissue mercury concentration vs age class below RM 80 from 2004 and 2005.

Species	F	P-Value	F Critical	Ho
Northern hogsucker	10.4	3.87E-06	2.7	Reject
Redbreast sunfish	1.5	0.18	2.3	Accept
Rock bass	3.8	0.003	2.3	Reject
Smallmouth bass	3.0	0.009	2.1	Reject

As shown above, there was a significant difference between age and fish tissue levels noted for three of the four target species. While northern hogsucker showed the most significance, borderline differences were noted in smallmouth bass and rock bass between age and mercury fish tissue levels. Despite a positive (increasing) trend in the mean age to fish tissue mercury concentration, there was no significant difference between or within age classes for the sunfish. A marginal correlation was observed between fish tissue mercury concentrations and weight of fish from RM 77 to 22 (Figure 5) ($R^2 = 0.43$, $n=49$). There was virtually no relationship downstream of the Saltville facility and ponds between RM 77 and 61 ($R^2 = 0.13$, $n=30$).

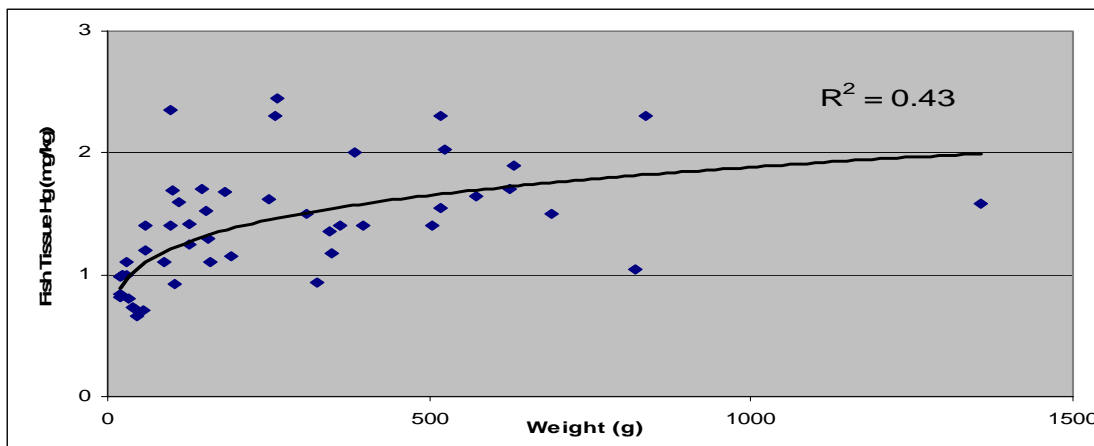


Figure 5. Scatter-plot of smallmouth bass fish tissue mercury concentrations by weight for the river (RM 77 to 22) from 2003 to 2005.

DISCUSSION

Based on the fish tissue mercury monitoring in 2004 and 2005, fishes sampled above RM 77 contained mercury below 0.5 mg/kg the criterion threshold. At and below RM 77, mercury levels increased and remained uniform with averages above 0.7 mg/kg through RM 22. Both sunfish and smallmouth bass showed a second peak around RM 61. Smallmouth bass displayed the largest range in mercury levels between up vs downstream sites with highest mercury levels

around RM 77 and decreasing downstream. Unlike smallmouth bass, little differences in fish tissue levels were noted for northern hogsucker and rock bass between RM 77 and 22. By RM 8, all four species display declining mercury fish tissue levels.

Previous studies showed that mercury was associated with river flows (Golder 2004). As flows increased so did mercury loads. This study demonstrated that fish tissue mercury was positively correlated with flows, but only to those from the previous year and only since 1997. Despite a low flow year such as 1999, fish tissue mercury levels remained at or above the criterion threshold of 0.5 mg/kg. With the increasing (positive) trend since 2000, fish tissue mercury concentrations were well above the criterion below the Saltville facility and ponds at RM 77. The status results infer that mercury continues to reach the river and fish species and remains available to higher trophic levels along the entire stretch of river. While surface water column data prior to 1997 ranged between 0.013 µg/L and 0.096 µg/L of total mercury, concentrations since 1997 were below the reporting limit of 0.005 µg/L (Olin 2005). Recently, methylmercury was detected in surface samples at concentrations ranging from 0.000335 to 0.000434 µg/L. While macroinvertebrates have been monitored for over 20 years, they were not considered in this survey since

CONCLUSION

Based on these results, there are both decreasing and increasing trends in total mercury fish tissue levels depending on the period of review. While there was a declining long term trend since 1982, mercury fish tissue levels have increased since 2000. All four target fish species maintained body burdens above 0.5 mg/kg (trigger value for the fish consumption advisory by the VDH) during 2004 and 2005. Mercury fish tissue levels remained high in older fish with all age classes of the sunfish and smallmouth bass reported mercury levels above the trigger value.

Fish tissue levels among target species remain high and relatively uniform along 80 miles of river. While the Saltville facility site and ponds continue to be a source of mercury contamination to the river, one or more other source may be contributing mercury to the river downstream. Additional studies point to contaminated floodplain soils further downstream may be a source due to surface soil redispersion through surface runoff or river overflows over time. The next phase of this study will be to conduct a mercury source assessment in support of TMDL development. In addition, a bioaccumulation factor will be calculated for the river.

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COMPARATIVE ULTRASOUND ASSISTED LEACHING OF SELENIUM AND ARSENIC FROM COAL MINING ASSOCIATED ROCKS AND SEDIMENTS

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ABSTRACT

The leaching of selenium and arsenic species from coal mining waste into streams has recently been identified to be a problem in regions where mountaintop removal and valley fill mining is practiced. Three core rock samples and three stream sediment samples were subjected to ultrasound extraction in nanopure water followed by analysis of the extracts using Graphite furnace and Hydride generation atomic absorption spectrometric techniques for total selenium, total arsenic and selenium (IV). Although stream sediment sampling locations are associated with valley fills, two were downstream from abandoned mining areas and one was downstream from an active coal mine. Ultrasound extraction revealed that accelerated release rate constants for arsenic were ten times greater than those of selenium. Additionally, there was a three order of magnitude difference between the released selenium and arsenic concentrations. Sediments samples collected from sampling locations with abandoned mining activities had lower selenium concentrations compared to the sample collected from an area downstream from an active mine. Further, water samples collected from the active mining sediment sampling site had higher selenium concentrations compared to the water samples associated with abandoned coal mines. Arsenic concentrations in stream sediment were in mg/kg range however in stream water the concentrations were below the limit of detection.



Session IV-C

Restoring the Chesapeake Bay: Innovative Monitoring Methods and Management Efforts

- GIS-Based Forested Riparian Buffer Analysis of Selected Streams in the Chesapeake Bay Watershed
- Deriving Local Benefits in West Virginia from Chesapeake Bay Program Efforts: Institutions for Effective Regional Collaboration
- Nutrient Assimilation Credits: Concept and Illustration to Oyster Aquaculture
- Water Quality Trading: An Examination of the Pennsylvania Nutrient Trading Program

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GIS-BASED FORESTED RIPARIAN BUFFER ANALYSIS OF SELECTED STREAMS IN THE CHESAPEAKE BAY WATERSHED

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KEY WORDS: riparian, buffer, Chesapeake Bay, stream, GIS

ABSTRACT

Chesapeake Bay jurisdictions have committed to major increases in forested riparian buffers as part of the restoration program. However, basic data is lacking on buffer status and its change in the watershed. A GIS/aerial imagery-based method was used to track change in riparian buffer forest cover from the 1993/94 period to the 2000/04 period in selected streams in eight counties in the Bay watershed. In three counties sub-watersheds were selected randomly for analysis and while in the other five “growth hot spot” areas were utilized. The orthophoto imagery, along with various ancillary spatial data, was used to map the historical and “present day” land use and land cover along each of the identified drainage features. This analysis was limited to an area defined by a fixed, 100-foot wide, buffer-zone bounding the drainage features. A detailed cover-type analysis was completed using a minimum mapping unit (MMU) of 1000 square meters. Results indicated that riparian forest cover in the “growth hot spot” areas decreased by 2.7%, ranging from 1.1% to 5.2% for individual counties. In the counties with randomly selected sub-watersheds, the results were mixed. Results indicate that forest buffer extent is still decreasing in areas experiencing rapid growth despite restoration efforts.

INTRODUCTION

Forested riparian buffers have been recognized as valuable resources in the efforts to decrease nutrient and sediment loads into the Chesapeake Bay. In addition, forested areas along streams provide canopy, shade, inputs of leaf litter for stream food webs and habitat for a variety of both terrestrial and aquatic organisms. The importance of these contributions has been recognized by the Chesapeake Bay Program and its partners and incorporated into goals for restoration of these streamside forest buffers. The original goal of 2010 miles of restored buffers by 2010 was reached well ahead of time and the goal was increased to 10,000 miles by 2010.

Restoration efforts may be offset by continued destruction of forest riparian buffers by land clearing activities for urban, suburban, and agricultural activities. Accounting of the restored mileage can be obtained relatively easily by tracking known restoration projects throughout the watershed. However, the elimination of existing buffers is not so easily tallied as these activities are not generally required to report their buffer impacts.

This paper reports on a study designed to determine net changes in riparian buffer extent in selected areas of the Chesapeake Bay watershed over a roughly decadal period using a combination of GIS and aerial photography.

METHODS

The study was conducted in two phases. In the first phase, three counties were chosen that represented the three states constituting the majority of the watershed: Frederick County, MD; Lancaster County, PA; and Prince William County, VA. Each county was divided into the smallest, standardized subwatersheds available as defined by the National Watershed Boundary Dataset (NWBD). Within each county, five sub-watersheds were randomly selected for analysis. The selected subwatersheds constituted about 10-15% of the total county land area.

In the second phase, study sites were selected based on an analysis prepared by Peter Claggett of the CBPO-U.S. Geological Survey. This analysis mapped “growth hot spots” which are areas of relatively high residential growth (change in single-detached housing from 1990–2000) and high impervious surface growth (1990–2000). From this analysis, parts of five counties were selected for comprehensive stream buffer analysis: Dauphin County, PA; Prince George’s County, MD; Spotsylvania County, VA; Henrico County, VA; and James City County, VA. The area of comprehensive analysis constituted 25-50% of each county; thus, the total mileage assessed was substantially greater in the second phase than in the first.

USGS, Hydrology DLG files Spatial Data Transfer Standard (SDTS) and 1:24,000 scale, digital raster graphics (quadrangles) were used to define and map the permanent and temporary (seasonal) streams, rivers, and open water bodies in each of the selected study sites.

Historical (circa 1993) and “present day” (circa 2003) orthophoto imagery was collected for each selected county. This imagery came in a variety of resolutions ranging from 1-foot (VBMP) to 1-meter (DOQQ) ground sampling distance (GSD). The orthophoto imagery, along with various ancillary spatial data, was used to map the historical and “present day” land use and land cover

along each of the identified drainage features. This analysis was limited to an area defined by a fixed, 100-foot wide, buffer-zone bounding the drainage features.

A detailed cover-type analysis was completed using a minimum mapping unit (MMU) of 1000 square meters. All analysis and digitizing was completed at a scale of 1:4,000.

All vector data in the GIS is provided as ESRI Shapefiles. Raster image data is provided in either ERDAS Imagine, GeoTIFF, and Mr. SID file format. The data is projected in UTM (Zone 18), NAD83, with units in meters.

Results on riparian buffer extent by land use were reported in acres. These units were translated into miles by multiplying by 435 (a 100 ft wide buffer that is 435 feet long would yield an area of 1 acre) and dividing by 5280 ft per mile.

RESULTS AND DISCUSSION

The three counties surveyed in phase one exhibited somewhat different patterns in land use change in the riparian buffer zone (Table 1). Frederick and Prince William Counties exhibited decreases in forested land uses within the riparian zone of surveyed streams while Lancaster County exhibited an increase. The net decrease in Frederick County forested buffer extent was very small and forest riparian buffer as a percent of total buffer remained at a relatively low level of about 41%. The main change in Frederick County was a shift from agricultural uses to developed land uses. In the Lancaster County subwatersheds surveyed, a more substantial gain in forest buffer of about 3% was observed over the 10 year period bringing the percent of buffer in forest to 57.4%. In these watersheds, the main losses were grasslands and shrub/scrubland. In the Prince William County watersheds examined, a substantial loss of forest riparian land cover of about 3% was observed. Developed land uses including transportation accounted for most of the net loss bringing the percent of buffer in forest to just below 60%.

Table 1. Results from the phase one study.

County	Riparian Forest Acreage Surveyed	Forest as % of Total Buffer		Percent Change in Acreage/Mileage
		Base Year	Final Year	
Frederick (1993/2000)	4522	41.4%	41.3%	-0.3%
Lancaster (1993/2002)	7031	55.7%	57.4%	+3.2%
Prince William (1994/2002)	8831	61.1%	59.4%	-2.8%

Changes in buffer mileage in the portions of each county surveyed over the study period were calculated at -0.4 miles for Frederick County, +10 miles for Lancaster County, and -13 miles for Prince William County. Since the watersheds were chosen randomly within each county, an extrapolation to the whole county is justified. A rough scaling of 8 to the whole county would yield changes in buffer mileage ranging from -104 to +80 on a per county basis.

Pooled results from the phase two effort concentrating on rapidly developing areas revealed a decrease of 2.7% in the extent of forest riparian buffers over all watersheds studied (Table 2). Forest as a percent of total buffer extent decreased from 62.4% to 60.7%. In the areas studied, a net loss of 166 miles of forest was found in the riparian zone. A net loss was found in the examined areas of each of the five counties ranging from 1.1% to 5.2% (Table 3). The area of Henrico County examined lost 5.2% of its forest riparian buffer with the majority going to residential and commercial/industrial land uses. In the other counties, losses of forest land use were generally coupled with gains in developed land uses such as developed open space, residential, and commercial/industrial. In some cases gains in shrub/scrubland were important and in Prince Georges County, herbaceous wetlands showed a gain.

Table 2. Overall results from the phase two study.

	All Areas Surveyed
Forest Buffer Acreage – Base Year (1993/1994)	73556 acres
Forest as % of Total Buffer – Base Year	62.4%
Forest Buffer Acreage – Final Year (2002/2003)	71543 acres
Forest as % of Total Buffer – Final Year	60.7%
Change in Riparian Forest Buffer Acreage	-2013 acres
Change in Riparian Forest Buffer Miles	-166
Percent Change in Acreage/Miles	-2.7%

Table 3. Change in riparian forest buffer area by county. Phase two study.

County	Percent Change in Riparian Forest Buffer
Dauphin Co., PA	-1.7%
Prince George’s Co, MD	-1.1%
Spotsylvania Co., VA	-2.7%
Henrico Co., VA	-5.2%
James City Co., VA	-1.8%

Results from the phase two effort are difficult to generalize to the watershed or even to the whole county because the areas were selected as areas of rapid population growth. In that context they are representative of areas that may pose challenges to the goal of increasing forest riparian buffer due to heavy development pressure.

Results from this study indicate that while restoration efforts may be progressing rapidly toward Bay Program goals, net increases in riparian forest extent may be more difficult to achieve. This results primarily from the potential for forest clearing in developing areas which may partially or even completely offset restoration gains. The original restoration goals were set in 1996 so the period of this study roughly overlaps with the period in which restoration has been credited with increasing forest riparian buffer length by over 2000 miles. However, the results presented in this study indicate that many areas continued lose forest buffers during this period. This suggests that if net increases in forest buffer extent are to be achieved, then restoration efforts will have to be intensified and/or steps will need to be taken to minimize ongoing losses.

CONCLUSIONS

A combined GIS-aerial photography approach was successfully used to determine changes in forest riparian buffer extent in portion of eight counties in the Chesapeake Bay region over about one decade from the early 1990's to the early 2000's. Results indicate that losses in forest riparian buffer may offset gains from restoration efforts, particularly in rapidly growing areas. To obtain the net gains in riparian forest buffer envisioned by the Bay restoration effort, losses due to forest conversion must be estimated and offset by additional restoration efforts.

ACKNOWLEDGEMENTS

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DERIVING LOCAL BENEFITS IN WEST VIRGINIA FROM CHESAPEAKE BAY PROGRAM EFFORTS: INSTITUTIONS FOR EFFECTIVE REGIONAL COLLABORATION

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KEY WORDS: water resource management, institutional arrangements, common pool resources

ABSTRACT

Chesapeake Bay restoration efforts are institutionally and conceptually disconnected from local decision-makers in many communities that are far upstream or in headwater areas of the Chesapeake Bay Watershed. Participation by these upstream/headwater communities in state tributary strategy projects can yield important Bay and local benefits if there is effective cross-jurisdictional collaboration. Lessons are drawn from efforts to restore the Great Lakes using projects in communities far from the Lakes' shores. Four suggestions for the Chesapeake Bay are made which address formal and informal institutional obstacles to collaboration among stakeholders that could otherwise yield win-win/positive sum outcomes.

INTRODUCTION

The Chesapeake Bay states and the District of Columbia have agreed to substantially reduce nutrient and sediment loadings in Chesapeake Bay drainages as part of the Chesapeake Bay restoration program. Implementation of this agreement requires significant financial outlays at local, state and regional levels as well as for individual land managers and permittees who face strengthened regulations on discharging nutrients and sediments into public waters. Currently, there seems to be an institutional and conceptual disconnect between regional water quality goals and the local planning and decision-making. Bridging this gap can help cap load allocations be attained more quickly and cost effectively through the development and implementation of innovative and integrated local institutional solutions which yield local benefits (e.g. water quality, source water protection, infrastructure cost savings, etc.).

Particularly in communities farther upstream from the Bay, some parties (communities, private businesses, or local decision makers) may be directly affected by the Bay Program *costs* but have only an indirect or no stake at all in the improved health of the Bay. This physical distance is exacerbated by political distance (in this case state boundaries) that can reduce local buy-in for projects that are not apparently benefiting the "home team." The nutrient reduction practices and investments targeted in the WV Tributary Strategy for Urban and Mixed/Open lands, however,

can generate important direct benefits for local WV communities, counties, or even private business interests and permit holders, which are ancillary to the Bay restoration benefits.

Tackling the geographically and organizationally fragmented and quickly expanding sector of non-point source pollution in urban/exurban areas is critical to the health of the Bay and to protecting local water quality in the face of rapid development that is increasingly occurring in upstream and headwater areas. In fact, according the Bay Program website,

“Urban storm water runoff is responsible for about 16% of phosphorus, 11% of nitrogen, and 9% of sediment loads to the Bay. Chemical contaminants (such as metals) from urban runoff can rival or exceed the amount reaching the Bay from industries, federal facilities and wastewater treatment plants. Urban storm water runoff is responsible for impairments in over 1,570 miles of assessed streams in the Bay watershed and has caused flooding, streambank erosion and habitat and living resource degradation in many areas throughout the watershed.”

As this expensive and complicated source of pollution continues to grow, finding inexpensive, practical and implementable prevention practices is imperative, and it has the potential to save significant public and private money in headwater as well as in tidal water communities.

Coordination among multiple levels of governance can provide an opportunity to design institutions and target investments expressly to facilitate the development and acknowledgement of direct local benefits. Such coordination has the potential to increase the total benefits from the Bay Program investments; it also has the potential to reduce resistance to (and therefore costs of) the Bay program implementation. Acknowledging the headwater/upstream situation to be a classic example of a Common Pool Resource tragedy of the commons or Prisoners’ Dilemma¹, it is critical to identify 1) how to ensure collaboration and communication among stakeholders, and 2) how to ensure that collaboration yields positive environmental (and cultural and economic) benefits for all parties involved.

This paper uses the Rockymarsh Run Watershed, a rapidly developing area watershed within Jefferson County, WV as an example of how cooperation at various levels of government has the potential to reduce overall costs of reaching interrelated environmental, public health, and economic development goals. Conditions for achieving this ideal are drawn from academic literature in collective action, ecosystem management, and environmental policy. In particular, research and analysis of the Great Lakes Remedial Action Plan program (started in the mid-1990s) is used as a very similar case of mobilizing local actors over a large geographical area to act on radiation projects that are not borne out of immediate local needs but rather driven by an organization with downstream water quality goals. Research indicates that collaborative action across jurisdictions for local and regional goals may be more successful where there is adequate

¹ A Prisoners’ Dilemma is one in which two parties, who do not communicate with one another, face the prospects of gaining (if both choose to support each other) or losing (if either or both “defect”). The most simplistic version of this situation results in both parties defecting (not cooperating) for fear that the other will defect and both will lose anyway. Communication and collaboration can overcome the lack of trust so that both parties cooperate and gain. Baland and Platteau (2000) describe several ways that similar games apply to natural resource management and how different situations that lead to non-cooperation can be turned into games that lead to cooperation if influenced by various institutional arrangements.

buy-in, mutual benefits, flexibility, authority, and capacity building support. There must also be strong local leaders with vision to coordinate around a goal, and a strong driver(s) (incentives/sanctions) at the local level which is important for yielding effective results of collaboration.

This paper discusses pertinent literature and some of the overlapping local, county, and state benefits that can be generated from investments and behavior changes intended to help restore the Bay based on the example of Rockymarsh Run. In conclusion, four suggestions for how Bay restoration efforts can be more successful in headwater/upstream communities are detailed. These suggestions include a few innovative approaches that other Bay states and local governments have taken to ensure that investments made toward gains in the Bay are also generating benefits at home. It also suggests that conceptualizing and promoting Chesapeake Bay Program efforts with a focus on the goal of a healthy Chesapeake Watershed (rather than simply on the restoration of the Bay itself) might improve buy-in and perception of program legitimacy in upstream/headwater areas.

Regional environmental management literature: Looking to the Laurentian Great Lakes

Integrated water resource management, collective action and collaborative management of common pool resources, and ecosystem management are among the concepts that are increasingly used to design and/or explain social and political involvement in the efforts to sustainably manage natural resources. Evaluation of collective action situations that require collaborative networks has yielded instructive hints at how to strengthen outcomes. Heikkila and Gerlak (2004) describe the classical Prisoners' Dilemma:

“While the potential benefits of collaborative resource management offer obvious incentives for stakeholders to come together, this is no guarantee that collaboration will emerge around a particular resource management dilemma, especially in settings where actors have held traditionally adversarial relationships...Transaction costs involve the time, money, and effort associated with searching for collaborative partners, bargaining with those partners, as well as monitoring and enforcing agreements (Heikkila and Gerlak, 2004).”

Identifying strategies to overcome this dilemma has been the subject of collective action literature for many years, with seminal works authored by Elinor Ostrom, Baland and Platteau, and others. The US-Canadian Great Lakes Commission looked to this literature in the design of its Remedial Action Plan (RAP) program – 43 remediation and protection projects in non-riparian areas throughout the watershed. Like the Bay Program, the Commission is a large network of collaborative partners representing a significant geographical spread. To reach headwater communities and involve them in local restoration and protection projects, the Commission roundtable recommended the following:

- (1) RAP institutional frameworks should be empowered to pursue their mission of restoring uses. Empowerment would be demonstrated by: a watershed focus, **inclusive and shared decision-making, clear responsibilities and sufficient authority**, creative **funding**

capability, **flexibility and continuity** in the process, an **iterative** process of continuous improvement, and commitment to education and outreach.

- (2) RAP institutional frameworks should be used as mechanisms to **coordinate programs at the local level**. Such local coordination should be complemented with governmental commitments to intra- and interagency coordination in work plans.
- (3) RAP institutional frameworks can help **build the capacity** of governments to achieve their goals.

Highlighted in this passage are among the concepts we believe to be critical to the Chesapeake Bay program efforts in the watershed headwaters. Additionally, Hartig and Law (1994) cite a wealth of case study and experimental literature to support that “**policy entrepreneurs**” or mobilizers (i.e. strong local leaders with capacity for long term visioning) play a crucial role in helping bridge multi-jurisdictional planning levels and diverse interests.

Beierle and Konisky (2001) contend, however, that successful collaboration does not necessarily lead to successful and effective implementation of programs and resulting environmental benefits. In their analysis of the RAP program, they found that the collaborative action and local orientation of the RAPs was intended to facilitate implementation by ensuring local ownership and building local capacity. In evaluating each of the 43 RAP areas targeted, they found support for improvements made toward the first three of the following goals, but not for gains in area number four: (1) the quality of decisions made; (2) the relationships among important players in the decision-making process; (3) the capacity for managing environmental problems, and (4) real improvements in environmental quality. With these results, the authors question whether investment in local collaboration is truly a more effective policy strategy than command and control or market based incentives.

Rather than asking whether one approach or another is better, this paper allows that a mix of these strategies is possible and likely to be more effective than relying on any one approach alone. MacKenzie (1997) draws on a slightly different perspective in her analysis of the RAPs, emphasizing the legal and regulatory institutions needed to ensure collaborative management efforts yield results. Four areas of emphasis in her analysis are as follow:

- (1) [Local] stakeholders must have a clear **incentive** to participate **or a sanction** for delinquency [in the regional program];
- (2) Scientific information must be translated into **public policy** and framed within the legal structures that govern society. Individuals with a social science background are uniquely situated to link data to policy;
- (3) In order to sustain support in the political system, programs need to **cultivate legitimacy**. Programs that cannot justify their existence in the public arena are vulnerable to attack or disregard, both of which are anathema in a political climate of cutback management and ultracompetitive funding (federal, state *and* municipal).
- (4) Success of [a conceptual commitment to cooperation and participation among individuals with multiple interests] depends on the cultivation of **an individual and a collective sense of place** and belonging in the area of concern. From these sensibilities a commitment to local stewardship may be fostered. Upstream communities in several areas of concern could be described as actively disinterested.

The following section briefly describes the Rockymarsh Run Watershed and Targeted Watershed Grant project and is followed by a section that draws from the literature to make recommendations for the project and the Chesapeake Bay Program.

Rockymarsh Run watershed

Rockymarsh Run is a direct drain to the Potomac River. It flows through Jefferson County, but its watershed is in both Jefferson and Berkeley Counties – the two eastern-most counties in WV and two of the fastest growing counties in the state and nation. Most of the watershed is likely to be developed in the next five to ten years, and it is primarily sensitive karst topography. Rockymarsh Run is listed on the 303d list for fecal coliform impairment, and it is the state’s third highest priority stream in the Trib Strategy Implementation Plan, second priority stream in the WV Brook Trout Strategy Plan, and important from a variety of perspectives in the Jefferson County Green Infrastructure Plan, including that of source water protection.

The Rockymarsh Run project intends to bridge a remaining disconnect between regional water quality goals and local (headwater/non-tidalwater) planning and decision-making, allowing cap load allocations to be attained more quickly and cost effectively by developing and implementing innovative and integrated institutional solutions at the local level. The project plans to install nutrient and sediment reduction BMPs on the urban and mixed open land of the Rockymarsh Run watershed - in active coordination with utility, state, county, and privately-funded projects and local development plans - in order to maximize local economic and environmental benefits, leverage public support, and align complementary interests and funding (namely for native brook trout restoration, source water protection/public health, and conservation-based development).

This project will demonstrate the benefits of coordinated project planning in Jefferson County, WV as a template for local implementation of Tributary Strategy goals throughout Jefferson and neighboring counties while simultaneously creating a locally supported sustainable pathway for future cap load *maintenance* in rapidly developing rural areas. Central to the project will be the development of West Virginia Department of Environmental Protection (WVDEP)-sanctioned NPS nutrient credits, helping set the stage for the still-developing WVU/NRCS Trading Program to integrate local benefits into its Upper Potomac strategy by 2009. Anticipated project outputs follow.

- Creation of the first WVDEP-approved NPS nutrient credits;
- Attract/utilize funding for “local benefit” projects on Rockymarsh Run that reduce sediment and nutrient loading (e.g. on-site septage pumping and collection, Brook Trout habitat restoration, stormwater-friendly development). Donations of riparian conservation easements or land use covenants for all forested riparian buffers will be actively marketed among participating landowners;
- Develop local/county planning and ordinance templates linking county planning, stormwater, and development priorities to the WV Tributary Strategy;
- Develop “Local Benefits and Restoration of the Chesapeake Watershed” marketing materials, economic analysis, and local government toolbox to facilitate the knowledge transfer of lessons

learned in Jefferson County to Berkeley and Morgan Counties in anticipation of a trading program throughout the WV jurisdiction.

Existing organizations with expressed local, state, and regional interests in projects and goals identified in the Urban Mixed/Open section of the WV Tributary Strategy Plan are listed below. The 2005 WV Tributary Strategy Implementation Plan, however, underscores that, “Implicit in each sector’s Plan, and the overall Plan for WV, is that the activities required to meet the Cap Loads **will not occur if funding is not secured.**” The document estimates the strategy implementation to cost \$900 Million. It is clear, however, from the list above and from Table 1 below, that making the investments described in the WV Trib Strategy are as advantageous to the state, county, and local area as they are to the restoration of the Chesapeake Bay. Acknowledging the situation to be one of a classical common pool resource prisoners’ dilemma (everyone stands to net gain from cooperation and net lose from not cooperating), can help move the RMR project and other interventions in the region forward more effectively, ideally setting an example for other headwater communities and other region-wide environmental restoration/protection projects.

STAKEHOLDERS & PLANS

National/Regional:

- National Fish Habitat Action Plan (Rockymarsh, top priority)
- Mid-Atlantic Highlands Action Program
- Chesapeake Bay Program

State:

- WV Brook Trout Strategy (Rockymarsh, top priority)
- Department of Health & Human Resources Source Water Assessment & Protection Program
- Ranked in top 5 streams to be addressed by WV Tributary Strategy

County:

- Jefferson Co. Green Infrastructure Plan
- Jefferson Co. Farmland Preservation Commission
- E. Panhandle Homebuilders’ Association

Watershed:

- Rockymarsh on 303d list for fecal coliform
- Home to Freshwater Institute
- Much of the riparian farmland slated for development in near future

OVERLAPPING CONCERNS

- Loss of rural aesthetic
- Loss of local agricultural economy
- Local public health protection
- Long term eco/econ sustainability of development

- Legal and regulatory uncertainty
- Reckless development of forested, agricultural & open lands
- Fragmentation/deterioration of habitat
- Eutrophication of local streams or regional waters
- Run-off transport of toxics and other pollutants into local and regional waters

Table 1. Relationship of WV Tributary Strategy projects to various locally-determined goals and benefits in Jefferson County.

WV's BAY IMPLEMENTATION STRATEGY URBAN MIXED OPEN BMPs								
Construction sed. controls		y	y	y				
LID Retrofits	y	y	y	y	y	y	y	
Low Impact Dev Design	y	y	y	y	y	y	y	y
Septic pumping, hookups, & avoid new			y			y		y
Smart Growth planning	y	y	y	y	y	y	y	y
OUTCOMES	Reduce high flow frequency & severity	Reduce Susp. Solids	Reduce TP/TN	Habitat restoration & protection	Temperature	Reduce toxins & metals in runoff	Ground-water recharge	Less land developed per capita
LOCAL BENEFITS to Jefferson County	Reduce property damage , reduced erosion	Improve habitat for recreation & healthier ecosystem/ better pollution sink	Some local stream quality improvements	Tourism economy , preserve local quality of life & property values	Improve habitat, recreation quality of life, property values	Improve/protect public health , locally & down-stream, Source water protection	Increase drought protection , safer water supply , reduce flashiness in streams	Capital and O&M infrastructure savings to public, open space/ ag preservation

CONCLUSIONS AND RECOMMENDATIONS

Reaching the conclusion that many of the stakeholder organizations do have at least a medium to long term stake in supporting many of the WV Tributary Strategy goals for the Urban Mixed/Open sector, we must then ask, “What can be done to ensure effective collaboration and planning as well as effective environmental and economic outcomes for the stakeholders involved?”

This paper makes four recommendations intended to address some of the lessons and cautions identified in literature review from collaborative efforts for regional natural resource

management in the Great Lakes region. These recommendations are listed and discussed as follows:

(1) Revise “framing” of Chesapeake Bay Program outreach efforts to headwater communities. Emphasize the private and public *local* benefits and importance of watershed-wide environmental health, redefining the goal and message of the CBP to be that of a healthy *watershed* not just a healthy Bay. The CBP website states, “*Chesapeake 2000* provided *the opportunity* for Delaware, New York and West Virginia to become more involved in the Bay Program partnership. These headwater states now work with the Bay Program to reduce nutrients and sediment flowing from their jurisdictions...to help restore the Chesapeake Bay.” Headwater communities, locally-elected officials, and public works managers do not necessarily identify with the Bay. Many residents, even officials, in fact may have yet to even lay eyes on the Bay. Being tasked with cleaning up their jurisdictions is not the most inclusive, supportive approach to gaining local buy-in or developing a sense of legitimacy for goals and strategies.

Outreach that emphasizes the protection of a healthy Chesapeake Watershed Community, combined with an emphasis on local benefits in the “Bay Watershed Communities,” could go a long way toward earning local support and cooperation and making the program politically justifiable.

(2) Binding land/water management outcomes that local jurisdictions must achieve with flexible paths for compliance (establishing minimums and maximums as appropriate). Incentives for achieving these outcomes can be positive (e.g. promises of funding) or negative (sanctions). Making state or federal infrastructure funding contingent upon local level implementation of LID or smart growth ordinances is an example of binding incentive – but only if funding is available. Lack of NPS or local caps with associated deadlines and sanction systems creates a largely voluntary program with no room for using sanctions as a tool for expediting change. Without strong incentive or sanction systems, collaboration can lead to superficial or no significant outcomes, undermining prospects for reaching and maintaining loading caps for states, even if targeted reductions are met within the regulated point source community (entities which will end up bearing an unfairly large share of the burden).

The 2007 EPA Report, *Development Growth Outpacing Progress in Watershed Efforts to Restore the Chesapeake Bay: What We Found*, identifies two primary obstacles to be “lack of community-level loading caps and ineffective use of regulatory program to achieve reductions.” The report singles out the weaknesses of the MS4 stormwater program including “lack of numerical water-quality goals, implementation that evolves with each 5-year permit cycle, no requirements to retrofit stormwater systems to achieve greater environmental protection, and their reliance on technology-based rather than water quality-based approaches...and [regulations] do not apply to all small developed areas.”

Without these locally-defined regulatory drivers (sanctions/incentives), local government employees in public works, planning and other city/county departments will be challenged to voluntarily invest time and effort into reducing nutrients and sediment flowing from their jurisdictions; they would be even more unlikely to justify requests for tighter regulations or funding to locally-elected officials. Maryland’s 2007 Stormwater Act and the VA Chesapeake

Bay Act (in Tidewater communities particularly) both require communities to take some specific outcome-oriented actions locally. This gives local decision makers legal *authority* to take appropriate measures for compliance *as well as the obligation* to do so. For those who understand the potential for local benefits as well as Bay benefits, this external facilitation supporting locally proactive measures may in fact be welcome.

(3) Building technical/financial capacity at the local level is another identified need in the EPA 2007 report. Headwater communities, especially those without severe nutrient or sediment problems locally, may likely need detailed technical workshops and guidance - not just pamphlets or slides about green roofs, but the specific technical guidance, ordinance templates, and (funded) workshops for local officials interested in transforming ideas into effective ordinances, but who may lack time or other resources to develop a local strategy and program independently. Such workshops and intensive trainings can/should be produced at the state level in some cases, but federally sponsored trainings and active facilitation of interstate resource sharing would advance this, particularly in non-tidal water states.

Some of these workshops may help communities to find sources of funding internally (fees, taxes, budgets of existing local programs that stand to benefit from projects), from other state or federal agencies, or they may have a grant-writing component to them.

(4) Creative win-win institutional arrangements (or positive sum games) need to be developed, discussed, and disseminated throughout the Watershed, but particularly in areas with fewer state-level mandates to rely upon. Williamsburg, Virginia, for example, does face a variety of state level mandates, but it has developed a program that could be copied throughout the watershed as an exemplary flexible win-win program. The City's program allows for tight restrictions and requirements on developers and others because it has a built in flexibility mechanism and high-profile local benefits. Where requirements and restrictions feasible on a given parcel, developers may opt to pay into an *in lieu* program instead, funds collected pay for locally-identified environmental projects designed for water quality improvement objectives, but which also serve recreational and aesthetic goals. Consolidated projects can also reduce monitoring and enforcement costs over time if fewer parcels are legally committed to maintaining detailed stormwater or low impact development plans.

Examples such as these should be collected and disseminated as templates in workshops for local decision-makers with assistance provided for those interested in adopting and adapting them for their own communities.

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NUTRIENT ASSIMILATION CREDITS: CONCEPT AND ILLUSTRATION TO OYSTER AQUACULTURE

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KEY WORDS: Chesapeake Bay, nutrient assimilative services, economic incentives

EXTENDED ABSTRACT

State and federal efforts to achieve the water quality goals of the Chesapeake Bay have focused exclusively on nitrogen and phosphorus source reduction programs. To date, the implementation of mandatory and voluntary point and non-point source control programs in the Bay states have not, and will not in the immediate future, achieve the nutrient reduction goals established by the Chesapeake Bay Program. Another, but largely unexplored, strategy to meet Bay water quality goals is to remove nutrients directly from Bay waters through enhancing the assimilative capacity of the ecosystem in order to process and remove nutrients. Nutrient assimilation is the capacity of an ecosystem to reduce nutrients within the ambient environment through biological processing, sequestration, and bioaccumulation. In doing so, the ecosystem is able to accommodate the discharge of additional nutrients from point or non-point sources without harmful effects on the aquatic biota or overall water quality.

Nutrient assimilation services could be produced in a number of ways. For example, native Chesapeake Bay oysters have long been recognized for their water filtering capacity. Wild, unmanaged populations, however, have been dramatically reduced by disease, habitat degradation, and predation. Oyster aquaculture, however, has been shown to successfully cultivate native oysters in high densities. Through culturing activities, new oyster aquaculture facilities have the ability to remove nutrients directly from the Bay. Nutrients entering the Bay are energy sources for phytoplankton and algal growth. These organisms, in turn, are the

primary food source for oysters. A portion of the nutrients contained in the phytoplankton becomes part of the oyster and through oysters harvest; nutrients are removed from the Bay as oyster meat and shell. In addition, a portion of the nitrogen within oyster biodeposits may be removed from the ambient water through nitrification and denitrification processes at the sediment-water interface. Nutrient assimilation services can also be enhanced through the restoration or creation of forested riparian wetlands and through biomass harvesting (algal harvest for example).

Incentives to make investments in nutrient assimilative services could possibly be created through nutrient assimilation credits. Nutrient assimilative credits are the public acknowledgement and certification of nutrient assimilation services. Nutrient assimilative credits are defined as the total amount of nitrogen or phosphorus removed from ambient waters within a given time period. Nutrient assimilative credits, however, are not formally part of state and federal efforts to improve water quality conditions in the Chesapeake Bay. Current efforts aim at reducing point and non-point source discharges by providing public subsidy or cost-share programs to install new equipment (ex. capital upgrades to wastewater treatment plants) or specific best management practices (e.g. agricultural practices such as riparian buffers, cover crops, and conservation tillage). Regulatory programs focus on placing stringent mass nutrient load caps on point source discharges. Nutrient trading programs are now being implemented to provide point sources compliance strategies to stay under these caps in the face of growth. Financial incentives to produce nutrient assimilation credits could potentially be created through revisions within state cost share programs or regional nutrient trading programs.

Oyster aquaculture is used to illustrate how nutrient credits can be created and whether nutrient assimilation credits could potentially be provided at a cost comparable to source reduction strategies. Oyster aquaculture offers a way to successfully cultivate oysters in the Bay, but only at a higher cost than traditional oyster harvest. The opportunity to produce nutrient assimilation credits would provide new incentives to expand oyster production. Attention will focus on how the cost of nutrient removal can be computed and compared to nutrient removal costs from point and non-point source reductions.

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WATER QUALITY TRADING: AN EXAMINATION OF THE PENNSYLVANIA NUTRIENT TRADING PROGRAM

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EXTENDED ABSTRACT

Pennsylvania's Nutrient Trading Policy and Program were designed through an open dialog process and can be considered a model of successful collaborative government. Various stakeholders, including point sources, agricultural entities, environmental groups, businesses and leading experts on trading from NatSource and World Resources Institute worked together to develop a flexible trading platform for Pennsylvania's needs. The Nutrient Trading Program positions Pennsylvania to take significant steps in achieving cost-effective nutrient reductions while allowing local communities to make critical investments in infrastructure.

Initial trading efforts have focused on the Commonwealth's obligations regarding the Chesapeake Bay. For Pennsylvania, the challenge is formidable: yearly nitrogen, phosphorus and sediment discharges to the bay must be reduced to no more than 71.9 million pounds, 2.46 million pounds and 0.995 million tons, respectively. To achieve the targeted Chesapeake Bay reductions, Pennsylvania is establishing nutrient reduction goals that will require continued collaboration between farmers, point sources and others in the Basin. As a result, new strategies for improving water quality have been developed that are cost effective and receive wide support from the community.

Through the discussion, the audience will obtain an overview on the Nutrient Trading Program as it was developed and how it is enhancing water quality benefits in Pennsylvania. The discussion will touch on the need for trading including the identification of key stakeholders, as well as the drivers and barriers to trading. The discussion will also focus on some trading program specifics that make the program unique, and provide some examples of innovative uses of the trading program to accommodate local development.

Pennsylvania's program has empowered local communities to use nutrient trading to solve local issues, while providing long term benefits. Groups are seeing how effective nutrient trading can be in meeting environmental regulatory goals at less expense than traditional command and control regulations, and are eager to determine ways to be even more competitive. It is through innovations like water quality trading that will further contribute to Pennsylvania's already successful efforts to restore vital natural resources and make crucial investments in our local communities.



Session V-A

Below the Surface: Insights into Groundwater Resources

- Groundwater Ages in West Virginia
- Requirements Analysis for a Groundwater Assessment Information Management and Decision-making Tool: Case Study of Albemarle County, Virginia
- Making Sustainable Development Decisions in Albemarle County

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GROUNDWATER AGES IN WEST VIRGINIA

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EXTENDED ABSTRACT

Groundwater age is useful in predicting the susceptibility of ground water to contamination and in understanding fate and transport of contaminants from point and non-point sources. The U.S. Geological Survey determined the apparent ages of ground water in West Virginia by using chlorofluorocarbon (CFC) and tritium dating methods. Apparent age is a model-calculated approximation based on the measured concentrations of environmental tracers. Apparent ages are determined with consideration of chemical sorption, biodegradation, and physical mixing processes that can alter age interpretations. After the introduction of CFC gases as refrigerants in the late 1930s, atmospheric concentrations increased until production ceased in the mid-1990s. CFC dating is based on production records that date to the early 1940s and on the preservation of atmospheric CFC in groundwater recharge. During the 1950s and early 1960s, atmospheric testing of nuclear weapons raised concentrations of atmospheric tritium hundreds of times above natural levels. After atmospheric testing ceased in the 1960s, atmospheric tritium began decreasing and is now approaching natural levels. Tritium dating is based on records of atmospheric concentrations and the preservation of those concentrations in groundwater recharge. Tritium is used to qualitatively determine if ground water is younger or older than about 50 years.

Apparent CFC groundwater ages in West Virginia ranged from 5.9 to 56 years. The median apparent age of all samples was 19 years. Median apparent ages of ground water in the Appalachian Plateaus, Valley and Ridge, and Ohio River alluvium were 21, 13, and 20 years, respectively. The percent of young water was lowest in the Appalachian Plateaus samples while Ohio River alluvium samples were 100 percent young water.

In the Appalachian Plateaus, topographically driven groundwater flow is evident from comparing differences in median apparent ages of water from hilltop, hillside, and valley settings (12, 14, and 25 years, respectively). Topographic relations are not evident in the Valley and Ridge (median apparent ages of 18, 21, and 13 years for hilltop, hillside and valley settings) where bedding and geologic structure exert greater control on groundwater flow. Younger water in valley settings may be due to surface runoff flowing down hillsides and infiltrating through colluvium at valley margins. Tritium-derived groundwater ages and geochemical analysis of thermal gradients indicate deep groundwater circulation and long residence time (> 50 years) for structurally controlled thermal springs in the Valley and Ridge.

In coal-mined areas of the Appalachian Plateaus, groundwater ages are more homogenous (hilltop, hillside, and valley settings have median apparent ages of 35, 31, and 34 years in the northern high-sulfur coal region and 18, 26, and 26 years in the southern low-sulfur coal region). Similar ages in hillside and valley settings may be due to fracturing of rock by blasting during surface mining.

CFC contamination, identified by concentrations exceeding modern atmospheric levels, was most common in water samples from Ohio River alluvium and Valley and Ridge karst aquifers, indicating their vulnerability and possible contamination by domestic and industrial wastewater.

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REQUIREMENTS ANALYSIS FOR A GROUNDWATER ASSESSMENT INFORMATION MANAGEMENT AND DECISION-MAKING TOOL: CASE STUDY OF ALBEMARLE COUNTY, VIRGINIA

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KEY WORDS: requirements analysis, groundwater assessment, GIS, rural development

ABSTRACT

As cities like Charlottesville develop, they spread into surrounding rural areas, resulting in urban sprawl with increased dispersion of housing, groundwater consumption rates, and likelihood of contamination from septic drainfields in these areas without public water or sewer services. County decision-makers, who grant permits for new home sites, lack access to groundwater assessment (GWA) information about availability and quality of the resource. In order to plan for development that can be sustained by its groundwater capacity, county decision-makers require such regional GWA information. This will help them link decisions about regional development to groundwater availability and quality.

The goal of this work is to establish a list of requirements for a Decision-Support system that could be used by County officials and other stakeholders in planning residential development that affects rural areas. The goal was accomplished by; i) assessing the state-of-art of regional groundwater assessment and how this information is used by officials, ii) assessing the county groundwater-related planning decision-making process, iii) assessing and documenting the state-of-art in groundwater information management, including use of the internet and GIS to facilitate collaborative decision-making by the multiple stakeholders with an interest in regional development, iv) transform all gathered information into system requirements for a AGWA Decision-support system (DSS).

INTRODUCTION

The result of this urban sprawl is increasing housing densities, groundwater consumption rates, and likelihood of contamination from septic drainfields in rural areas without public water or sewer infrastructure. Between 1983 and 1998, there was a 20% increase in the number of people relying on groundwater for water supply in Charlottesville. This combination of factors is cause for concern about the quality of groundwater in rural areas. The fractured igneous and metamorphic bedrock geology of Albemarle County increases the difficulty of developing a predictive regional GWA methodology that can support this form of decision-making. The geometry and density of these bedrock fractures determine the amount of groundwater available, but the unpredictability of these fractures makes the possible yield difficult to evaluate from the surface. County officials are looking for ways to protect the community by introducing smart

growth practices and developing policies linking development planning to groundwater availability and quality (Louis, 2006).

However we know that “decisions are based on available information, so that correctness of a decision, in principle, depends on the availability and quality of relevant information” (Kukuric, 1999, p. 7). Several counties and states have developed Decision-Support Systems (DSS) to disseminate water data, groundwater assessment programs information and groundwater status in their states and counties. This work focused on finding all characteristics that such a tool should have in order to help county officials make effective groundwater-focused planning policies.

METHODS

Decision-Support Systems (DSSs) are the type of information systems expressly developed to support the decision-making process. To develop a DSS, we will be looking at all elements – data and models – to include in it as well as the scenarios for the user interactions with the system. For that, Requirements Engineering (RE) will be applied.

Requirements analysis is an important part of Requirements Engineering (RE), which is accepted as one of the most crucial stages in software design and development as it addresses the critical problem of designing the right software for the customer (Aurum and Wohlin, 2005). Leite and Doorn (2004) define Requirements Engineering (RE) as “a field of knowledge concerned with the systematic process of eliciting, modeling and analyzing requirements.” A requirement is a condition or capacity that must be met or processed by a system or system component to satisfy a contract, standard, specification, or other formally imposed documents (IEEE, 1998).

The common RE activities are elicitation, analyzing and documentation, negotiation, verification and validation (Aurum and Wohlin, 2005). Conceptually, the first three activities are part of what is called “Requirements analysis”.

The elicitation of requirements is the activity most often regarded as the first step of the RE process (Nuseibeh and Easterbrook, 2000). Aurum and Wohlin (2005) define requirements elicitation as “the process of seeking, uncovering, acquiring, and elaborating requirements for computer based systems”. In the literature, several techniques and approaches have been identified for requirements elicitation. These include interviews, task analysis, questionnaires, brainstorming and prototyping. In this work interviews were used to gather information. Interviews are the most traditional and commonly used technique for requirements elicitation. They provide an efficient way to collect large amounts of data quickly. For this project, we are going to conduct semi-structured interviews conducted using a predefined set of questions which can be changed or adapted to meet the respondent’s understanding, belief or even to prompt more details about a previously given answer. Unlike structured interviews, which are considered rigorous and limited, the semi-structured interviews are more flexible and advocate adapting each interview to the individual interviewed.

A set of questions was developed to gather information needed from the stakeholders. For the sake of simplicity the set was built like a questionnaire. It has 21 questions, including open-

ended, multiple choices and, Yes/No questions. The questionnaire is divided into the following five sections:

- **Groundwater information and planning decision-making:** This section seeks an understanding of the groundwater-related planning decision-making process (1 question and 7 follow-up questions).
- **Groundwater data Collection and Analysis:** This section addresses groundwater data needs of the DSS (5 questions and 16 follow-up questions).
- **Private well water information and decision-making:** This section addresses groundwater data needs by understanding private wells data management (4 questions and 5 follow-up questions).
- **Mapping groundwater information and DSS functionality:** This section concerns the necessary information about expected applications of the DSS (7 questions).
- **Private Wells and groundwater quality perception:** This section provides feedback about the current mechanisms that are used to assess and manage groundwater data in the county (1 question and 2 follow-up questions).

Identifying the complete list of stakeholders is next step of requirements elicitation. In this case, the principal stakeholders are groups of people who could provide data to the system, use directly the system and/or affect directly the system. Local government agencies – Virginia Department of Health (VDH), Virginia department of Environment Quality (VDEQ) or USGS – do implement groundwater assessment programs, so they could supple data to the system. County officials and staff will use the DSS in their planning decision-making process. Nine people were selected from these two groups.

Other secondary stakeholders like homeowners and property developers have to follow zoning and land-use county regulations. So if the DSS come to introduce any change of planning policies, they are the one who will be affected. But they won't be making any decisions. Three other people - not from the main groups above - were approached but were not formally interviewed; one was heard during a forum and two were contacted by phone. One was a driller and the other two are member of NGO dealing with groundwater and environmental matters in Albemarle County.

Between February 2007 and July 2007, interviews were conducted. As noted above, the interviewees are counties officials or staff and local government agencies employees. The county staff members, who will be the primary users of the DSS, were to provide all characteristics of their idea of the best system. The interviews were semi-structured interviews and almost all of them took place at the interviewee's place of employment. Each interview lasted between 35 and 60 minutes

In order to supplement the information gleaned from interviews, the study also reviewed groundwater regulations and ordinances, as well as official planning regulations to understand the interactions between different stakeholders. The information gathered from the interviews will be analyzed and converted into requirements for the Albemarle County groundwater DSS.

RESULTS AND LIMITATIONS

Interview findings

The county officials indicated that their first priority about any DSS is for it to be a visualization tool, which would provide them with an overview of groundwater status in the county. The staff emphasized that there is a need of a visualization system with basic analysis on the available data such as basic statistics (averages, minimum, maximum). Such visualization tool will be used to inform the public and home developers of the groundwater state in the county.

Because these decision-makers are very interested in the visual aspect of the DSS, they selected some other data which could provide more information about groundwater in the county, among other we have topographic data (relief (DEM)), Tax Map Parcel information, School locations, Public water systems information, Hydrologic data (rivers, lakes, reservoirs), Geological data (soils, rocks families), Streets/roads and Existing structures. The former groundwater manager proposed that the DSS should have contour maps. About the reporting qualities of the system, the staff proposed that the DSS could have graphs and tables displays of the data because the board of supervisors prefers them to technical reports.

For analysis and environmental models were suggested. Basic statistics of the data (average well depth, yield, natural flow) such as trends or average should be displayed. Studies should be completed and analyzed such as the correlation between age and depth – with the hope to establish what area and which wells are more likely to be affected by drought -, the correlation between well depth, yield and drought, the distribution and density of wells. The staff and officials would like to have some groundwater information like hardness or temperature. The hope of these studies and analyses on the data is to provide some information about the groundwater carrying capacity of the region.

Through county water and planning regulations review, there are some of the analyses that are currently envisioned and could be introduced in the DSS. The comprehensive plan suggests that the mapping of groundwater vulnerability is needed. The county groundwater monitoring program is conducting a correlation analysis between water levels and precipitations. A hydro-geologic analysis was conducted in 1996 to study problems related to groundwater. That project looked at the correlation between geologic formation in the county and well yield (Hostettler, 1996). The models used for the analyses above would be added into the DSS.

Another aspect of the DSS that was emphasized was the data management functionality. The actual database is only relatively useful, the county would want a system which could manage current data, future data and help maintain the accuracy as well as the quality of the data. In short, a system should sustain any GWA data gathering program implemented in the county. The staff member insisted that the more pressing need for the department of community development is data; this makes the data management aspect of the DSS very important.

Requirements

These requirements describe the expected attributes and capabilities of the system as a whole and do not attempt to allocate capabilities to specific modules or subsystems within the *AGWA DSS*. The high-level requirements provide a basis for components in system elaboration, and detailed requirements are subsequently tied to specific components. Converting the desires of the users into requirements, an overview of the High-level requirements of the AGWA system is presented in the figure 1 below.

This DSS is primarily expected to be a visualization tool, which will provide an overview of the county groundwater quality and availability status to users. It is also expected to collect, store and disseminate groundwater information.

The data management subsystem of this DSS will collect, store, and organize the groundwater data as well as other complementary data and deliver all possible pieces of information that can be obtained from them through GIS maps. Among other information, the system should be able to present basic statistics (average, median, standard deviation) of groundwater data and provide an easy way to understand them.

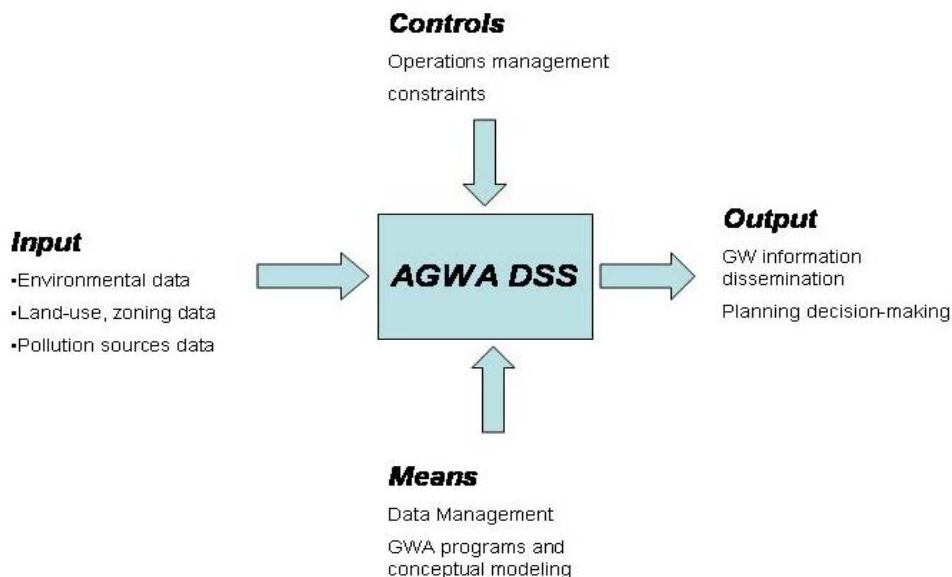


Figure 1. High level requirements for the AGWA DSS

All these functions will be managed using web-based Geographic Information Systems (GIS). GIS software serves as a database management tool that allows multiple types of data to be accessed and integrated for both analytical and visual analysis. The powerful visualization properties of GIS software are what had made GIS the most used software in environmental data management over the last few years. For the visualization capacities, the GWA information should be displayed on GIS maps. The user should have the option to view any attributes of GW information that he wishes. The GIS maps should have the option to be queried by location or attributes. The system should have other display options such as tables and/or graphs. The search options of the system should present graphical as well as tabular results.

The DSS model management subsystem will contain many computational models and analyses. Some of the models will respond to some the objectives detailed on the comprehensive plan regarding groundwater protection. That is, mapping groundwater vulnerability will be one of the capacities of the DSS.

The data will be entered – either manually or by uploading a geodatabase - in the system only by the county staff; thus the county staff will be the system administrator of the DSS. The AGWA system will collect data entered by the county staff and the data will be displayed on the online GIS maps. The system will also automatically perform data quality operations such as verifying that each data well record has accurate geographic location coordinates – that is, location in North America and in Albemarle County. To achieve the automation of these quality tasks, the system will have the capacity to receive a piece of information and check if it is valid according certain pre-entered conditions.

The system should be flexible enough to accommodate conceptual models to analyze data and publish resulting maps on the GIS map on the website.

The system shall have the capacity to generate reports to appropriate users – board of supervisors or planning commission members. Thus the system should have security features discriminating which types of operations every kind of users should perform. The maps should be well-formed and usable.

The AGWA system should be a data collection system capable of handling a vast range of data in a flexible manner that permits new data types to be added. The system should be able to manage different groups of users. However the county staff will be the only users with the capacity to enter data into the system as well as administer the system. The system should provide an interface for the system administrator for data management functions. From this interface the staff would enter information about newly drilled wells and update the well database. This interface should be easy to use because the staff member entering data might not be a computer savvy.

Determining data types will be a significant challenge. Proper understanding of the available data versus the required information will dictate how the data gathering processes and the database itself should be designed for greatest efficiency. Figure 2 below presents an overview of the AGWA system with its users, data and processes.

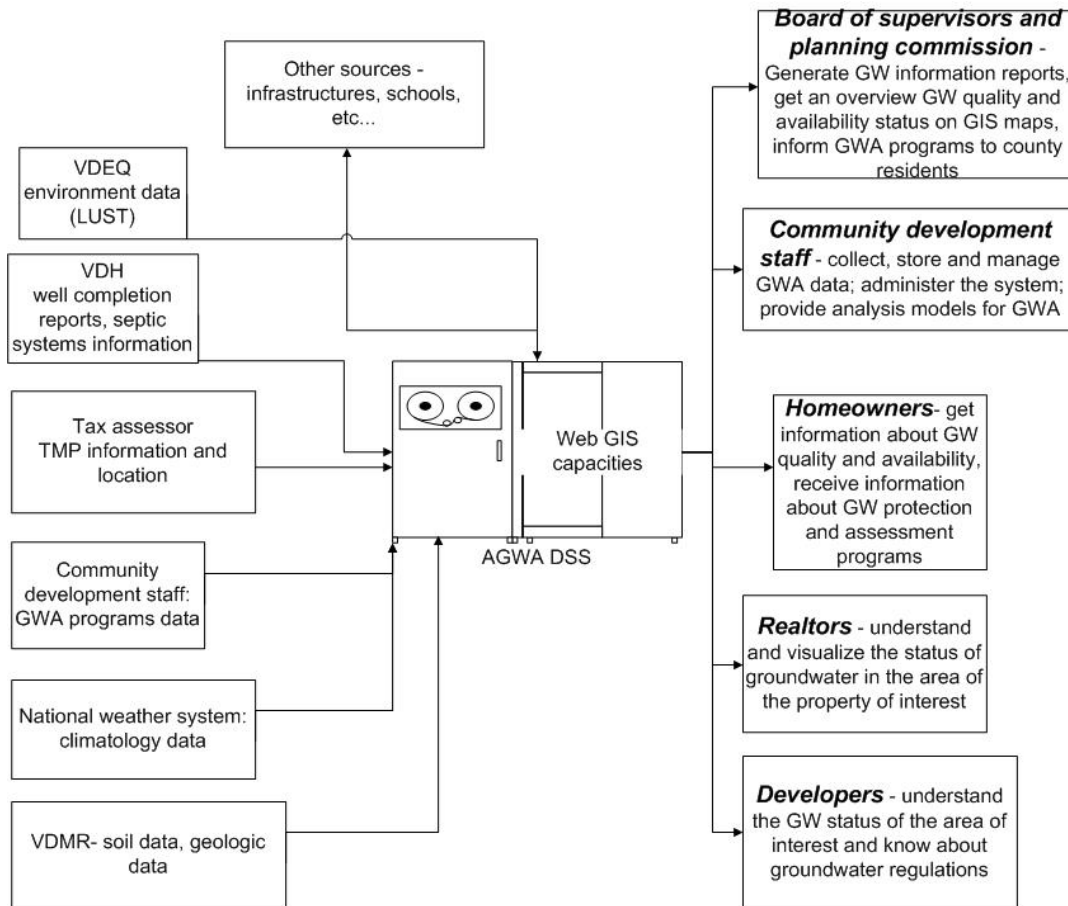


Figure 2. Overview of the AGWA DSS with its users, data and processes

The groundwater data accuracy was an issue; the current county database only has data from wells drilled before 2000. It also contained about 3% of duplicates and crucial information such as well depth, yield or static water level are missing. Different agencies datasets were not compatible. The VDH data – which has data for wells drilled after 2000 - could not be blended with the county database due to the lack of any common field. The use of GIS demands geo-reference information, which are missing; the VDH data does not have location information at all. In addition to that, none of these datasets have groundwater quality data, which is important to the development of some analysis models described above.

This problem revealed the need for the implementation of GWA programs in the county. The single water measurements program currently implemented is not enough; there is need for more such as up-to-date and frequent quality data and descriptive information – hardness, temperature. During the interviews, it was noticed that the lack of data brought serious limitations on the expected features of the system because the interviewees were not able to pass that issue and visualize a “perfect” solution to their problem. Furthermore many of the officials are not computer savvy and have never used an environmental DSS.

CONCLUSIONS

The DSS described by these requirements is a Data Analysis System, that is, help analyze historical and current data, either on demand or periodically, since it will contain only simple models for data analysis. The hope of this first system is that it could evolve with the availability of data to become a Focused-Oriented Data Analysis system, where the knowledge of the projected development could forecast its effect on groundwater resources. The county officials would have a tool to help them in the development of better and stronger water management and protection policies.

ACKNOWLEDGEMENTS

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MAKING SUSTAINABLE DEVELOPMENT DECISIONS IN ALBEMARLE COUNTY

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KEY WORDS: groundwater assessment; GIS mapping, economic valuation, decision making

ABSTRACT

Groundwater is an important resource for rural areas like the Ivy subdivision of Albemarle County. The availability and quality of this resource is increasingly threatened by population growth and associated urban sprawl activities. Realizing the need to consider the sustainable use of groundwater resources in development planning, the goal of this paper is to demonstrate the value of using Geographic Information System (GIS)-based information for making sustainable development decisions at the county level. This goal is achieved through the following objectives that demonstrate the ability of decision-makers: (1) to predict future water-stressed areas; and (2) to calculate and compare the cost of using private wells against the cost of extending the public drinking water supply infrastructure into the Ivy subdivision of Albemarle County. The study results showed that projected water-stressed areas would occur first in areas near the City of Charlottesville and then in the northern part of the County. In addition, based on current capital, operational, and maintenance costs, it is more economical to continue to use existing private wells rather than extending the public drinking water supply infrastructure into the Ivy subdivision of the County. This paper presents research efforts that are part of the Albemarle County Groundwater Assessment Project located at the following website: www.sys.virginia.edu/research/environment/albemarlegroundwater/.

INTRODUCTION

The conversion of farms and forests to residential and commercial properties has significant adverse impacts on the environment, the natural resources and the ecology. This land conversion is occurring at an accelerated pace in many parts of Virginia. Usually, properties developed in rural areas do not have access to public water supply and sewerage so they must draw on private groundwater wells for their water supply, and use septic tanks with drain fields for their sewage.

According to the Virginia Department of Environmental Quality (DEQ, 2006), 55 of the 95 counties in Virginia obtain over half of their drinking water supply from groundwater resources. Among households using private wells, 92% also use septic tanks, 31% live on lots that are smaller than an acre, and 35% use fuel oil as primary heating source. This combination of factors raises concern about the availability and quality of groundwater resources in rural areas of Albemarle County. However, little is known about the County's groundwater resources. The County is underlaid by metamorphic rock fracture geology, which makes it difficult to develop a

comprehensive groundwater assessment (GWA) methodology for sustainable development. Therefore, County decision-makers, who grant permits for new developments, currently lack sufficient GWA information to minimize the impact of new developments on groundwater resources. This lack of GWA information results in adverse impacts from urban sprawl activities, which are characterised by increasing housing densities, groundwater consumption rates, and risk of aquifer contamination.

This paper presents the research efforts completed by a group of 4th year undergraduate students. Their research efforts extend the research efforts from the 4th year undergraduate student teams of the two previous years. The goal of this paper is to demonstrate the value of using Geographic Information System (GIS)-based information for making sustainable development decisions at the county level. This goal is achieved through the following objectives that demonstrate the ability of decision-makers: (1) to predict future water-stressed areas; and (2) to calculate and compare the cost of using private wells against the cost of extending the public drinking water supply infrastructure into the Ivy subdivision of Albemarle County.

METHODOLOGY

Objective 1 is achieved by demonstrating the ability of decision-makers to identify future water-stressed areas in the County. The proponents of this research collected census data, number of public water connections, and average per capita water usage rate. The water usage rate was computed as the product of the number of public water connections, the average household size, and the per capita water usage rate. The FORECAST () function of Microsoft Excel was used to compute projected values. Finally, the pre-processed data were then imported into GIS software to create water-stressed area maps for the County over time.

Objective 2 is achieved by demonstrating the ability of decision-makers to compare the cost of continuing to use the existing private wells within the Ivy subdivision versus the cost of extending public drinking water supply infrastructure into this area. The cost functions consist of current capital, operation and maintenance expenses. The cost function for a typical private well within the Ivy subdivision was verified through interviews with a local well driller. The cost function for a typical private well includes capital costs for constructing a typical private well as well as annual operation and maintenance costs for average values of pump rating, electricity cost, pump expected life, and pump replacement factors. The cost function for extending the public drinking water supply infrastructure within the Ivy subdivision was verified through interviews with the Albemarle County, the Albemarle County Service Authority (ACSA). The cost function for the ACSA infrastructure includes capital costs for extending new water supply mains and individual household connections as well as annual production costs for abstraction, storage, treatment, and conveyance factors.

Data for the above research efforts were verified through interviews with applicable Albemarle County departments, the ACSA, the Dominion Power Company, the Thomas Jefferson PDC, a local well drilling company, and a residential property developer. Historical and in-situ GWA information for the Ivy subdivision had been collected and analyzed through research efforts by previous 4th year undergraduate student teams.

RESULTS AND DISCUSSION

The growth rates of the County's population and the number of public water supply connections are assumed to be constant throughout the analysis period. The average household size for the County which has the value of 2.44, and the per capita consumption rate which is 100 gallons per day, were also assumed to be constant throughout the analysis period. With these assumptions, **Figure 1** shows the projection of the number of public water connections through 2026 and **Figure 2** shows the projection of the public water usage rates through 2026.

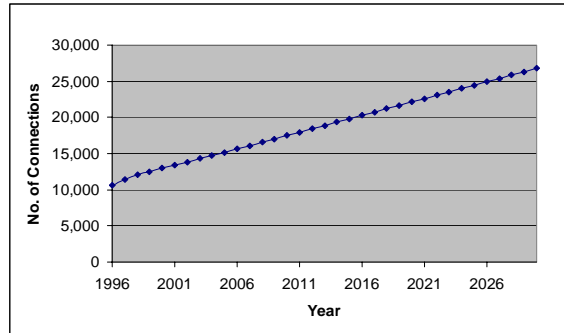


Figure 1. Trend and projection on number of water connections
Source: Albemarle County, 2006

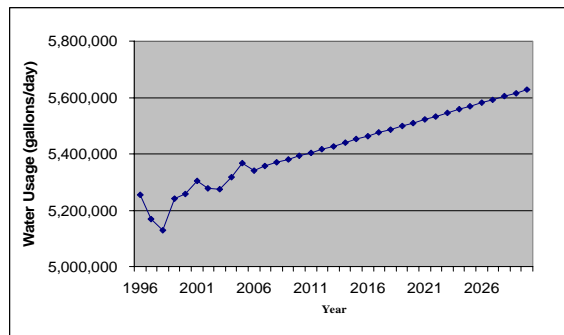


Figure 2. Trends and projection on water usage
Source: Albemarle County, 2006

The above projections provide the basis for identifying future water-stressed areas within the County. **Figure 3** below shows how water usage is expected to increase throughout the County. Intuitively, water-stressed areas will be highest in the areas near the City of Charlottesville and some areas in the northern part of the County. These GIS-based maps can be used by decision-makers to determine areas that are likely to experience groundwater resource stress in the future.

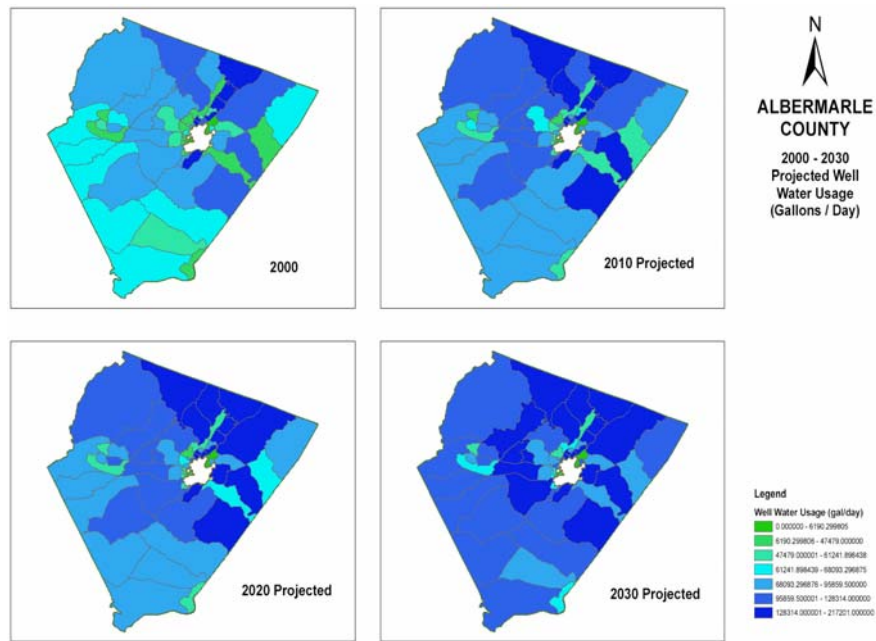


Figure 3. Projected water-stressed area maps

The cost of shifting away from existing groundwater resources, i.e. private wells, in projected water-stressed areas of Albemarle County is dependent on the results of a comparative cost analysis relative to extending the public drinking water supply infrastructure into the area. The current cost function for a typical private well was developed using values supplied by a local well driller from the County. The average depth of a private well in the County is 300 feet and costs \$9/ft to drill for a total drilling cost of \$2,700. A well casing is also necessary to provide structural integrity to the well near the surface. The usual casing depth is 50 feet, and costs \$8/ft for a total casing cost of \$400. Additionally, well grouting is applied and costs about \$750. A well pump is roughly \$850 plus an installation cost of about \$1,650. Thus, the average capital cost for a typical private well is about \$6,350. Meanwhile, the operation and maintenance cost for a typical well includes electrical consumption and pump replacement cost. Most pumps have ratings of 1 to 1.5 horsepower, which is roughly equivalent to operating costs of about \$1/day or \$365/year. In addition, the pump has an expected life span of 10 years. With pump replacement at \$850 and using a current 10-year T-bill of 4.61% (www.bloomberg.com), pump replacement costs is about \$69/year. Thus, the average operation and maintenance cost for a typical private well is about \$434/year.

The cost function for providing public water to the Ivy subdivision includes the average capital cost for extending the drinking water supply piping network. A residential property developer within the County estimates that the current capital costs associated for extending the public drinking water mains is about \$100/ft. The approximate distance between the current public water infrastructure and streets within the Ivy subdivision was estimated using GIS software, as shown in **Table 1**.

Table 1. GIS estimated distance between current public water infrastructure and neighborhoods in the study area

<i>Street</i>	<i>Distance (ft)</i>	<i>Street</i>	<i>Distance (ft)</i>
Morgantown	11,134	Layton St.	1,734
Mountain Laurel	1,332	Ivy Depot Rd	2,426
Grassmere	7,072	Ivy Depot Ln	335
Dry Bridge	4,801	Skyline Crest	1,299
Country Woods	1,215	+ <i>Dist to Main</i>	2,000
TOTAL	33,348		

Summing the street distances and multiplying by \$100/ft gives an average capital cost for extending the public drinking water infrastructure of about \$3,334,800. Since there are 146 houses in the Ivy subdivision, this translates into \$22,841 per household. Additionally, the ACSA charges \$6,583 per household to connect to the mains, thus the average total capital cost for using public drinking water is \$29,424 per household.

Table 2: Public drinking water price structure (ACSA, 2006)

<i>Usage (gal/month)</i>	<i>Price Structure (\$/thousand gal)</i>
0 – 3000	3.37
3001 - 6000	4.04
6001+	7.22

The household cost for using public drinking water is shown in **Table 2**. ACSA’s current drinking water rate structure is inclusive of operational and maintenance costs for producing, treating, and delivering public drinking water to a household. ACSA employs an ascending price structure, which encourages water conservation (Aden, 2007). As previously stated, the study assumes that an average household within the County consists of 2.44 persons and consumes 100 gallons per day per capita. The daily household consumption rate is equal to approximately 7,500 gallons of water per month. Each household will then pay approximately \$38/month or \$456/year.

Given the above cost functions, **Table 3** shows that it is more cost-effective, at least in the near term, to continue relying on groundwater resources within the Ivy subdivision than to consume public drinking water supplies. Moreover, because of new regulations and laws, the price of public water is expected to increase by 12% each year for the next five to ten years (Lynn, 2007). On the other hand, the price of electricity, which is the main operating cost for a private well, is expected to increase by only 4% each year (Dominion Power, 2007).

Table 3 Cost comparison of household drinking water supply alternatives

	Private Well	Public Drinking Water
Capital Costs (\$)	6,350	29,424
O&M Costs (\$)	434	456

Therefore, the goal to demonstrate the value of GIS-based information for making sustainable development decisions at the county level is achieved. However, the models that were used to demonstrate the ability of decision-makers to predict future water-stressed areas and to compare the cost of shifting from private wells to the public drinking water supply are limited by their respective assumptions. Actual cost comparisons will need very accurate estimates for future groundwater abstraction rates and the amount of resource available for consumption. The overall research efforts of the Albemarle County Groundwater Assessment Project look to providing answers to these complex sustainability questions.

CONCLUSION

Groundwater is an important resource for rural areas within Albemarle County. This study demonstrates how GIS-based information can be used as a tool for county-level decision making that considers the sustainable use of groundwater resources. Given the current population and service connection growth rates, it was shown how the water usage rate can be expected to increase. GIS-based information was used to create maps that show decision-makers the projected water-stressed areas within the County over time. Also, with the help of GIS-based information, it was shown that based on current cost functions containing capital, operational and maintenance factors, it is more cost-effective to continue to use existing private wells for the Ivy subdivision than to switch to public drinking water supplies.

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Session V-B

Managing Aquaculture Production Systems

- Freshwater Aquaculture and Farm Pond Management in Virginia
- A Cost-Benefit Analysis of an Alternative Raceway Material
- The Effects of Pollution Reduction on a Wild Trout

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FRESHWATER AQUACULTURE AND FARM POND MANAGEMENT IN VIRGINIA

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KEY WORDS: freshwater aquaculture, farm pond management, cage culture

ABSTRACT

The Virginia Agricultural Statistical Service conducted four surveys (1993 – 2003) to follow industry developments relevant to the economic value of the state's aquaculture industry. Survey results demonstrate significant changes occurred in production and sales among the principal freshwater aquaculture species. Production of all species, except for tilapia, declined over this 10-year period and continues to remain low. Aquaculture research emphasizing farm ponds in the Central Piedmont Region of Virginia has been conducted at Virginia State University since 1985, primarily on the culture of channel catfish, rainbow trout, and hybrid striped bass. Studies have focused on the use of plastic-mesh, floating cages to rear fish in confinement similar to the small-farm practice of growing rabbits or chickens for home use and niche market sales. Also, a large number of production pond experiments have been carried out with catfish, hybrid striped bass, and other warm water companion species. Research and industry results have shown that winter cage culture of rainbow trout, golden trout, and brook trout is biologically feasible, and channel catfish can be reared in cages from spring to fall. Pond production of catfish requires attention to special management conditions that can limit profitability if not understood and acted on accordingly. Hybrid striped bass have not proven to be a viable aquaculture candidate for pond or cage culture in Virginia farm ponds. Small-scale fish culture in farm ponds requires fisheries biology knowledge of all species involved and management of pond characteristics, as well as keen economic skills, in order to be to be successful.

INTRODUCTION

Government agency reports on the existence of over 80,000 farm ponds provided the initial stimulus for development of a small-scale aquaculture industry in the Commonwealth of Virginia. The presence of many multi-purpose ponds suggested potential opportunities for small farm diversification. Research has demonstrated the biological feasibility of catfish and trout cage culture in farm ponds. However, many factors must come together for establishment of profitable commercial operations in farm pond situations. Freshwater aquaculture in Virginia has not, as yet, demonstrated commercial profitability.

Prior to the mid-1980s, Virginia freshwater aquaculture production was limited primarily to rainbow trout (*Oncorhynchus mykiss*). Beginning in the late-1980s and into the 1990s, a period of growth and development with warm water aquaculture began as a result of research and extension program emphases at Virginia State University (VSU) and Virginia Polytechnic Institute and State University (VPI). State funding for aquaculture activities in the Commonwealth commenced with the “Aquaculture Initiative” enacted by the Virginia General Assembly in 1987. Hybrid striped bass (*Morone saxatilis* x *M. chrysops*) and channel catfish (*Ictalurus punctatus*) were the primary species studied for commercial production in farm pond, cage culture, and indoor tank operations. Since the onset of initiative funding, aquaculture research has been conducted at Virginia’s land grant universities and cooperatively with other institutions, industry, producer associations, and government agencies.

An Aquaculture Plan for the Commonwealth of Virginia was developed by the Virginia Department of Agriculture and Consumer Services (VDACS) beginning in 1993 (Newton 1995). As a result of activities associated with the Aquaculture Plan, the Virginia Agricultural Statistical Service (VASS) conducted the first survey in 1993. Goals of this comprehensive survey were to determine the status and to establish production benchmarks for both freshwater and marine aquaculture industries. Three more surveys were conducted to document changes in industry growth and value. These surveys summarized industry data based on producer production and sales information. These survey reports were used to make comparisons and interpretations as they related to changes that occurred in freshwater aquaculture (Newton 2006).

METHODS

Virginia aquaculture surveys were conducted by VASS in 1993, 1995, 1997, and 2003. The formats for all four surveys were similar and were developed by Jim Lawson, Deputy State Statistician, VASS, and the author in conjunction with the development of the State Aquaculture Plan. All industry data collected for these surveys are protected by confidentiality regulations established by the U.S. Department of Agriculture (USDA) and the National Agricultural Statistics Service (NASS) (USDA regulations, Title 7, Chapter 55, Section 2276). Survey reports are available from the Virginia State Statistician, VASS, Richmond, Virginia. The Virginia Department of Agriculture has led in the development of the Virginia aquaculture industry; however, regulation of the industry is the responsibility of state conservation agencies. The Virginia Department of Game and Inland Fisheries is the regulatory agency that issues Aquaculture Permits, and the Virginia Marine Resources Commission issues permits to allow stocking of hybrid striped bass. Ponds with hybrid striped bass cannot discharge into streams

leading to the Chesapeake Bay. Permits issued by these two agencies are required in order to sell fish reared in Virginia.

Farm ponds differ in a number of physical, chemical, and biological characteristics that determine suitability for aquaculture applications. Aquaculture aspects of farm ponds are based upon research studies conducted and published by the author, along with forty years of first-hand experience working with farm ponds in many states of the southern region of the United States. At Virginia State University (VSU), research investigations and ancillary observations on the feasibility and utilization of Virginia farm ponds for aquaculture production have been conducted since 1985.

DISCUSSION

Freshwater aquaculture industry status

Virginia aquaculture industry overview

VASS tracked the Virginia aquaculture industry from 1993 through 2003. Each survey report provided information on both marine and freshwater industry sales. A review of freshwater aquaculture sales from 1993 to 2003 revealed the greatest increase in sales occurred soon after passage of the 1992 Virginia Aquaculture Development Act (Newton 2006). The Aquaculture Development Act also established a Governor appointed Aquaculture Advisory Board. The Virginia Aquaculture Plan was developed beginning in 1993 and was published in 1995 by VDACS. Also, several industry producer associations were formed during the early 1990s. These activities had a positive impact upon aquaculture awareness and encouraged significant expansion of aquaculture during the mid-1990s (Newton 1995). These positive impacts did not last long, however. Two USDA Census of Aquaculture conducted in 1998 and in 2005 showed a marked decline in the Virginia freshwater aquaculture industry (NASS 2000, 2006). Farms reporting aquaculture activities declined over 50% in number, from 62 farms to 28 farms, and acreage, from 324 acres to 143 acres (NASS 2006). Tilapia reared in recirculating systems increased from a small beginning in the early 1990's to about 75% of total gross sales in 2005.

Gross sales of principal freshwater species

The principal freshwater species utilized for aquaculture in Virginia as documented by the VASS surveys are channel catfish, hybrid striped bass, rainbow trout, and tilapia. Survey results for Virginia aquaculture, summarized below, are discussed in detail by Newton (2006).

Catfish production was consistently low in both state and national surveys. Percentage of total freshwater sales of catfish declined from a low level of 1.2% in 1993 to 0.4% in 2003. The industry value of catfish in 1993 was \$24,000, rose to a high of \$36,000 in 1997, and declined again to \$21,000 in 2003. The 2005 USDA Census reported only \$9,000 in Virginia catfish sales (NASS 2006).

Hybrid striped bass culture has not developed as hoped in Virginia. Hybrid striped bass had its largest sales year in 1995, with \$42,000, while total sales in 2003 dropped to \$29,000.

Percentage of total freshwater sales ranged from 1.3% in 1993 to 0.5% in 2003. In contrast, the North Carolina hybrid striped bass industry value was \$7 million in 2004. The primary difference between Virginia and North Carolina hybrid bass production is related to water quality. Virginia farm ponds are typically characterized by low alkalinity and hardness (soft water), whereas the North Carolina aquaculture industry relies on the Castle-Haynes aquifer to produce hybrid striped bass. This aquifer is an excellent ground water source characterized by high mineral content and low salinity.

Rainbow trout culture is the oldest established freshwater aquaculture industry in Virginia and the United States. While Rainbow trout is the principal cultured species, brook trout (*Salvelinus fontinalis*), the only trout native to Virginia, brown trout (*Salmo trutta*), and golden rainbow trout (*O. mykiss aquabonita*) (Nelson *et al.* 2004) are also produced in Virginia. From the late 1980s into the late-1990s, gross sales of freshwater trout were steady at about \$2.3 million annually. Annual sales reported for 2003 were down one million dollars from previous report years. Over the survey period (1993-2003), the number of trout producers declined due to increased regulations and lack of water (sites and drought). Aquaculture statistics show that total freshwater production doubled in size from 1993 to 2003; however, trout sales decreased by over \$1 million and the increase was due to tilapia sales going from \$27,000 to over \$4 million (Newton 2006). The trout industry contribution is still large when compared with the combined sales of hybrid bass and catfish at only \$50,000 in 2003.

Tilapia and rainbow trout traded places with regard to percentage and volume of freshwater sales in Virginia. In North Carolina, where tilapia may be reared in ponds as well as indoor facilities, sales were about \$3 million in 2004. In Virginia, tilapia production is primarily by a single, large producer. Presently, tilapia is the number one freshwater aquaculture species produced and sold in the Commonwealth of Virginia (NASS 2006).

Alternative aquaculture species

There are several fishes that may be produced either as alternatives to or in conjunction with the principal aquaculture species in certain situations. These fishes may be grown specifically for specialty markets or in polyculture combinations. Although some species may perform satisfactorily under certain conditions, their main contribution will be through increased market diversity, not as a basis for industry establishment. Some possibilities for live sales include largemouth bass (*Micropterus salmoides*), sunfishes (*Lepomis* sp.), other game fishes, koi carp, and other species designated by the permitting agencies. However, tropical and exotic species, such as prawns and ornamental aquaria fishes, are not recommended for novice and recreational producers in order to avoid disappointment and monetary losses. Research has shown prawn culture to have low commercial potential and very limited marketing appeal (Tidwell *et al.* 2002, D. Miller, *pers. comm.*, Davis College of Agriculture, Forestry, and Consumer Sciences, West Virginia University, 2007). An economic analysis of freshwater prawn culture in Kentucky (Tidwell *et al.* 2002) reports a break-even price of \$5,056 per acre required to cover fixed and variable costs. An analysis for prawn culture in Virginia (Nerrie and Prior 2000) reported “returns to land and management varied from \$400 to \$900 per acre based on survival.”

Aquaculture management aspects of farm ponds

Multiple use farm ponds

Regardless of size or location, most farm ponds serve several purposes. Generally, the primary uses are for livestock watering or irrigation. Both of these uses conflict with aquaculture because cattle can cause erosion, muddy water, and deteriorated water quality, while irrigation may remove water from a pond at times when it is also important to aquaculture activities.

Depending on pond size, recreational activities, such as swimming, boating, or fishing, occurring near aquaculture cages could disturb and stress caged fish potentially causing reduced growth, disease, or death. Thus, in order to overlay an aquaculture operation into an existing farm pond situation, there must be management compromises along with changes in some activities that may have an adverse impact upon fish production.

Farm pond location

The main aspects of pond location are related to pond accessibility, production security, and marketing. Ponds used for aquaculture should be accessible by a gravel road for year round access regardless of weather conditions and should have electrical service available for pumps and aerators, lighting for night work, and security. Pond location is important for daily access to feed fish and tend the operation, and it is important is for distribution of harvested fish or pick-up by customers. Travel distance to local or regional markets, processing plants, and restaurants is economically important to both small and large market sales. A comprehensive overview of marketing aspects for small-scale producers was published recently in *Aquaculture Magazine* (Quagranie, 2007).

Physical characteristics

Farm ponds are very diverse in size, shape, and related physical characteristics. However, there are some common factors relevant to ponds used for aquaculture operations. Farm pond size should be at least 2 surface acres of water. At least one-third of the pond should have a depth of six to ten feet for storage capacity and for suitable placement of cages to promote water exchange through and under cages. There are cases whereby smaller ponds may be satisfactory, but they will have less production potential. And, regardless of size, larger ponds may be limited in relation to number and size of suitable areas for the aquaculture production of fish.

The most common characteristic of farm ponds is their irregular shape. Due to irregular shapes and uneven bottom slopes, usually with stumps or obstacles, few farm ponds are suitable for seining with nets to harvest fish. Additionally, there is great variation in the types and amounts of vegetation that grows naturally in farm ponds. Depending upon pond water fertility and other factors, vegetative growth may range from algae to floating aquatics to emergent plants along the shoreline and in shallow water areas.

An access pier that extends into deep water is essential for cage culture in farm ponds to secure cages for easy access for feeding and for harvesting fish. Otherwise, access to cages is restricted to a small boat, which can be dangerous when carrying feed, bulky equipment, or harvesting

cages. Accessing cages by boat also takes extra time when tending and feeding fish on a daily basis.

Water chemistry characteristics

Most farm ponds rely upon rainwater as their primary source of water. Ponds may be replenished by rainwater that flows across either pastures or forested lands. Water supplied to ponds from springs or by streams occurs in isolated situations. This is apparent from low pond water levels that occur during periods of drought. Surface water is less desirable for aquaculture operations because of the likelihood of introducing wild fish, diseases, and other water quality contaminants. Overall, ponds used for aquaculture are subject to the risks of an undependable supply as it relates to water volume and quality and of the possibility of contamination by surface pollutants. The main factor against large-scale pond aquaculture is the lack of a major ground water aquifer with appropriate water quality characteristics to supply pumping systems.

Direct rainfall, watershed runoff, and other water sources affect the water chemistry of farm ponds. In most cases, the alkalinity and calcium hardness of ponds is closely related to the surrounding land. Especially in Virginia's Piedmont Region, most ponds have very low alkalinity and hardness, usually less than 50 ppm, which greatly reduces pond productivity and fish yields. A standard recommendation is to take soil samples of the surrounding land, and possibly pond-bottom mud to determine the amount of mineral additives needed to enhance aquaculture yields. Nevertheless, the addition of one to two tons of agricultural lime per surface acre of water annually or bi-annually will increase alkalinity and, thereby, pond productivity. Lime is best added in the fall and does not require an even distribution because wave action will distribute the material from one or two locations. Gypsum is used to increase calcium hardness and is applied in similar amounts and manner as lime to achieve the desired results. An increase in alkalinity and calcium hardness enhances pond productivity due to better assimilation of fertilizers and feeds that go into ponds as a result of aquaculture.

Farm ponds have two types of water discharge structures, a primary standpipe overflow device used for maintaining the water level and a secondary emergency overflow channel on one side of the pond dam that allows rapid exit of surplus water around the pond dam. Excess water flushes through farm ponds and exits to a receiving stream or flows temporarily until it becomes land applied. Any time ponds overflow there is the risk of pond fish losses due to escapement and also entry of unwanted wild fishes can occur during high water. Ponds with hybrid striped bass must have containment structures to prevent the escape of these fish into streams that lead to the Chesapeake Bay. Additionally, pond effluent that enters into natural waterways is subject to regulation by state and local agencies and thereby would require appropriate permits.

CONCLUSIONS

Small-scale farm pond aquaculture has not proven to be a viable commercial alternative for Virginia farmers. Aquaculture surveys conducted from 1993 to 2003 reflect numerous changes in Virginia aquaculture. Currently, the principal freshwater aquaculture species ranked according to sales value are tilapia, rainbow trout, hybrid striped bass, and channel catfish. Tilapia, reared in indoor, recirculating systems, dominates as the number one freshwater cultured

species (over 75% of total gross sales) and remains in a growth phase for Virginia aquaculture. Rainbow trout sales have declined over 60% due to environmental and regulatory impacts. Small-scale farm pond and cage aquaculture operations with channel catfish, rainbow trout, and hybrid striped bass have not developed as anticipated in Virginia despite 20-plus years of effort. Farm ponds may support limited aquaculture production, but only if compatible with other uses. Consequently, efforts to establish profitable, freshwater aquaculture operations based on farm ponds and the present concept of the “small-scale food fish approach” should be reconsidered and restructured in light of observed industry changes and in consideration of practical farm-pond management strategies relevant to existing conditions in Virginia. The present aquaculture scenario is dominated by sales of live fish from out-of-state distributors for stocking farm ponds. Deliveries of these imported, live fish occur almost every month. Thus, one Virginia aquaculture marketing approach should be directed toward sales of live fish for pond stocking, and there are others that may be adaptable to specific circumstances. Promotion of projects lacking supportive research on biological and economic feasibility does not provide a solid foundation toward the commercial development of a viable aquaculture industry suited for the Commonwealth.

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A COST-BENEFIT ANALYSIS OF AN ALTERNATIVE RACEWAY MATERIAL

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KEY WORDS: “U” shaped plastic tank, aquaculture, economics

ABSTRACT

The costs of purchasing, installing and maintaining the growout facility may represent the biggest investment cost for an aquaculture operation. Therefore, alternative materials that have the potential to maintain or increase productivity while reducing costs can have a large impact on profitability. One such material is high density polyethylene (HDPE) plastic. The costs and benefits of purchasing, installing, and operating a 2000 gallon (7560 liter) plastic “U” shaped raceway in West Virginia were estimated and compared to a traditional flat bottom concrete raceway system of similar volume in Pennsylvania. The cost of installing a 2000 gallon (7560 liter) concrete system was estimated using recent quotes from local concrete tank manufacturers.

Preliminary results indicate that the plastic “U” shaped tanks are considerably less expensive to purchase, install, and operate than the comparable concrete system. This provides medium and small sized aquaculture operations with an adaptable product that lowers the cost of fish production, provides flexibility with design changes, and allows for a resale value due to the mobility of the lightweight tank. Other benefits include lower labor requirements for waste management. The solid removal manifold in the plastic tank allowed for efficient tank cleaning to occur, which ultimately helps lower operating costs and increase profitability.

The study has implications for small- and medium-scale aquaculture operations in West Virginia and surrounding states with similar resource endowments. The increased adoption of new materials such as HDPE will lead to increased production and statewide economic development benefits.

INTRODUCTION

Traditionally trout have been raised for stocking purposes in concrete rectangular raceways due to the labor efficiency, ease of handling or harvesting, and the ability to reuse the water (Boardman, Maillard, Nyland, Flick, & Libey, 1998). The large-scale production facilities,

operated by state and federal agencies are responsible for stocking large areas for public recreational fishing.

Within the past decade a number of public hatcheries have come under increased pressure to reduce waste discharges as well as to reduce the cost of production (Ewart, Hankins, & Bullock, 1995; Flemlin, Sugiura, & Ferraris, 2003; Hulbert, 2000). This pressure has resulted in the closing of some hatcheries, reduced production in other hatcheries (Hulbert, 2000; Westers, 2000), and the purchase of trout from private producers to reduce the cost of stocking public waters used for recreational fishing.

Private trout producers usually have smaller water sources than the larger public facilities and therefore have less production. As more states turn to private suppliers for stocking public streams for recreational fishing, there is a growing recreational market for private trout producers. This creates more demand for developing small to moderate flowing water sources suitable for trout production.

Due to the coal mining industry, West Virginia has dozens of biosecure free flowing water sources that can be used for small scale fish production (Jenkins *et al.* 1995). Economies of scale usually result in a higher cost of production for smaller producers, making it difficult to compete with larger operations. This study uses a small mine water discharge site to determine if there is a way for smaller producers to reduce their costs by using a “U” shaped plastic tank for fish production. The advantages of this new tank include lower purchase and installation costs, easy modifications, transportability, which allows for resale value, and reduced labor for cleaning. The disadvantages of the new tank include a life span that has yet to be determined, and like concrete tanks, if installation is not done properly, poor performance may result.

METHODS

In November of 2006 one group of 4 inch rainbow trout fingerlings were stocked into a series of 2,000 gallon “U” shaped plastic tanks at a density of about 4 fish per cubic foot. The plastic tanks were located in a remote area of southern West Virginia that had a reliable high quality mine water discharge. A chain link fence surrounded the tanks and three electrified wires around the fence were used to minimize external variables (wildlife intrusion). The cohorts of this group, from the same hatchery, were stocked at similar densities into a concrete flat bottomed tank of similar volume at a commercial trout hatchery. Both of these sites had a history of normal trout growth from previous production cycles.

Both groups were fed a high protein (42%) high fat (16%) commercial trout diet throughout the 31 week production cycle. Demand feeders were used at both sites and initially nylon netting was used to deter aerial predators. The cost of purchasing, installing, and cleaning the custom plastic tanks during the study was compared to the estimated cost of purchasing, installing, and cleaning precast concrete tanks as well as poured concrete tanks. Businesses that specialize in building concrete tanks provided recent quotes for this cost estimate.

The annual labor cost for cleaning the plastic and concrete tanks was determined by using an average of five cleanings taken over the last five months of the study. Growth data were

collected every six weeks from a random sample of at least 50 fish from both systems, using an Ohaus bench scale. As the trout approached marketable size, fin condition was recorded using a scale from 0 (perfect) to 5 (> 90% missing or eroded) for each of the 7 rayed fins. This meant that each fish had a score of between 0 and 35. A photographic key, developed by Hoyle (2007), for each of the fins, was used as a reference during the fin condition data collection.

Water quality was monitored in the plastic tanks using a YSI 600XLM sonde that recorded temperature, pH, oxygen and conductivity every hour. A YSI oxygen meter was used for temperature and oxygen readings from the concrete system. A certified analytical laboratory analyzed water samples from both sites for anions and cations in order to identify any parameters that were outside the accepted range for growing rainbow trout.

Financial, physical, and biological assumptions

Enterprise budgets require various assumptions to be made. The financial assumptions included an annual interest rate at 10% and annual depreciation of 5% on the investment. Biological assumptions include: a constant flow rate of 200 gallons per minute gravity flow for the 20K production scheme (one line of tanks); a three foot drop between each level of production; the concentration of un-ionized ammonia remains below 0.03 mg/l; and a stocking rate of 1000 fingerlings per tank with a 5% mortality. With 48 weeks of production per tank per year and an average growth rate of 2.1 grams per day, each tank will produce approximately 1,476 pounds per year.

RESULTS

Critical water quality parameters remained stable at both sites for the majority of this study. Water temperatures remained between 10 and 15 degrees Celsius at both sites. Water analysis from both sites showed that all measured parameters were within the tolerance range of trout. The water quality monitoring that was done at each site showed that the concrete tank had one low oxygen event during the last week in May (week 28), which resulted in the precautionary removal of 400 trout (40%) from the system. The plastic tanks had two low oxygen events in the lower tanks, one in May and one in June, due to the intrusion of a bear which managed to divert the water from the lower tanks. The upper two tanks were unaffected by this diversion of water.

Annual maintenance / Cleaning costs

Assuming labor costs of \$10/hr. and that cleaning occurs every 5 days (73 times per year), an average of 5 cleanings during the study resulted in 25 minutes per cleaning for 9 plastic tanks. This translates into 3.4 hours per tank per year or \$34/tank/year. The concrete tank used a pump to remove the solids from the flat bottom. The average cleaning of the settling zone (4 feet long by 4 feet wide) in this tank required 6.75 minutes. This translates into 8.2 hours per tank per year or \$82/tank/year (Table 1).

Table 1. Cost comparison for 2,000 gallon concrete and plastic tank system (10 tanks)

	Cost (\$)	Install / Prep.	Cleaning	Total Cost (\$)	% precast
Concrete precast	45,850	6,000	820	52,670	100%
Concrete poured	33,110	4,000	820	37,930	72%
Plastic (HDPE)	24,507	3,000	304	27,811	53%

Growth, fin condition, and mortality data are presented based on the average of the top two plastic tanks compared to the concrete tank. After 30 weeks the final weights were averaged from a random sample of at least 50 fish from the approximately 1000 fish stocked in each tank. The trout in the concrete tank showed better growth but higher mortality (2.20 gm / day growth and 5.6 % mortality) than those in the plastic tanks (1.75 gm./day growth and 3.96 % mortality, see Table 2).

Table 2. Growth, condition factor, mortality and fin condition from concrete and plastic tanks

	volume (m ³)	Growth rate Gm / day	Standard deviation (length)	Condition Factor (metric)	% mortality	Fin Condition
Concrete	7.84	2.20	2.73	0.0142	5.62	7.93
Plastic	7.57	1.75	2.15	0.0134	3.96	8.08

A one way ANOVA was performed on the fin condition data using total scores. When the trout from the two plastic tanks were compared to the trout in the concrete tank, the ANOVA procedure showed there was no significant difference ($\alpha < 0.05$) of fin erosion comparing the two trout populations.

Using information from the enterprise budget a capital budgeting (cost-benefit) analysis was conducted. The Net Present Value (NPV) is relatively large, which is a positive trait for investing in a project. Assuming a 10-year planning horizon, the Net Present Value is shown for three different interest rates (Table 3).

Table 3. Net Present Value (NPV) for three different interest rates over 10 years for 20,000 lbs. trout per year

Cost of Capital	NPV – 20,000 lbs./yr.
7%	\$103,005
10%	\$82,980
13%	\$66,614

The internal rate of return over the same 10 year period is shown in Table 4.

Table 4. Internal Rate of Return (IRR), 20,000 lbs. trout per year

Time Period	Internal Rate of Return
10 years	36%

Using the data collected from this research, enterprise budgets were developed for a representative small farm, using the new plastic tanks, assuming annual production of 20,000 pounds (Table 5). The enterprise budget included costs for a chain link fence and an emergency pump. The interest rates were set at 10% and depreciation for the initial investment was 5% per year.

Table 5. 20,000 pound per year trout farm - Plastic tanks

Enterprise Budget:				
Construction	Unit	price (\$) /unit	# units	Total \$
Site Preparation	dollar			3000
Water diversion	dollar			500
Plastic tank (2000 gallon)	tank	2,450	14	34300
Emergency pump / pipe		1,100	1	1100
Screens (1 per tank)	each	22	14	308
Chain link fence (option)	foot	20	1000	20000
sub-total				59208
Equipment				
Demand feeder (installed)	each	200	14	2800
Net , gloves, boots		250	1	250
sub-total				3050
Total initial investment				62258

	Unit	price (\$) /unit	# units	Total \$
Annual sales				
Recreational market	lb.	2.5	15,000	37500
Food market	lb.	1.5	5,000	7500
Total Sales	lb. or dollar		20,000	45000
Variable Costs				
fingerlings (3")	each	0.21	14,000	2940
Feed (FCR=1.2:1)	lb.	0.4	24,000	9600
Electricity	month	10	12	120
Labor (8 hours/week)	hour	10	416	4160
Interest on operating capital	dollar	0.1	16820	1682
Delivery Costs	mile	0.5	1000	500
Total Variable costs				19002
Fixed Costs				
Interest on Ave. Ann. Inv.	percent	10%	31129	3113
Property taxes	percent	2%	62258	1245

Land lease	\$/month	100	12	1200
Repairs and depreciation	percent	5%	62258	3113
Total Fixed Costs				8671
Total Costs				27673
Total cost / pound produced		1.38		
	Returns to land & operator's mgmt.			17327

DISCUSSION

A single series of tanks was chosen to accommodate flows as low as 200 gpm. If more water is available (at least 400 gpm), a paired series of parallel tanks would be possible which would reduce the fence requirement as well as the threat from oxygen or ammonia issues that arise with multiple water reuse in raceways.

The purchase price of precast tanks was highest. Installation was also higher due to the need for a heavy crane to lift the tanks into place. The poured tanks require the rental of a concrete pump, on the recommendation of two concrete contractors, for pouring the walls. Land preparation for the plastic tanks was slightly less than the flat bottomed concrete tanks because the plastic tanks require a narrower leveled pad due to the “U” shaped nature of the tank.

What is not known at this point is the useable life span of the plastic tanks. If the plastic tanks have a shorter life span than concrete tanks, then the depreciation costs would be higher for the plastic tanks. The HDPE material is extremely resistant to weathering and it is expected that a properly installed tank, with welded strips on each side to protect the open ribs, will last for at least 10-20 years and possibly as long as concrete tanks. Ultraviolet radiation would be expected to make the tanks less resilient over time. The manufacturer of the tanks provided ample documentation from various states (transportation departments) showing the expected useful life of HDPE to be exactly the same (50 years) as reinforced concrete products.

The difference in growth rate was likely due to the increased stress and reduced access to feed due to the predator problem in the plastic tanks. In an effort to reduce the smell of fish feed, which was drawing black bears to the plastic tanks, daily hand feeding began on April 26th. Prior to that, feed was placed in the demand feeders every other day and the caretaker noted if the feeder was empty. The concrete system did not have a predator problem and the demand feeder consistently contained feed. This resulted in improved access to trout feed for the concrete system.

There are many considerations to take into account when deciding between the two materials used in this study. A list of these considerations is shown in Table 5.

Table 5 Comparisons between concrete and plastic (HDPE)

CONSIDERATIONS	CONCRETE	PLASTIC (HDPE)
Purchase Cost	Higher	Lower
Tank weight	36,000 lbs.	760 lbs.
Site Prep. Cost	Higher	Lower
Vulnerability	Low	Moderate
Installation	Critical	Critical
Easily Modified	No	Yes
Useful Life	20 years	20 years??
Waste Removal	Slower	Faster
Flexibility	None	Some
Production volume	2000 gallons	2000 gallons
Size restrictions	Customized	Max. 60 inch diameter
Outside use	No restrictions	Recommended 20" in ground
Inside use	No restrictions	HDPE cross supports
Resale or Transfer	More difficult	Less difficult

Waste removal

The waste removal system for the plastic tanks was a simple and fast process. Taking advantage of the “U” shape and the smooth plastic material, an 18 inch squeegee was used to move the solid waste toward the 3 inch diameter manifold pipe that ran along the lowest portion of the settling zone. The ¾ inch openings in the lower portion of the manifold pipe allowed the accumulated solids to exit through the pipe, due to head pressure from water in the tank, when an external valve was opened. This process avoided siphoning and pumping, which added to the variable cost. The cost of the plastic tank system included the 3 inch valve and the manifold pipe for solids removal.

The difference in solid waste removal, as described, resulted in lower labor costs for the plastic tank system. There is however an inherent added risk that is not found with the pumping or siphon waste removal system used in concrete tanks. The risk is that the caretaker may forget to shut the valve that removes the solids from the bottom of the tank. An open valve with a 3 inch drain will divert much more water from the system than a siphon or pump which usually does not exceed a 2 inch diameter hose.

From November until May the water quality remained within the accepted parameters for trout at both of the sites. Growth and survival were normal for the plastic tanks until the intrusion of black bears became in issue. The bear intrusion caused the trout in the plastic tanks to jump out of the tanks which made it difficult to determine which tank they originated from. The fish feed appeared to be the bear’s target as many feeders were found strewn about the site, some needing repairs. Efforts to repair the fence were unsuccessful as a corner pole set in concrete was eventually bent over and the concrete base was shattered. On May 1st a 250 pound male black bear was trapped inside the research site and removed from the region. Two nights later another

bear had breached the fence. The fence has since been fortified and no additional bear damage has occurred.

ACKNOWLEDGMENTS

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THE EFFECTS OF POLLUTION REDUCTION ON A WILD TROUT STREAM

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KEY WORDS: fish hatchery, water quality, benthic macroinvertebrates, effluent treatment

ABSTRACT

Spring Run, a spring fed stream located in Grant County, West Virginia, is recognized as one of the best "wild" rainbow trout fisheries in the state. In recent years, fishermen have reported a decline in the fishery and aquatic insects, and an increase in algae in the stream. They are concerned about possible impacts from a WV Division of Natural Resources trout rearing facility, located upstream of the managed fishing section. The facility was cited in 2004 for discharging excess biochemical oxygen demand and total suspended solids into Spring Run in violation of their NPDES permit. In response, the WVDNR installed an effluent treatment process at the facility which became operational in June 2007. This study is investigating the response of Spring Run's biological communities to pollutant reductions following installation of the treatment system. Two years of baseline data were collected in Spring Run and a nearby control stream prior to installation of the effluent treatment system. The first year of post treatment data were collected this year. Data include water quality, stream flow, benthic macroinvertebrates, fish surveys by WVDNR, and fisherman catch records. This paper summarizes the baseline data and preliminary results from the first year of post-treatment data collections.

INTRODUCTION

Spring Run, a spring fed stream located in Grant County, West Virginia, is recognized as one of the best "wild" rainbow trout fisheries in the state. Since the early 1960's, landowners and other interested parties have installed and maintained various structures to form hiding and feeding habitat for trout along a one mile long section of the stream, and managed it for catch-and-release only fly fishing. In recent years, fishermen have noted a decline in the fishery, a decline in aquatic insects, and an increase in algae. They suspect pollution from the Spring Run Trout

Hatchery (SRH - a WV Division of Natural Resources trout rearing facility), located upstream of the managed fly fishing section, could be the reason for the apparent decline. SRH was cited in 2004 for discharging excess biochemical oxygen demand and total suspended solids into Spring Run in violation of their NPDES permit. In response, SRH installed an effluent treatment process at the facility that became operational in June 2007.

Concerns over the impacts of fish hatchery effluent on aquatic communities are not new. A study of five Virginia trout farms found that downstream waters had increased ammonia-nitrogen and nitrate nitrogen, decreased dissolved oxygen at post feeding and predawn hours, and found no effect on pH, nitrate nitrogen, and total phosphorus (Selong and Helfrich, 1998). The same report noted reductions in richness and abundance of sensitive macroinvertebrate taxa and increases in pollutant tolerant taxa (isopods and gastropods) below the outfall.

A Total Maximum Daily Load report for six Virginia streams impaired by trout farm effluent noted difficulties in assessing fish farm impacts: the effluent from trout farms is variable and episodic; the natural benthic community composition in limestone spring fed streams is uncertain, poor understanding of natural and other stressors and pollutant thresholds, and the challenge of finding suitable reference streams (VWRC, 2002).

The impending installation of an effluent treatment system at SRH provided an opportunity to assess the chemical and biological effects of the upgrade. A partnership between public and private entities was formed to develop and carry out this study. Two years of baseline data were collected in Spring Run and a nearby control stream prior to installation of the effluent treatment system. This paper summarizes the baseline data and provides partial results from the first year (2007) of post-treatment data collections.

METHODS

The project has two experimental components, an upstream/downstream design in Spring Run, and a control/experimental design that includes Dumpling Run, another spring fed stream nearby. Both streams are spring fed, with origins in the same geology: limestone (Helderberg and Tonoloway/Wills Creek) and sandstone (Oriskany, McKenzie) formations. Spring Run flows off the ridge to the northwest into South Mill Creek, a tributary of the South Branch of the Potomac River. Dumpling Run flows east into the South Fork of the South Branch of the Potomac River.

The upstream/downstream component includes three sites in Spring Run: the first site is near the spring upstream of the hatchery; the second site is below the hatchery near the upper end of the fly fishing stream section; and the third is near the lower end of the fly fishing section. There are two sites on Dumpling Run, one just below the spring, the other some distance downstream. Overall, this design allows both within stream and between stream comparisons. The samples collected near the two springs provides information on each stream's primary source water.. Under most conditions of flow the springs constitute the main source of water in both streams. Both streams also have periodic surface flow entering the main channel upstream of the springs.

Water samples were collected monthly from April through September, typically on Wednesday. Collections were avoided on hatchery cleanout days because the biosolids from aquaculture effluent are notoriously patchy and difficult to characterize (Joe Hankins, Freshwater Institute, personal communication). We chose to focus on the residual chronic impacts rather than the “cleanout plume.” However, due to scheduling requirements, samples in September 2006 were collected on a Monday during the cleanout.

Water quality parameters included nitrogen in the forms of ammonia-nitrogen, nitrate/nitrite, total Kjeldahl nitrogen, total nitrogen (as the sum of nitrate/nitrite and TKN), soluble reactive phosphorus, total phosphorus, total suspended solids (TSS), biochemical oxygen demand (BOD₅), pH, temperature, conductivity, and dissolved oxygen. All tests were performed by a certified laboratory using standard methods. Flows were measured on the same day as water samples were collected in each stream.

Benthic invertebrate and periphyton samples were collected twice each year at all sites, in the spring and the autumn, according to standard WV Department of Environmental Protection protocols. WVDNR conducted electro shocking fishery assessments. The fly fishermen of Spring Run recorded information on the size and location of trout caught and released.

RESULTS

Stream flows on sampling days for the most downstream sites in Spring Run and Dumpling Run are given in Figure 1. Flow in Spring Run was always at least twice as high as in Dumpling Run. Sampling day flows in 2005 were much more variable than in following years, and the streams were notably low between July and September in 2007. These flow patterns should be taken into account when reviewing the water quality data.

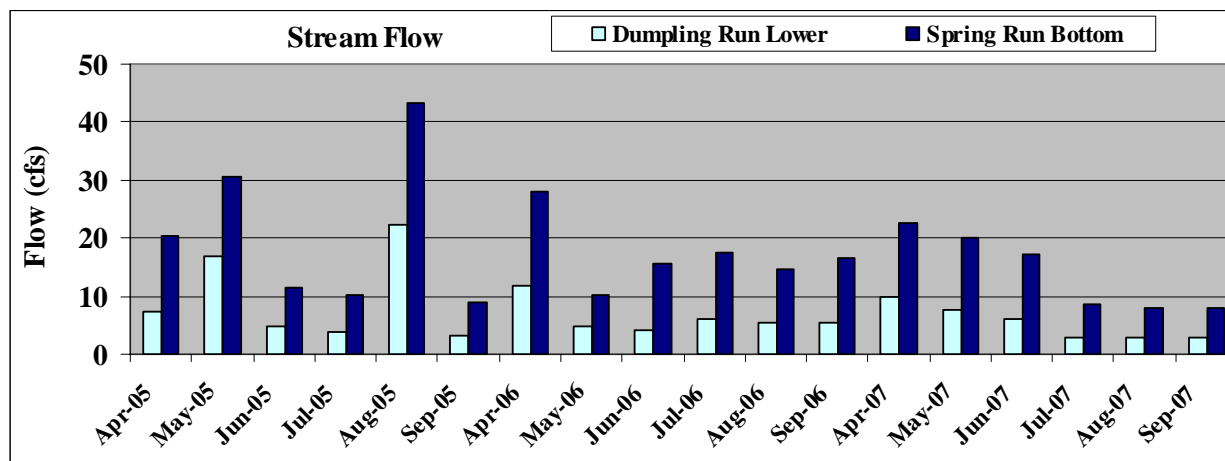


Figure 1. Stream flow measurements at sites in Spring Run and Dumpling Run.

Median values for pH, conductivity and dissolved oxygen by site and by year are provided in Table 1. These data indicate the source water in each stream was very similar, and these median values varied narrowly across all sites and all years. Both streams are alkaline, with moderately high conductivity, and high dissolved oxygen levels. No difference between pre-treatment (2005 and 2006) and post-treatment (2007) periods was evident.

Median values for total phosphorus (TP) and total nitrogen (TN) by site and year are provided in Table 2. Source water TP was similar in each stream, and did not increase in the downstream site in the control stream (Dumpling Run). Both sites below the hatchery in Spring Run (Spring Run Middle and Spring Run Bottom) had significantly higher median TP than all other locations. No difference in TP between pre-treatment (2005 and 2006) and post-treatment (2007) periods was evident. TP varied widely over time at all sites and, based on correlation analysis, did not vary with flow levels. However, the highest TP concentrations at all sites (except one) were recorded during an active runoff event in April 2006. Elevated TP from the hatchery effluent was evident at all flows at Spring Run Middle and Bottom.

Table 1. Median pH, conductivity, and dissolved oxygen by site and year.

Site	Median pH			Median Conductivity			Median Dissolved Oxygen (mg/L)		
	2005	2006	2007	2005	2006	2007	2005	2006	2007
Dumpling Run @Spring	7.8	7.6	7.5	286.9	254.2	257.0	10.3	10.6	10.5
Dumpling Run Bottom	8.1	7.8	7.6	283.5	263.9	271.5	10.4	10.6	10.3
Spring Run @Spring	7.9	7.7	7.7	296.1	330.3	358.0	10.3	10.5	10.1
Spring Run Middle	7.8	7.7	7.5	255.0	255.2	253.5	10.3	10.4	10.3
Spring Run Bottom	7.5	7.6	7.5	247.5	253.2	255.0	10.6	10.7	10.4

Median total nitrogen was significantly higher in the Spring Run source water than in Dumpling Run. Median TN did not increase in the downstream site in the control stream (Dumpling Run). Both sites below the hatchery in Spring Run (Spring Run Middle and Spring Run Bottom) had significantly higher TN than control sites in Dumpling Run, and higher (but not significantly) TN than the source water in the Spring Run spring. No difference in TN between pre-treatment (2005 and 2006) and post-treatment (2007) periods was evident. Total nitrogen (TN) varied widely and, based on correlation analysis, generally tracked with flows at all sites. The highest levels at all sites were observed in August '05 during a high water event. TN was always higher in all Spring Run sites than Dumpling Run.

Table 2. Median total phosphorus (TP) and total nitrogen (TN) by site and year.

Site	TP (mg/L) Median			TN (mg/L) Median		
	2005	2006	2007	2005	2006	2007
Dumpling Run @Spring	0.028	0.054	0.041	0.341	0.364	0.312
Dumpling Run Bottom	0.026	0.044	0.038	0.364	0.356	0.336
Spring Run @Spring	0.025	0.049	0.036	0.641	0.570	0.517
Spring Run Middle	0.075	0.103	0.106	0.887	0.634	0.775
Spring Run Bottom	0.087	0.103	0.085	0.877	0.734	0.747

Median values for BOD5 and TSS by site and year are provided in Table 3. Source water BOD5 was distinctly (but not significantly) higher in Dumpling Run than Spring Run. There was no marked change in BOD5 in the downstream direction in either stream. However, median BOD5 decreased in each sampling year at all sites. Correlation analysis indicated that BOD5 at Spring Run point source impacted sites tended to vary with flow, while patterns of BOD5 concentrations in non point sites had no apparent relationship to flow. BOD5 concentrations were notably low during the one active runoff event in April 2006, and notably high at all sites except DR Spring during a high water event in August 2005.

TSS was similar in the source water for the two streams, with data ranging broadly from 1.15 to 45 and 1.0 to 78.0 (mg/l) in Dumpling Run and Spring Run, respectively. Median TSS tended to increase somewhat in a downstream direction in both streams. Correlation analysis indicated that TSS roughly tracked with flows at all sites. The highest levels for all sites were observed in August '05 during a high water event, but were not notably high during an active runoff event captured in April 2006.

Table 3. Median BOD5 and TSS by site and year.

Site	Median BOD5 (mg/L)			Median TSS (mg/L)		
	2005	2006	2007	2005	2006	2007
Dumpling Run @Spring	1.54	1.400	0.68	4.50	1.15	2.50
Dumpling Run Bottom	1.515	1.100	0.605	2.08	5.50	4.50
Spring Run @Spring	0.985	0.645	0.415	1.58	2.58	3.50
Spring Run Middle	0.91	0.760	0.53	5.50	5.00	3.00
Spring Run Bottom	1.01	0.425	0.415	6.50	7.00	6.00

Benthics

Certain benthic macroinvertebrate data for 2005 and 2006 is reviewed below in Tables 4 and 5; the data for 2007 is not yet available. All sampling sites were dominated by one of two macroinvertebrate families: Gammaridae and Chironomidae. Gammaridae were the dominant organism at four of the five sites, accounting for 41% to 88% of all the organisms collected. Chironomidae were notably abundant in both of the Spring Run sampling sites located below the hatchery, and dominant in the site closest to the hatchery. The latter site was notable for the large amount of organic matter and matted algae entrained in the stream sediments. Chironomidae were present in relatively low numbers at the non point source impacted sites (Dumpling Run and Spring Run above the hatchery).

Table 4. Benthic macroinvertebrate metrics (%EPT and % Dominant) for samples collected in 2005 and 2006.

Site	% EPT				% Dominant			
	Spring 2005	Fall 2005	Spring 2006	Fall 2006	Spring 2005	Fall 2005	Spring 2006	Fall 2006
Dumpling Run @Spring	14.7	7.1	10.4	10.7	80.4	87.9	82.0	83.0
Dumpling Run Bottom	31	25.3	1.0	10.2	43.8	68.8	88.0	84.0
Spring Run @Spring	40.4	6.4	32.1	5.8	41.3	87.2	55.0	90.0
Spring Run Middle	38.5	4.6	27.9	13.5	42.3	76.9	63.0	80.0
Spring Run Bottom	8.9	1.6	14.6	13.0	65.5	77.6	53.0	70.0

Percent EPT (Ephemeroptera, Plecoptera, Trichoptera) is a standard benthic invertebrate index where higher values are considered indicative of good water quality. The %EPT metric was never particularly high. It was always low at DR Spring, and SR Bottom. It was quite variable at DR Lower, SR Spring and SR Lower. It tended to be lower in the fall at the latter two sites.

Table 5. Relative density of benthic macroinvertebrates.

Site	2005		2006	
	Spring	Fall	Spring	Fall
Dumpling Run @Spring	5600	5975	4600	2929
Dumpling Run Bottom	3500	2975	2986	7533
Spring Run @Spring	3833	3800	4180	5625
Spring Run Middle	3042	19500	523	3071
Spring Run Bottom	7800	4800	3767	2300

Abundance, or density, of benthic macroinvertebrates is not a reliable parameter because of the difficulty in collecting truly quantitative samples on hard bottomed streams. However, as the collection method and number of replicates for each site is the same, extrapolating from the numbers collected in the sorted subsample to the entire sample allows a rough estimate of relative density. Table 5 provides these estimates. With the understanding that such data are not terribly reliable, it is notable that relative density varied by a factor of three at the non point source impacted sites and SR Bottom. Relative density was much more variable at SR Middle, ranging from a low of 523 in Spring 2006 and 19,500 in the Fall 2005. This great variability, along with abundant Chironomids and a very heavy mass of entrained algae and organic matter at this site, were probably causally related.

Benthic macroinvertebrate results from this study are skewed by a combination of the extreme dominance by Gammaridae at most sites, and due to the standard 200 organism subsample procedure in use by WVDEP. Organisms that were readily observed during sample collection in

many cases did not show up in the final counts due to the two factors noted above. For example, Glossosomatid caddisflies were found in abundance on nearly every rock in Dumpling Run, but rarely showed up on the species list for this reason.

Fisheries

The West Virginia Division of Natural Resources, in cooperation with the West Virginia Conservation Agency, conducted two fishery surveys in Spring Run in 2005. Those results are available in the baseline report posted on Cacapon Institute's website.

Anglers with permits to fly fish, catch-and-release were invited to report the date fished, species, length, and stream location of their catch. The fly-fishing, catch-and-release section of Spring Run extends for about $\frac{3}{4}$ mile. This section was arbitrarily divided into 10 sections of unequal length, marked at streamside: numbered 0 thru 9, beginning with 0 at the downstream boundary and increasing upstream. Anglers fished wherever they chose. Fishing sessions ranged from less than an hour to several hours. Anglers reported on a card designed with stream sections vs. 6 length categories, in inches; 0-7, 8-10, 11-13, 14-16, 17-19, 20-up. A member of the monitoring team collected reports frequently and summarized data monthly. The purpose of the study was to acquire data on number, size, and location of Spring Run trout, not to evaluate angler success.

Table 6. Spring Run angler catch reports, rainbow trout: 2005 – 2006.

<i>Length</i>	<i>Stream section</i>										
	0	1	2	3	4	5	6	7	8	9	Total
0-7	95	154	119	219	354	488	313	188	88	80	2098
8--10	40	49	46	121	249	330	312	351	267	228	1993
11--13	11	15	35	45	73	121	152	193	279	311	1235
14--16	0	5	4	24	34	41	64	69	72	178	491
17--19	0	1	1	2	8	6	11	16	14	43	102
20--up	0	0	1	1	4	5	7	11	7	21	57
Total	146	224	206	412	722	991	859	828	727	861	5976
%	2.4%	3.7%	3.4%	6.9%	12.1%	16.6%	14.4%	13.9%	12.2%	14.4%	

From April through December 2005, 65 anglers reported 230 fishing sessions resulting in 16.1 rainbow trout/angler session. From January through December 2006, 76 anglers reported 232 fishing sessions resulting in 9.8 rainbow trout/angler session. During both baseline years, the smaller trout (0 – 10”) were most abundant in the middle sections, while the larger trout (11” to >2-“) were most abundant in the further upstream. Fishermen indicate that this pattern represents a shift from past years, but there is no specific data on distribution.

DISCUSSION

The two study streams are impacted by a variety of potential sources of pollution, some readily apparent and some not. The Spring Run watershed contains the trout rearing facility, which is a

known source of BOD, TSS and nutrients, as well as a number of non point sources including poultry houses, residences, roads, and occasional cattle. The Dumpling Run watershed has no point sources, and apparently no poultry houses, but includes residences and small farms with livestock, as well as a dirt and gravel road. In addition, the source springs in both watersheds both originate in limestone and sandstone strata and show rapid changes (turbidity, increase in flow) following heavy precipitation; this is indicative of solution channel connections through limestone at the surface of the ground.

Despite the wealth of confounding variables, some patterns are reasonably clear from the baseline data. The spring source water for the two streams has similar pH, conductivity, dissolved oxygen, TSS, and phosphorus. Source water in Dumpling Run tends to have less nitrate, and total N than Spring Run, and higher BOD5. Conductivity and pH tend to increase or not change in a downstream direction in Dumpling Run, and tend to decrease in a downstream direction in Spring Run. Nutrients and TSS are generally similar in the two Dumpling Run sites, and tend to increase in a downstream direction in Spring Run, often dramatically.

The decision to collect water samples two days after the scheduled cleanouts at the hatchery probably contributed to the apparently anomalous result of Dumpling Run, the control stream, having somewhat more BOD5 and TSS than Spring Run, the stream with the effluent containing excess BOD and TSS. It is quite clear that we do not observe a significant residual impact in the water column from those cleanouts two days after the fact, as suspended material is readily observable in Spring Run on cleanout days.

Preliminary review of post-treatment water quality data indicates that the plant upgrade did not change the water quality characteristics of the water downstream of the plant on non-cleanout days. Phosphorus remains elevated, and TSS and BOD5 remain lower in SR than DR. However, it is also clear that we are not capturing the reductions in the pollutant plume as a result of the new effluent treatment system. The cleaning process, which went on line June 4, 2007, involves placing blocking weirs in front of the quiescent zones, after which the quiescent zone is brushed cleaned and the wastewater from the quiescent zones is piped into the clarifier. The clarifier is filled to its holding capacity but is not allowed to overflow. Wastewater in the clarifier settles for 24-48 hours and the clarified water is then decanted and mixed with hatchery water back into Spring Run. The sludge remaining in the clarifier is pumped to the sludge holding tank for later disposal by land application. Data provided by WVDNR indicates that this process reduces pollutant loads related to cleanout by roughly 90%. For example, TP concentrations in the effluent stream during cleanout fell from an average of 4.5 mg/L to 0.54 mg/L. WVDNR data also indicates that the hatchery's effluent stream, prior to initiating the cleaning process, contains very low levels of phosphorus that are not at all consistent with the concentrations we typically observe further downstream. As of this writing, we are working to resolve that anomaly, and plan to include the results of that investigation in our presentation. We also hope to have preliminary results of biological monitoring in 2007.

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Friends of Spring Run's Wild Trout, Cacapon Institute (CI), the WV Conservation Agency (WVCA), WV Department of Agriculture (WVDA), WV Division of Natural Resources

(WVDNR), WV Department of Environmental Protection (WVDEP), and the Freshwater Institute are partnering in this study. This project is funded primarily by West Virginia Conservation Agency's participation through the Chesapeake Bay Program. An associated sediment reduction project was funded through a Friends of Spring Run's Wild Trout 2005 Stream Partners Grant. Additionally, a home school group is monitoring the lower portion of Spring Run on a regular basis. WVDA, WVDEP and WVDNR are all contributing in-kind services to the project. WVDA is collecting water samples, taking flow measurements, and performing field and laboratory water quality analyses. WVDEP is participating in collections of benthic invertebrate and periphyton and paying for analysis of those samples. WVDNR is performing fish surveys and Friends of Spring Run's Wild Trout is providing information on size and location of trout caught and released by permitted fly fisherman. The Spring Run Baseline report is available for download at www.cacaponinstitute.org.

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Session V-C

Improving Constructed Wetland Design for Water Quality Treatment

- Vegetation-Induced Roughness: A Flume Study with Natural Wetland Plants
- Treatment Wetland Research in Atlantic Canada
- Evaluating a Constructed Floodplain Wetland for Nutrient Removal Efficiencies

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VEGETATION-INDUCED ROUGHNESS: A FLUME STUDY WITH NATURAL WETLAND PLANTS

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ABSTRACT

Wetlands are important ecosystems, providing habitat for wildlife and fish and shellfish production, water storage, erosion control, and water quality improvement and preservation. Models to estimate hydraulic resistance due to vegetation in emergent wetlands are crucial to good wetland design and analysis. Models should not be so complex so as to require large amounts of expensive data or so simple that major hydraulic processes are ignored. Ideally an existing stage-discharge model consistent with wetland flow can be adapted and hydraulic resistance determined using known or easily measurable properties of the vegetation of interest. The goal of this project is to improve modeling of emergent wetlands by linking properties of the vegetation to flow. Various existing resistance equations proposed to estimate hydraulic resistance due to vegetation will be evaluated and the model parameters related to quantifying roughness will be related to measurable properties of the vegetation. A large outdoor vegetated flume was constructed at Price's Fork Research Center near Blacksburg, Virginia to measure flow and water surface slope through woolgrass (*Scirpus cyperinus*), a common native emergent wetland plant. Measurements of clump and stem density and diameter, volume, blockage factor, and stiffness were collected after each set of flume runs. Following data analysis, data will be fit to existing wetland resistance models. Regression analysis will be employed to correlate vegetation properties to the model roughness parameters. The study goal is to better understand the relationship between vegetation and hydraulic roughness in emergent wetlands with the ultimate goal of improving emergent wetland design and analysis.

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TREATMENT WETLAND RESEARCH IN ATLANTIC CANADA

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KEY WORDS: treatment wetlands, agricultural wastewater, cold climate

ABSTRACT

Constructed wetland systems have been utilized throughout Canada for a number of years. One of the first systems constructed in Atlantic Canada was in Pictou County, Nova Scotia. This system was constructed on a dairy farm to evaluate the treatment of manure runoff and milkhouse wash water. Initial results and monitoring of this system showed promise that constructed wetlands could treat agricultural wastewater year-round in cold climate regions. As a result, an innovative wetland research facility was established at the Nova Scotia Agricultural College's Bio-Environmental Engineering Center (BEEC) in Nova Scotia, Canada. This facility included two pilot scale surface flow wetland treatment systems that allowed for precise inflow and outflow monitoring. Subsequent research included assessments on: (i) over-winter

performance, (ii) hydraulic retention time estimates, (iii) mechanical aeration, (iv) seasonal vs. continuous loading, (v) source-sink relationships regarding greenhouse gases and (vi) rate constant estimates. Overall wetland performance results will be presented.

INTRODUCTION

Agriculture has been identified as a potential contributor to rural water quality impairment. As farm land area decreases and livestock numbers increase, the disposal of agricultural wastes such as milkhouse washwater, silage leachate, and manure runoff becomes increasingly important (Kashmanian 1994). Agricultural wastewaters are often high in five-day biochemical oxygen demand (BOD₅), total suspended solids (TSS), and nutrients such as nitrogen (N) and phosphorus (P). Constructed wetlands are alternative wastewater treatment systems that have low capital and maintenance requirements compared to conventional wastewater treatment systems. For these reasons, wetlands have become popular for agricultural applications, specifically in Atlantic Canada. Wetlands have been utilized and studied more prolifically in warmer climates (e.g. southern United States). It was therefore, necessary to conduct local research, to assess the suitability of these systems in Atlantic Canadian cold climate conditions, before they could be widely adopted. A research program was initiated at the Nova Scotia Agricultural College (NSAC) to address these issues. Several on-farm wetlands have been constructed and monitored in co-operation with local farmers. In addition, significant wetlands research capacity has been developed at the NSAC's Bio-Environmental Engineering Center (BEEC), where 2 pilot scale wetland research sites were developed. This paper will focus on research conducted at the BEEC sites.

METHODS

Two separate constructed wetland sites, BEEC 1 (Fig. 1) and BEEC 2 (Fig. 2) were established at the BEEC, in Bible Hill, Canada. Both sites allowed wastewater to be loaded automatically, providing some control over loading rates. Both sites were loaded with dairy wastewater. A more detailed description of the sites follows.

Site descriptions

BEEC 1 Wetlands

The BEEC 1 site consisted of two similar surface flow (SF) wetlands (~100 m² each) (Fig. 1). One was a control (unmanaged) wetland and the other was used as a *managed* wetland, allowing the effects of different management practices to be evaluated. Each wetland cell contained alternating deep (~0.85 m deep) and shallow (0.15 m deep) zones. Shallow zones were vegetated with cattails, while duckweed (*Lemna* sp.) typically floated on the surface of the deep zones. A polyethylene liner was installed to prevent seepage due to the high hydraulic conductivity of the native soil. Heated sampling huts located at the inlet and the outlet allowed continuous, year-round monitoring of water quality and flows. A more detailed description of the experimental site was provided by Smith *et al.* (2006).

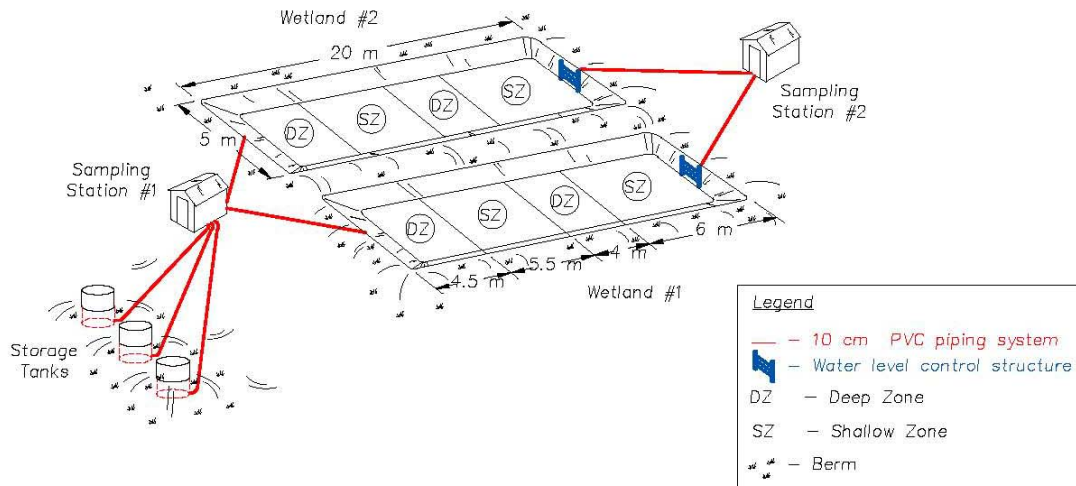


Figure 1. A schematic of the BEEC 1 constructed wetlands located at the Bio-Environmental Engineering Center.

BEEC 2 Wetlands

The second site (BEEC 2) consisted of 6 pilot scale wetlands (6.6 m² each) (Fig 2.). Initially all the wetlands were SF wetlands, with later research involving the conversion of three wetlands to subsurface flow. Each cell was configured with deep (0.8 m deep) and shallow (0.3 m deep and vegetated with cattails) zones (Fig. 2). This site was designed such that GHG fluxes can be continuously monitored to determine if there are tradeoffs between water quality treatment and air quality. Methane (CH₄) and nitrous oxide (N₂O) were quantified using tuneable diode laser trace gas analyzers (Campbell Scientific, Edmonton, AB, Canada), while carbon dioxide (CO₂) was quantified using a closed path infrared detector (Li-Cor, Lincoln, NE, USA). Ammonia emissions were also determined by a gas trapping method. Heated inlet and outlet sampling huts allowed for continuous, year-round water quality and flow monitoring.

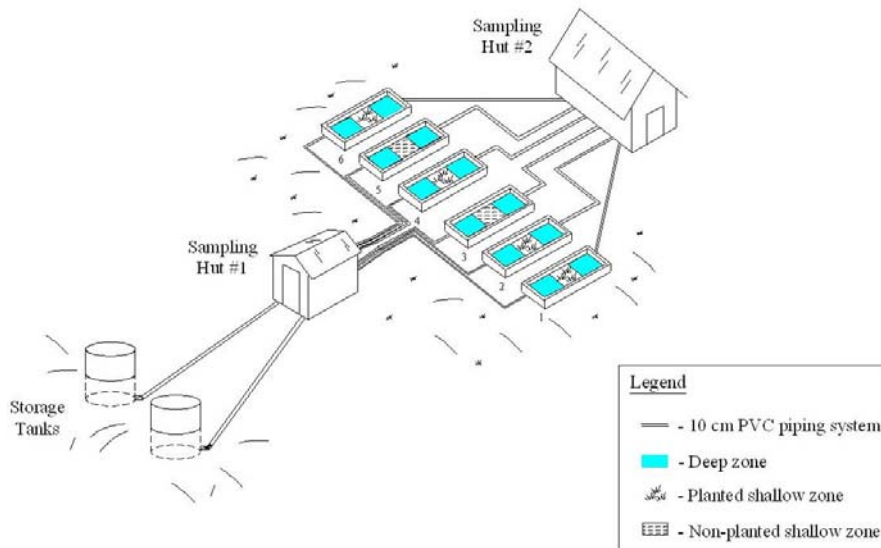


Figure 2. A schematic of the BEEC 2 constructed wetlands at the Bio-Environmental Engineering Center.

Water quality and meteorological monitoring

Water quality, flow rates and meteorological conditions were monitored at BEEC 1 and BEEC 2 sites. Weekly samples were collected at the inlet and outlets of all wetlands using sterile high density polyethylene bottles. Water quality was monitored for BOD₅, TSS, total Kjeldahl nitrogen (TKN), ammonia+ammonium-N (NH₃+NH₄⁺-N), nitrate-N (NO₃⁻-N), total P (TP), soluble reactive P (SRP) and fecal coliforms (FC). Analyses were conducted according to standard APHA based methods (Clesceri *et al.* 1998). Flows were monitored using calibrated tipping buckets wired to dataloggers

RESULTS AND DISCUSSION

Year-round treatment performance

Results from BEEC 1 demonstrated that SF wetlands in Atlantic Canada are capable of providing effective year-round treatment of various agricultural wastewaters (Smith *et al.* 2005a; Smith *et al.* 2006). Average daily loading rates and overall mass reductions from 2000 through 2004 for the BEEC 1 *control* wetland are presented in Table 1. Reductions in mass were similar to those reported for other SF wetlands receiving agricultural wastewaters (e.g. Cronk 1996; Newman *et al.* 2000). As expected, P treatment was limiting in these systems (Table 1). An analysis of operational data from the *control* BEEC 1 wetland from 2000 through 2005 found reduced P treatment when external hydrological loading (e.g. rainfall and snowmelt) was high. When monthly outflows were <100 mm mass reductions were at least 50%, however, when monthly outflows exceeded 100 mm, mass reductions were <50%, and much more variable. Shortened retention times during these hydrological events were identified as the main factor inhibiting wetland performance. Decreased retention times are known to inhibit P treatment (Sakadevan and Bavor 1999).

Table 1. Average daily loading rates (kg ha⁻¹ or CFU ha⁻¹ for FC) and mass reductions (%) for selected parameters for the BEEC 1 *control* wetland from 2000 through 2004.

Parameter	Daily Loading Rate (kg ha ⁻¹)	Mass Reduction (%)
BOD ₅	57.8	94.0
TP	1.70	76.0
TKN	8.10	80.2
NO ₃ ⁻ -N	0.13	82.0
NH ₃ -N	7.25	81.8
FC	1.91 x 10 ¹⁰	99.1

Wetland tracer studies

Numerous tracer studies have been conducted to characterize wetland hydraulics at BEEC 1. Studies were conducted during different seasons to assess seasonal impacts. Other studies were

carried out to determine the effects of selected management practices on retention times. At BEEC 1, retention times were assessed in the spring and winter for the *control* wetland, and for a wetland with *managed* water depths (creation of air layer beneath ice) (Smith *et al.* 2005b). Retention times were slightly longer in the managed wetland and were also longer during winter months. Actual retention times for these wetlands were around 50% of the theoretical retention time, demonstrating significant short circuiting and the presence of dead zones (Smith *et al.* 2005b).

Artificial wetland aeration

At BEEC 1, a study was carried out in which a diffused air aeration system was installed in the first deep zone of the *managed* wetland, while the other wetland (control) was not aerated. The impacts of artificial aeration were found to be more pronounced for pollutants removed by biological processes, such as nitrogen (N). Significantly ($P < 0.05$) higher TKN and $\text{NH}_3 + \text{NH}_4^+ - \text{N}$ removals were observed for the aerated system, however, $\text{NO}_3^- - \text{N}$ accumulation was also greater ($P < 0.05$) in the aerated wetland. Improved TKN treatment in aerated wetlands was also reported by Oullet-Plamondon *et al.* (2006) for subsurface flow wetlands receiving fish farm effluent. Nitrate-N accumulation at BEEC 1 was attributed to inhibited denitrification caused by increased dissolved oxygen within the aerated system. Aeration did not impact the removal of BOD_5 , TSS, TP and FC ($P > 0.05$). Tracer studies indicated that aeration did not affect the retention time when compared to a non-aerated wetland. Temperature profile data did however suggest that mixing was more uniform in artificially aerated zones. Despite the above mentioned benefits to N treatment, aeration did not prove to be cost effective. Therefore, aeration is not being recommended, since effective performance was also observed for the non-aerated wetland.

Derivations of first order rate constants

Areal first order rate constants (k) were calculated using operational data from the *control* wetland at BEEC 1 and reported by Jamieson *et al.* (2007). Monthly k values were determined according to the $k-C^*$ model described by Kadlec and Knight (1996). Mean, maximum and minimum k values derived by Jamieson *et al.* (2007) are provided in Table 2. Monthly variation was high as evidenced by the high range of the values (Table 2). Mean rate constants were lower than those reported by Knight *et al.* (2000) for SF wetlands receiving livestock wastewater. This research has provided region specific k values and thus improved local design criteria.

Table 2. Mean, maximum (max.) and minimum (min.) first order rate constants (k) (m y^{-1}) computed from data collected at BEEC 1 for selected wastewater parameters.

Parameter	k (m y^{-1})		
	Mean	Max.	Min.
BOD_5	9.7	73.8	0.5
TP	4.3	38.4	5.8
TKN	6.8	59.9	0.4
$\text{NH}_3 + \text{NH}_4^+ - \text{N}$	7.0	52.3	0.5
TSS	8.7	66.2	0.6
FC	11.0	54.5	0.1

Seasonal vs. Year-round loading

Another study at BEEC 1 investigated differences between seasonal (growing season [GS]) and continuous (year-round) loading of constructed wetlands. No benefits to either approach were observed in terms of yearly areal mass removal or export rates. The continuously loaded wetland did however perform better during periods of high external hydrological loading (e.g. Fig 3). Poorer performance by the seasonally loaded wetland under these conditions during the GS was attributed to higher wastewater loading rates experienced by this system. Higher wastewater loading rates meant that the seasonally loaded wetland was less able to buffer external hydrological inputs, resulting in reduced retention times. Shorter retention times were believed to be the main cause of treatment inhibition. Due to improved performance under high external hydrological loading, and reduced construction costs, continuous loading is being recommended.

Greenhouse gas assessments

Research at BEEC 2 was initiated in 2001 to assess source-sink relationships regarding GHG's within SF wetlands. Initial studies focused on vegetative and loading rate effects on GHG emissions. The GS net GHG budget for vegetated SF wetlands loaded at $125 \text{ kg ha}^{-1} \text{ d}^{-1}$ was $-7.12 \text{ g CO}_2 \text{ eq. m}^{-2} \text{ d}^{-1}$. The wetlands however, functioned as net GHG sources during the NGS with a budget of $+30.71 \text{ g CO}_2 \text{ eq. m}^{-2} \text{ d}^{-1}$. Differences between the GS and NGS were attributed to reduced plant activity resulting in limited CO_2 sequestration during the NGS.

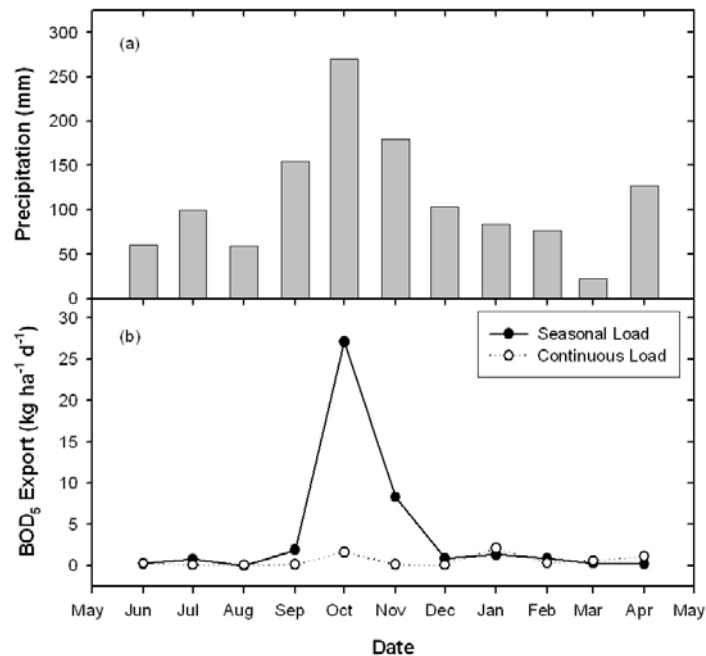


Figure 3. Plots of (a) monthly precipitation and average daily mass exports ($\text{kg ha}^{-1} \text{ d}^{-1}$) from the seasonally and continuously loaded wetlands (BEEC 1) in 2005-2006 for (b) BOD_5 .

Recently, 3 wetlands at BEEC 2 were converted to subsurface flow (SSF). This allowed a comprehensive comparison of surface and subsurface flow wetlands in terms of both water and air quality. Wastewater treatment was comparable for surface and subsurface flow wetlands. Greenhouse gas emissions from subsurface flow wetlands were however, 1/3 of those from SF wetlands. Ammonia emissions from SSF wetlands were also significantly lower than from SF wetlands.

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CONCLUSIONS

A research program established in 2000 at BEEC in Bible Hill, Nova Scotia Canada has evaluated agricultural constructed wetlands in Atlantic Canada. The overall goals of the collective research have been to promote regional acceptance of wetlands as wastewater treatment systems, improve local design criteria, and to identify cost effective management practices to improve system performance. Effective year-round treatment has been observed, with reductions in mass ranging from 76 to 99% for various wastewater parameters. Modifications to the systems, such as adding artificial aeration, have not proven cost effective enough to warrant wide spread use. Manipulating loading in an attempt to avoid winter loading was also found to be impractical and as such year-round loading is recommended. Local design criteria have been improved through the derivation of 1st order rate constants specific to Atlantic Canada. Research has also demonstrated that when properly managed wetlands can function as net sinks for GHG's. Comparable wastewater treatment was observed for both SF and SSF wetlands at BEEC 2, however, SSF wetlands were found to have GHG benefits.

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EVALUATING A CONSTRUCTED FLOODPLAIN WETLAND FOR NUTRIENT REMOVAL EFFICIENCIES

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ABSTRACT

As part of a basin-scale nutrient reduction project, a floodplain wetland will be designed and implemented in the upper Opequon Creek basin in Northern Virginia, USA. The goal of the project is to implement innovative best management practices (BMPs) to reduce nonpoint source nitrogen and phosphorus loads from the urban and agricultural lands in the basin. The floodplain wetland will be designed to capture event flows from Opequon Creek that carry sediments and nutrients, treat the flow in the wetland, and return the treated flow back to the creek. Retention time, water depth, and wetland volume will be adjusted for maximum available biological treatment. Various depth zones through the wetland allow for high- and low-flow patterns and a compilation of native wetland plant species. Water quality and discharge characteristics will be monitored at stations at the floodplain wetland inlet and outlet. Comparisons of nutrient and discharge data between the two stations will provide an accurate measure of success of the treatment processes implemented in the wetland. Successes will be carried through to subsequent water quality BMPs in the project.



Session VI-A

Providing Incentives for Farmers to Improve Water Quality

- Farmers as Producers of Clean Water: Getting Incentive Payments Right
- Inducing Farmer Participation in a Watershed Level Program to Improve Water Quality
- The Opequon Creek Targeted Watershed Project – A State Partnership to Benefit the Chesapeake Bay
- Stone Soup: The Rediscovery of Landscapes and Community, Introducing Land Care in Grayson County, Virginia, USA

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FARMERS AS PRODUCERS OF CLEAN WATER: GETTING INCENTIVE PAYMENTS RIGHT

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KEY WORDS: economic incentives, nonpoint pollution, payment estimation.

ABSTRACT

This paper presents a field test of economic incentives designed to reduce agricultural nonpoint pollution. In this two year experiment, farmers receive payments based on surface water quality and quantity. The monthly payment is the product of water quantity times a price per unit of water times a water quality adjustment factor. Difficulties arise when estimating a price for water and likely payments, given an absence of detailed data on stream flow and water quality. In this paper, we present how we estimated prices and payments, and we evaluate this preliminary work based on the first six months of the experiment.

INTRODUCTION

In this research, we are field testing a novel approach to economic incentives designed to reduce agricultural nonpoint pollution (ANP). Numerous authors have discussed the challenges posed by ANP (Cabe and Herriges 1992, Smith and Tomasi 1995, Shortle *et al.* 1998, and Ribaudo *et al.* 1999). The premise underlying our research is that if farmers are paid as a group based on water quantity and ambient water quality measured at the watershed level, then they will respond in a way that cost-effectively reduces ANP. Group-based approaches to water quality have been investigated by Spraggon (2002), Romstad (2003), Poe *et al.* (2004), Taylor *et al.* (2004), and Vossler *et al.* (2006). For more background on ambient-based approaches, see Segerson (1988),

Horan *et al.* (1998), and Segerson and Wu (2006). To examine our premise in the field, we need a payment schedule that indicates to farmers how much they can expect to earn. However, developing such a payment schedule is difficult when prices for water are absent and data on water quality and stream flow are lacking.

Payments need to do the following: 1) assist in motivating farmers to participate in the experiment, 2) provide an incentive to abate ANP, 3) sensibly reflect environmental conditions, and 4) be seen as fair and likely to enhance the well-being of participants. We make the assumption that farmers will be motivated to participate and abate when their well-being is enhanced by these actions. Payments that provide only positive economic incentives can motivate farmers along with other social considerations (see companion paper by Collins *et al.* 2007). Reflecting environmental conditions means that payments need to make allowances for natural fluctuations beyond the control of farmers. Lastly, Breetz *et al.* (2005) argue that fairness and equity concerns can weigh heavily when farmers are considering whether or not to participate in a conservation program.

In this paper, we describe how we estimated and simulated payments for a field experiment in Cullers Run watershed. Payments are evaluated as to whether they provide the proper incentives, and actual payments-to-date are evaluated compared to preliminary modeling. We also address whether or not farmers were motivated to participate and pursue ANP abatement.

Cullers Run watershed occupies 7,360 acres in West Virginia's largest poultry production county (USDA National Agricultural Statistics Service 2005). Sixteen percent of the watershed is devoted to agriculture of which most is pasture or hay land. Row crops comprise 3.63% of the agricultural land, mostly in the floodplain (Cacapon Institute 2002). The rest of the watershed is forest. There are approximately twelve poultry houses conducting intensive poultry production in the watershed. Much of the local demand for agricultural fertilizer is met at a relatively low cost with manure from local poultry production.

METHODS

Equation 1 presents the generalized payment formula used in this experiment:

$$\text{Watershed Payment} = (\text{volume of water}) \times \left(\frac{\text{price}}{\text{unit volume}} \right) \times (\text{quality adjustment factor}) \quad (1)$$

This payment formula has intuitive appeal. Like a traditional agricultural commodity, the greater volume of water that farmers "produce" the more they get paid because water volume flowing from a watershed is multiplied by a per unit price for water. Also, a quality adjustment factor is included to increase payments with increasing quality. However, for farmers to evaluate the feasibility of participating and pursuing ANP abatement, they need to know how large the payments are likely to be so that each of the three components in this payment formula must be estimated prior to the experiment.

Estimating flow

With no existing flow gauge, we needed to estimate the acre-feet of water per month “produced” by Cullers Run watershed. Using 32 months of mean monthly flow data from the US Geological Survey (2005) for a nearby watershed of comparable size, and monthly rainfall data from Cacapon Institute (unpublished data), we generated “best fit” non-linear equations with Microsoft Excel 2003 that relate this watershed flow data to rainfall. These equations provided estimates of watershed level flow based on rainfall. Estimating separate equations for months of active and inactive evapotranspiration, more than doubled the amount of variation explained by these equations based on the value of R^2 ($R^2 = 0.42$, and $R^2 = 0.48$ respectively).

Solving for a water price

A water price was estimated based on the opportunity cost of water quality improvement to farmers. To derive this opportunity cost, we assumed that 10% of the agricultural land area in the watershed was set aside to grass riparian buffers. Implementation of these buffers was assumed to reduce pollution loading of nitrate-N by 75% based on Palace et al. (1998) expected reductions to sediment and bacteria. The loss of agricultural income was approximated with crop net revenue from the USDA census of corn and hay production costs and returns for the Eastern Uplands between 1996 and 2004, standardized using the Bureau of Labor Statistics consumer price index (US Bureau of Labor Statistics 2005). Estimates for pasture revenue are based on the average pasture rental values as presented by Whittle and Stanley (2003).

Using this information, we set up an optimization program using the CONOPT nonlinear programming solver within GAMS Integrated Development Environment (GAMS) software. Our objective function assumed net revenue maximizing farmers who were subject to constraints on acreage, their ability to shift land between pasture and cropland, and non-negativity from water revenue. Intuitively, our model sets the cost of ANP abatement against the higher payments for water that result from improved water quality (see Adjustment Factor discussion below). We ran separate GAMS programs for each season and rainfall level with successively higher water prices to determine the minimum prices that could be offered per acre-foot of water to induce Cullers Run farmers to undertake water quality conservation.

The prices generated by our optimization model are shown in Table 1. They make intuitive sense from both economic value and pollutant loading perspectives. During the growing season (May-September), the discharge is lower due to low rainfall, and loading of pollutants is decreased. Thus, higher per unit water prices are needed to induce BMP implementation. Conversely, high discharges and non-growing season leads to lower prices, as marginal water values are low and pollutant loads are higher. By using different prices, payment risk to both landowners and the regulator is reduced. A sensitivity analysis of the optimization program projected only small changes in payments between rainfall regimes (Maille and Collins 2006).

Table 1. Computed seasonal water prices.

May through September		October through April	
Monthly Discharge (acre-feet)	Dollars per Acre-Foot	Monthly Discharge (acre-feet)	Dollars per Acre-Foot
Up to 320	18	Up to 740	8
321-800	8	Over 740	5
Over 800	5		

Incorporating water quality

Based on data from the study area for bacteria, turbidity, various forms of phosphorus, and nitrate-N (Cacapon Institute, 2002) we selected nitrate-N as an indicator of water quality changes. Monitoring data from the Cacapon Institute showed that nitrate-N concentration in streams, and therefore load, varied more predictably with rainfall than other pollutants. Its presence also was positively related to the extent of agricultural land. In addition, nitrate-N is a relevant pollutant with respect to water treatment costs and stream degradation. It does however, entail some disadvantages. Unlike turbidity, nitrate-N is not visible and it resides in subsurface water, thus potentially producing a time lag between generation and contribution to ambient stream concentrations. Finally, it is present naturally which can introduce “noise” to the incentive scheme.

Recalling that the payment schedule needs to be sensible and fair, we think that payments should account for the background fraction of nitrate-N contamination that farmers do not control. With this in mind we developed the “index watershed” approach described by equation 2.

$$\text{Adjustment Factor} = \frac{\text{Index Watershed nitrate-N}}{\text{Experimental Watershed nitrate-N}} \quad (2)$$

Waites Run watershed is our index watershed. It is 96% forested with very little agricultural land, located in the same county as our experimental watershed, and is approximately the same size. Thus, it serves as a weather-sensitive baseline measure of nitrate-N loads.

To use equation 2 we needed estimates of nitrate-N loading in the watershed as a function of land use. For the current pasture and forest nitrate-N contributions, we generated a system of three simultaneous equations with pollutant contribution for forest and pasture as the independent variable using watershed-level data from Cacapon Institute (2002). We estimated the pollutant contribution from cropland based on Randall and Vetsch (2005).

In addition, the monitoring data show that nitrate-N concentration increases during periods of high rainfall. With this in mind we related our rainfall data to Cullers Run nitrate-N concentrations (Cacapon Institute, unpublished data) with Excel 2003 providing the best fit non-linear equations for the non-growing season and growing season respectively. Once again the explanatory power of the equations was boosted significantly by estimating them on a seasonal basis ($R^2 = 0.53$, and $R^2 = 0.48$ respectively). We formed a ratio of estimated nitrate-N during

periods of low, average or high rainfall, over the expected nitrate-N concentrations during average rainfall and used this ratio as a simple multiplier. Our low, average, and high rainfall levels correspond with the monthly rainfall totals for each season exceeded by 25%, 50%, and 75% of the months respectively. Thus, the estimate of Cullers Run nitrate is shown in equation 3 as the sum of current nitrate contributions for each land-use, increased or decreased slightly by a rainfall multiplier.

$$\frac{\text{Lbs. nitrate-N}}{\text{month}} = \sum_i \left[\left(\frac{\text{Lbs. nitrate-N}}{\text{acre landuse } i} \right) \times (\text{acres landuse } i) \right] \times \left(\frac{e^{0.3188 \times \text{rainfall level}}}{e^{0.3188 \times \text{ave. monthly rainfall}}} \right) \quad (3)$$

The resulting nitrate-N estimated loads from equation 3 make up the denominator for the operational version of the adjustment factor shown by equation 4, based on the non-growing season rainfall multiplier. The numerator in the adjustment factor is the average Waites Run nitrate-N load.

$$\text{Adj. Factor} = \frac{\text{Average nitrate-N load Waites Run}}{\sum_i \left[\left(\frac{\text{Lbs. nitrate-N}}{\text{acre landuse } i} \right) \times (\text{acres landuse } i) \right] \times \left(\frac{e^{0.3188 \times \text{rainfall level}}}{e^{0.3188 \times \text{ave. monthly rainfall}}} \right)} \quad (4)$$

RESULTS

Estimating payments

Farmers need an estimate of likely payments, not just prices, before deciding to participate. With this in mind, using equation 1, we calculated four years of monthly watershed payments. We used estimated discharge, nitrate-N concentrations in Cullers Run and Waites Run as proxies for load, and applied prices from Table 1. The annual total of these payments averaged \$7,721 with a range of \$4,593 to \$9,400.

Simulations

To assess the incentives created by our watershed payment formula, we examined simulated payments. These payments are represented by data points in Figure 1a. They were computed from equation 1 using data on regional rainfall and nitrate-N concentrations in Cullers and Waites Run from 1999 to 2002. The trend lines “Table 1 Prices” show the relationship between estimated payment and discharge separately at low, medium and high discharge levels using Table 1 summer prices. The “One Price” trend lines show the price-discharge relationship that would exist at these three discharge levels if price was held constant at \$8 / ac-ft. We see that, on average, payments are boosted at low discharge and decreased at high discharge by using prices that are sensitive to discharge. We also see that payments are still an increasing function of discharge. Thus, the schedule of prices in Table 1 appears to maintain the desired incentive, while diminishing the risk of overly low or overly high payments caused by variation in rainfall. Figure 1b shows how increasing abatement affects the payments. Looking at the payment for a single month (hollow triangle just above \$500 at a discharge of 950 ac-ft from Figure 1a) and maintaining the original price, payments increase at an increasing rate as nitrate-N decreases.

Assuming total cost of abatement increases with increasing abatement this is an appropriate shape for the payment function. We also estimated payments based on a 25% reduction in nitrate-N. At this level of abatement, payments were \$9,595 annually, ranging between \$5,898 and \$11,480. The difference between the payments with and without additional abatement represents an opportunity cost of not abating nitrate-N.

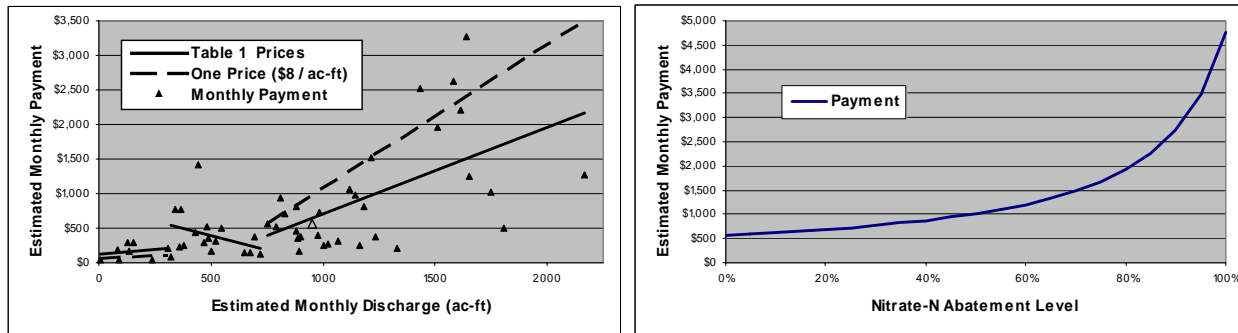


Figure 1a and 1b. Payment simulations.

Lastly, to what extent does the adjustment factor reduce natural variation of nitrate-N? To assess this question, we regressed estimated discharge on Culler Run nitrate-N concentration over our four years of data (Cacapon Institute, unpublished data). We found that the two are significantly related ($R^2 = 0.25$, $p < 0.001$). However, when the adjustment factor (the ratio of Waites Run nitrate-N concentration over Cullers Run nitrate-N concentration taken at the same time) is regressed on discharge this relationship is statistically insignificant ($R^2 = 0.03$, $p < 0.23$). Including a seasonal dummy variable to account for effects of growing season has very little impact on this result. Our conclusion is that the adjustment factor does focus the incentive on farmer actions by reducing the influence of fluctuating background nitrate-N on the payment.

Comparing estimated with actual measurements

Below we look at the accuracy of our estimates, and how successful the payments have been in inducing Cullers Run farmers to abate nitrate-N. Figure 2 compares estimated versus actual values for the discharge, adjustment factor, and monthly payment based on the first six months of the experiment. Estimates are averaged monthly estimated values for each measure based on four years of data, while actual values are based on direct measurements of watershed discharges and nitrate-N loads. We see that actual

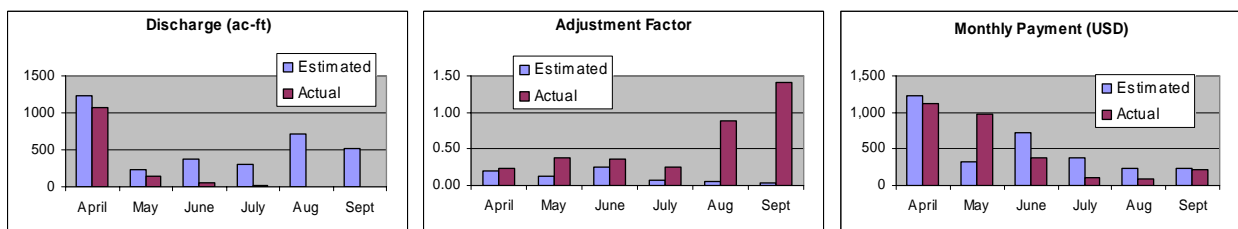


Figure 2. Comparison of actual and estimated from the watershed payment.

discharge and the adjustment factor in April are very close to the estimates. Since April, dry conditions have prevailed in the area. For example, for the five months following April, discharge of Waites Run was substantially less than the mean discharge for those months between 2002 and 2006 (USGS website). These dry conditions are partly responsible for the actual adjustment factor being much larger than the estimated factors. In the payment calculations, discharge adjusted prices and the adjustment factor have prevented large drops in payments. Total payments to-date have deviated from estimated payments by 7.5%.

Farmer participation

As mentioned earlier, estimated payments need to encourage farmers first to participate and then to abate nitrate-N. Currently, fifteen farm households have decided to participate in this project. These households own or operate approximately 41% of the agricultural land in the watershed. However, this land is not evenly distributed throughout the watershed. Cullers Run watershed can be divided into two main sections. The lower section is where most of the row cropping takes place. In this section, about 10% of agricultural land is within the project. In the upper section, hay fields and pasture predominate. Participating farmers operate on 49% of the land in this section. In Maille and Collins (forthcoming) we argue that for farmers in the lower section, participation entails more risk because they rely more on fertilization. Therefore, these farmers are participating at lower rates.

Looking at nitrate-N abatement, a detailed sampling run in April showed three locations where nitrate-N contributions were highest on a per-unit area basis. The farmer responsible for one of these has successfully sought cost-share support to undertake conservation measures although it is not certain he did this in response to the water payments. Another of these areas is the lower section of the watershed. This section is farmed by multiple households, only one of which is currently participating. This complicates abatement actions. Participating farmers are currently trying to recruit more of the farmers in this section into the project, and they are seeking additional water quality data to help focus abatement efforts.

CONCLUSIONS

Although it is still early in the experiment, we are encouraged by the results. The payment formula seems to have conveyed the appropriate incentive as a significant portion of farm households in the watershed are participating and there has been some nitrate-N abatement. Two important challenges have also presented themselves. First, water quality data indicate that the area most in need of nitrate-N abatement is the floodplain which is farmed by multiple households. Their participation or cooperation with the project is crucial to achieving meaningful nitrate-N abatement. In addition, we need to better understand the reasons behind the divergence between the expected and actual values for the adjustment factors presented in table 2. We expect continued water quality and flow data will help us in this respect.

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INDUCING FARMER PARTICIPATION IN A WATERSHED LEVEL PROGRAM TO IMPROVE WATER QUALITY

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KEY WORDS: agricultural non-point pollution, team approach

ABSTRACT

This paper describes part of a field experiment involving performance-based economic incentives to improve water quality that is being conducted on a small 3,000 ha watershed in Hardy County, West Virginia. It serves as a companion paper to Maille and Collins (2007) which describes development of the economic incentives. To effectively address agricultural non-point pollution in this experiment, farmers need to utilize a team approach based on voluntary participation, group interaction and decision-making. In this paper, the logic of a team approach is explained and farmer recruitment strategy is discussed. Elements of successful recruitment include having sufficient water quality data to provide evidence of a problem, conducting informational meetings with immediate benefits to attendees, creating partnerships with local organizations and elites to build trust among farmers, assisting group decision making with an advisory committee, and development of clearly written contractual provisions with numerous review and feedback opportunities provided to farmers. To date, participation of about one-half of the farmers in the watershed has been achieved.

INTRODUCTION

The current approach to addressing agricultural non-point pollution (ANP) is focused on voluntary conservation measures that are implemented by farmers with cost-share assistance and technical support from the government (Ribaud *et al.* 1999). These conservation measures are generally directed by federal agencies to conform to strict behavioral guidelines in order to receive the assistance. Some states are moving towards increased regulatory control of agricultural operations. For example, the state of Maryland, under the Water Quality Improvement Act of 1998, has implemented a requirement of nutrient management plans on all farms. In either case, farmer input into water quality improvement strategies is limited.

As total maximum daily load planning and implementation show, water quality represents a watershed-wide problem that can not be solved one discharger, landowner or farmer at a time, but within the context of an entire watershed. Thus, coordinated action at a watershed level is required. However, in order to induce a coordinated, team approach to ANP among farmers,

economic or regulatory incentives are required. A companion paper (Maille and Collins 2007) explores the development of performance-based payments at the watershed level that provide economic incentives to farmers. These payments provide an opportunity to change farmer perspectives of water quality conservation from an operational constraint to an income generating opportunity. In order to be effective, however, watershed level payments need interaction and decision-making among a group of voluntary participants within the watershed (i.e. a team approach). The objectives of this paper are to explain why a team approach is needed for ANP and to discuss the essential elements utilized for recruitment of farmers into acting as a team in a field experiment setting.

The literature on team approaches to water quality has examined theoretical and laboratory experiment aspects (see Spraggon 2002, Romstad 2003, Poe *et al.* 2004, Taylor *et al.* 2004, Millock and Salanie 2005, and Vossler *et al.* 2006). In contrast, this research involves a field experiment using actual farmers and real monetary payments. Therefore, there are real consequences for farmers from the research outcomes. Thus, the recruitment aspect of this paper covers an unexplored area in previous research on water quality economics.

The study site for this field experiment is Cullers Run watershed. This stream is a tributary of the Lost River in the eastern panhandle region of West Virginia that occupies 2,978 hectares in West Virginia's largest poultry production county. Sixteen percent of the watershed is devoted to agriculture, mostly pasture or hay land. Row crops comprise only 3.63% of the agricultural land, primarily in the floodplain (Cacapon Institute 2002). The rest of the watershed is forest. There are approximately twelve poultry houses conducting intensive poultry production in the watershed. Most agricultural fertilizer use in the watershed is provided by poultry litter.

The Cullers Run watershed has advantage of being small enough to limit the number of farmer households that could participate in the project. Small group size reduces the information burden on farmers (see Ribaudo *et al.* 1999). In addition, this watershed was included in federally-funded research project that generated water quality data prior to the experiment (Cacapon Institute 2002).

WHY A TEAM?

Figure 1 provides a schematic description of technical assistance and cost share that have been traditionally provided by the government for ANP. Acting as a principal, the government provides information and cost share opportunities to farmers and treats farmers as autonomous units with regard to their impacts on water quality. Farmers, acting as individual decision making units, decide whether or not to undertake water quality protection actions. Only a portion of farmers (C and F in Figure 1) decide to undertake these actions. These actions lead to water quality outcomes which do not directly relate back to the recipient of government cost share or technical assistance.

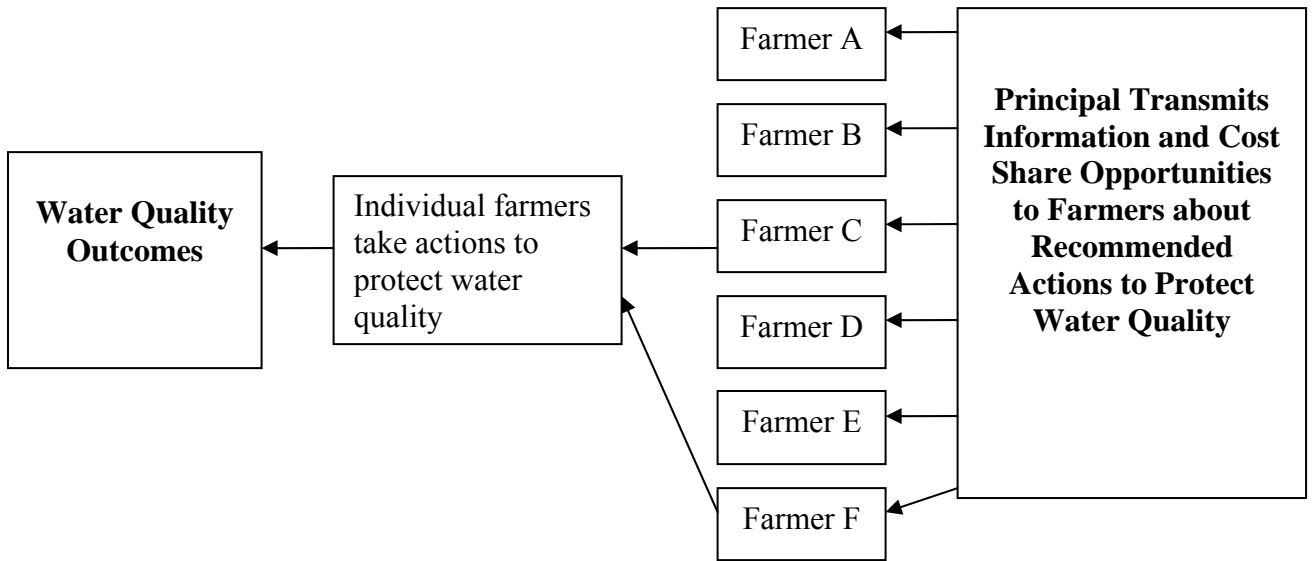


Figure 1. Cost share approach to water quality protection.

There are motivations, however, for both the principal and for farmers to engage in a team approach. From the perspective of the principal acting for society as a whole, there are four main motivations: (1) non-point pollution is difficult and costly to identify from individual sources so it is more feasible from an informational perspective to measure pollution at a watershed basis; (2) farmers have information about their own and/or their neighbors' pollution contributions that regulators would have difficulty obtaining; (3) a potential for dealing with fewer entities; and (4) teams allow for farmers to influence the water quality protection behavior of other farmers through the use of moral suasion. From a farmer perspective, watershed level decision-making is a reasonable approach to problem solve if farmers perceive a water quality problem from ANP pollution. Romstad (2003) describes motives for farmers to join a team approach under the threat of more costly individual regulation.

The economic incentives provided for team participation can be either negative (threat of individual regulation) or positive (provide a subsidy to participate). Threats imply that society operates from a polluter pays principle and/or that citizens have a right to clean water. Subsidies imply the property right to clean water lies with the agricultural landowner. A combination of negative and positive incentives can be used when water quality standards are implemented with penalties for pollution in excess and subsidies provided when pollution levels are less than the standard. In our field experiment, positive economic incentives were used.

Figure 2 depicts the principal's role under a team approach using positive economic incentives. Here the principal interacts with farmers as a team, thereby recognizing that interconnections exist between farmers in how they impact water quality. The team decides which actions to take regarding water quality and then the water quality outcomes are reflected in the watershed level payments made to participating farmers. As shown in Figure 2, farmers can be interconnected in terms of their water quality protection actions as Farmer C impacts Farmer B and D's water quality protection actions.

The farmer interactions presented in Figure 2 occur in the Cullers Run watershed in terms of poultry litter used for fertilizer. In this example, assume Farmer C is a poultry grower without sufficient land base to utilize the nutrients in litter. To dispose of litter, Farmer C transfers litter to Farmers B and D (with or without cash compensation) to apply on their agricultural land. In this case, team decision making at a watershed level has the potential to improve the prospects of individual level decisions that protect water quality. Since our field experiment involves nitrate-N pollution impacting a watershed level payment distributed to farmers (Maille and Collins 2007), under a team approach Farmer C has an incentive to encourage Farmers B and D to determine if the litter applied contains nutrients in excess of crop needs. If excess nitrate is being land applied, this could reduce the watershed level payment to all participating farmers. Thus under excess land application, Farmer C would be better off to sell a portion of his litter out of the watershed rather than only transferring it to Farmers B and D. In our field experiment, Cullers Run farmers are presented with such an opportunity to work within a team approach to investigate issues related to land application of litter.

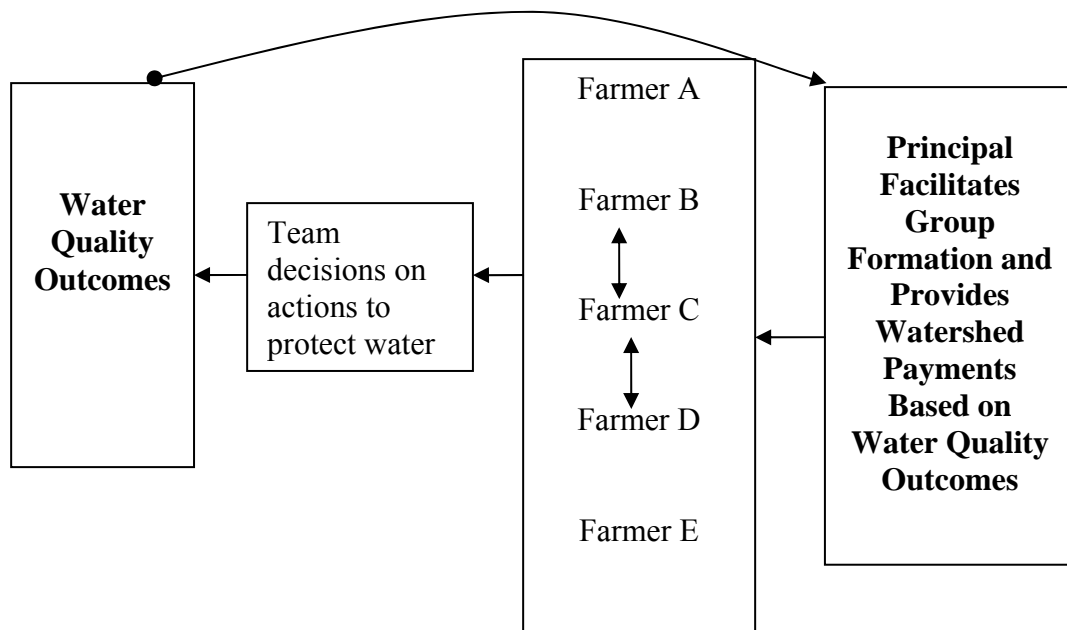


Figure 2. Team approach to water quality protection.

METHODS

In December 2006, we began to advertise and inform local farmers about this field experiment project. These efforts included: a presentation at the Mathias Ruritan Club (many of the watershed residents are members of this community service organization), articles in the community newspaper about the project, and a letter sent to all agricultural landowners in the experimental watershed inviting them to an introductory meeting.

During winter of 2007, an introductory meeting (February 5th) and three additional meetings (February 19, March 5th and March 26th) were organized by project researchers. From the beginning, it was important to engage local community resources in these meetings and to provide inducements for attendance. We utilized a community center operated by the Mathias

Ruritan Club to hold all the meetings (a monetary donation was made to the club). At each meeting, a dinner was provided for the attendees. We hired a local Future Farmers of America organization to prepare these dinners. Aerial photo maps of the watershed were prepared by the Natural Resource Analysis Center at West Virginia University. These maps were displayed and copies were given out to all attendees. These maps proved to be very popular among attendees.

Each of these meetings was attended by twenty to thirty people. Overall, a substantial portion of farmer households in the watershed attended at least one meeting. We also made efforts to involve community elites in the project. The county extension agent attended a meeting and a county commissioner was recruited to lead one of the meetings. State and federal government conservation agency personnel attended meetings and assisted with presentations.

During these meetings, the project was described as a unique field experiment involving economic incentives to abate ANP. These incentives would take the form of monthly payments over a period of two years based on the quantity and quality of water flowing from the watershed to farmers who chose to participate. The introductory meeting included a presentation about water quality as an issue in Cullers Run and the Lost River watershed in general. This presentation was made by this paper's co-author from Cacapon Institute who has extensive experience with water quality issues in the region.

There were two important outcomes from these meetings: (1) a written contract was created with the input of farmers; and (2) a farmer advisory committee was established to determine how to allocate the watershed payments among participants.

The contract was discussed and revised a number of times during the meetings. It served to clarify the institutional framework of the experiment and outlined the roles and responsibilities of both farmers and researchers. Comments and suggestions about this contract were obtained from a lawyer on the Agricultural and Resource Economics faculty at West Virginia University. Key stipulations for farmers in this contract included:

- Participation in this project is voluntary and is initiated by signing a contract.
- A participant who has signed a contract can choose to leave the project at any time with no penalty or further obligation.
- Payments will be made monthly to 'The Group'. The initial participants will determine how these monthly payments are allocated among the participants. The resulting allocation rules will be presented to the project investigators, who will use these rules to distribute the monthly payments and be responsible for disbursements.
- Participants are allowed to be enrolled in state or federal cost-share programs.
- A participant is able to select which best management practice (BMP) or other management change to implement in order to impact water quality.
- Signing a contract does not obligate a participant to implement any BMP.

Farmers, along with researchers, have difficulty projecting the amount and timing of pollution reductions resulting from BMP implementation (Park *et al.* 1994; Bracmort *et al.* 2006). Thus, risk reduction aspects of this contract include the voluntary aspects of participation, BMP selection, and BMP implementation. The two year time frame of the experiment may limit

farmer interest in BMP implementation so cost share participation is encouraged. In addition, participants have the ability to control for risk by allocating more of the monthly payments to those farmers that implement BMPs.

Key stipulations for project researchers included:

- Responsibility for determining if a potential participant qualifies for participation in the project based solely on the boundary of Cullers Run watershed.
- Participation is not voluntary. There are no provisions that permit withdrawal of project funds.
- Calculation of the amount of each monthly payment to ‘The Group’ and distribute this payment among participants based on written allocation rules provided by ‘The Group’. The amount of the payment will be computed by project researchers based on a payment formula and prices presented to the farmers.
- Setting up a water quality and quantity sampling, monitoring, and testing plan. Participants will be allowed to observe any sampling, monitoring, and testing being conducted under this plan.

Creation of an advisory committee emerged as a suggestion by farmer attendees during the March 5th meeting that was led by a county commissioner. The purpose of this meeting was to assist farmers in determining how they would organize themselves as a group in this project. This advisory committee consisted of five farmers from the watershed, each who had attended most of the meetings. The committee met only once on March 12th. This committee made recommendations that were then presented at the March 26th meeting and formally approved at the first meeting of project participants on April 16th.

RESULTS

Farmers were able to sign a written contract to participate in the project beginning April 1, 2007. To date, a total of fifteen farm households have signed a contract. This sign-up represents about one-half of the farmers who attended the series of meetings introducing the project to farmers. Using farmer reported agricultural use information as well as aerial photo data, we computed that participating farm households own or operate approximately 41% of the agricultural land in the watershed.

Participation is not evenly distributed throughout the watershed. Cullers Run watershed can be divided into two main sections. The lower section is where most of the row cropping takes place. In this section, about 10% of agricultural land is farmed by a project participant. In the upper section, hay fields and pasture predominate. Participating farmers operate 49% of the agricultural land in this section. Based on participating hectares, a simple Chi-squared test of independence indicates that the likelihood of a given hectare of land being included in the project is not independent of location ($p < 0.01$). Our interpretation is that farmland in the lower section is significantly less likely to be enrolled in the project than farmland in the upper section. Maille and Collins (forthcoming) surmise that lower participation rates for farmers in the lower section is the result of the greater risk to agricultural production that they face when participating in the

project because they fertilize at higher rates with poultry litter than do farmers in the upper section.

As a group, participating farmers have made two important decisions: (1) allocation of watershed payments; and (2) a request for a watershed-wide sampling to verify source areas of nitrate-N. As suggested by the advisory committee, their innovative payment allocation involved: (a) a \$50 signing bonus to each participant who signed up prior to June 1st, 2007, (b) 10% of each monthly payment is to be distributed equally among all participants, (c) the remaining 90% is reserved to financially assist farmers who engage in N-nitrate abatement, and (d) any remaining funds at the end of the year are to be paid out as a bonus to all participants. This allocation provides early participants with immediate rewards and addresses issues of risk from BMP implementation by individual farmers and provides immediate rewards for participation. The results of the watershed-wide nitrate-N sampling were presented to farmers at a June 2007 meeting. These results agreed with prior water quality data that showed the majority of nitrate-N originated from the lower section of the watershed. Dry conditions on the watershed prevented further watershed-wide sampling, but more are planned for the future.

CONCLUSIONS AND FUTURE DIRECTIONS

We are encouraged with the sign-up results to date. About one-half of the agricultural land in the watershed and about one half of farmers attending the meetings are participating in the experiment. Payments for water quantity and quality are being made to farmers based on a payment allocation scheme that they developed and approved. At the farmers' request, detailed water quality sampling of the watershed has taken place. To facilitate information sharing between researchers and farmers, we have established a project website (<http://www.cacaponinstitute.org/wvunri.htm>). To date, no group decision has occurred with respect to cost sharing from project funds for ANP abatement.

Given their low participation rate, a practical research question involves how to bring farmers in the lower section into the field experiment. This may provide a rationale for allocating a greater share of the payments to farmers in the lower section. We can also investigate the role that information on soil nutrients may play in determining farmer response. For example, Feather and Amacher (1994) find that information can increase farmer willingness to adopt BMPs. A potentially relevant example is presented by Fuglie and Bosch (1995). They determined that corn farmers decreased fertilization rates when provided with information from soil tests indicating that they could fertilize less without introducing additional production risk. To address these issues, participating farmers have discussed reviewing existing or creating new nutrient management plans for all farms throughout the watershed. Currently, participating farmers are seeking to persuade lower section farmers to participate in the project.

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**THE OPEQUON CREEK TARGETED WATERSHED PROJECT – A STATE
PARTNERSHIP TO BENEFIT THE CHESAPEAKE BAY**

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ABSTRACT

Opequon Creek begins south of Winchester, Virginia, and flows north through the West Virginia panhandle past Martinsburg to the Potomac River. This northern end of the Shenandoah Valley has been an agricultural area of orchards, pastures and cropland. The I-81 corridor and proximity to Washington are contributing to active commercial and residential development. Water quality assessments by both states have identified bacterial, benthic, and biological impairments, with sediments and nutrients identified as likely causes of degradation. A comprehensive, watershed-based approach is seeking to reduce nutrient loads from the watershed. Floodplain wetlands, pocket wetlands, and water quality swales will be designed to site-specific parameters to mitigate nutrient loads in runoff from urban and agricultural lands. An outreach program to encourage a flexible stream fencing program will be implemented with assessment of effectiveness. Water quality is monitored at stream locations throughout the basin to assess land use impacts on nutrient loads and effectiveness of specific practices in improving water quality. The goals of the project are to: a) develop, implement and evaluate the effectiveness of BMPs; b) develop creative, sustainable solutions to increase the adoption of both innovative and traditional BMPs; c) develop a point/nonpoint water-quality offset program for the public service authority to demonstrate and evaluate implementation and practice issues associated with nutrient trading programs; and d) develop a comprehensive nutrient reduction plan that addresses issues and opportunities for accelerating nutrient reduction in a complex urbanizing watershed.

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**STONE SOUP: THE REDISCOVERY OF LANDSCAPES AND COMMUNITY,
INTRODUCING LAND CARE IN GRAYSON COUNTY, VIRGINIA, USA**

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ABSTRACT

To protect open spaces and water quality, the New River Land Trust has organized landowners, federal and state agencies, businesses, cattle producers association, soil and water districts, state agricultural university, and local government to increase the incomes of landowners through improved management of pastures and forests. Financial returns to family owned agriculture are minimal when compared to other investment opportunities hastening the departure of young who, not infrequently, sell family land for more profitable investments elsewhere. Land buyers from nearby cities are ever present interested in rural retreats, recreation properties along the river, and mountain hideaways in the forests. The primary tax resource for rural local government is productive farm and forested lands to fund schools and other essential services making it increasingly difficult for farm families to financially cope. In meetings organized by the land trust, it was decided that the example set by the Land Care movement in Australia was practical for the people in Grayson County. The paper traces the emergence of a Land Care movement over 18 months. The approach is based upon building a culture of stewardship for community, health, landscapes, watersheds, native species and local economy.



Session VI-B

Research Experiences of Undergraduates (Part I)

- Solubility as a Mechanism for CSMR Effects on Lead Leaching
- Cement Mortar Linings Affect Drinking Water Quality
- Managing Manganese in Drinking Water: An Assessment for Microbes and Metals
- Determination of the Taste Threshold of Iron in Water

* * * * *

SOLUBILITY AS A MECHANISM FOR CSMR EFFECTS ON LEAD LEACHING

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ABSTRACT

The solution chemistry of Pb(II) at relatively low pH ($\approx 3-5$) was examined in the presence of chloride, sulfate, and phosphate. Lead sulfate solids are relatively insoluble even at pH 3. Pb(II) forms soluble complexes with Cl⁻ that can significantly increase the solubility of lead. Since lead solubility can be correlated to lead contamination problems in drinking water, the data suggest that chloride will worsen lead problems at low pH, whereas sulfate will decrease lead problems. The contradictory trends for the two anions may provide a mechanistic explanation for the success of the empirical chloride: sulfate mass ratio in explaining lead contamination of potable water. Specifically, the trends explain why lead contamination sometimes worsens when changing from alum (higher sulfate) to ferric chloride (higher chloride) coagulants.

* * * * *

CEMENT MORTAR LININGS AFFECT DRINKING WATER QUALITY

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KEY WORDS: cement, leaching, metals, hardness, alkalinity

ABSTRACT

Cement mortar is commonly employed for in-situ relining of iron pipes in drinking water distribution systems. Relined-distribution system pipes are usually put back into service within 24-48 hours of application of cement mortar. This research comprehensively investigated the effects of newly installed cement mortar on drinking water quality, including leaching of pH, alkalinity, organic matter, metals, and odors. Leaching tests were performed for 30 days and simulated stagnant water in 4" pipes with contact times of 1 to several days. The results indicate that during the initial curing period (roughly one week), the cement had major effects on the water quality. The pH increased from 8 to over 12; sufficient calcium carbonate leached to cause precipitation and alkalinities of several hundreds of mg/L; disinfectant was consumed. Calcium and magnesium also showed significant impacts, with values of over 200 and 70 mg/L, respectively for the initial week of the leaching test. However, once the cement had substantially cured, its effects on the water were drastically reduced and leaching of chemicals from the cement mortar declined sharply and disinfectant levels became stable. Lastly, throughout the test the water was marked by a distinct "cement" odor that had not faded or reduced by the end the test. As most environments utilizing drinking water will be characterized by at least some water

flowing, these results should not be typical of an average user's setting, but provide effective data for modeling the effects relining will have on the drinking water quality.

INTRODUCTION

Although cement mortar lining for the rehabilitation of potable water distribution system infrastructure has been used in the USA and Europe since the 1920's, few studies have evaluated the short term impacts of new rehabilitation on drinking water quality at the tap. Cement mortar linings are extensively used to rehabilitate corroded iron pipes and control red water, which is iron corrosion products or "rust", in people's tap water (Burlingame et al. 2006). Typically, cement mortar is sprayed onto the interior of existing pipes and allowed to cure for 24-48 hours. Upon curing, the pipe is immediately put back into service delivering water to consumers. There have been comprehensive studies related to the long-term inorganic water chemistry effects resulting from the corrosion of the mortar lining material but little work has specifically examined the material's affect on aesthetic water quality or disinfection by-products, particularly during the time shortly after in-situ installation.

Soft waters can be aggressive and may leach metals from cement-mortar pipe walls and linings. Aluminum has been known to leach in significant quantities and pose a health threat to dialysis patients. Chromium, barium, and cadmium are also of concern. Leaching of metals typically occurs under low flow conditions and from newly installed materials for as long as 4 years (Berend and Trouwborst, 1999). Field tests at locations in New Haven, Connecticut and Corte Madera, California demonstrated that leaching of lime from cement-mortar corrosion can significantly impair water quality by raising pH, alkalinity and calcium concentration (Douglas et al., 1996).

Unusual taste and odors in drinking water increase consumer concerns about the quality of the water and also can make them worry about adverse health effects (Dietrich, 2006). Distribution system materials are known to be a major source of off-flavor components to drinking water at the tap (Khiari et al, 2002). However, the comprehensive effects of cement mortar on the aesthetic qualities of drinking water have little documentation.

The purpose of this study was to determine the effects of newly installed in-situ relining with cement mortar would have on the quality of drinking water.

MATERIALS AND METHODS

Cement mortar coupons

Cement mortar coupons for water quality testing were fabricated in accordance with applicable sections of *AWWA Standard C602-06, Cement Mortar Lining of Water Pipelines* and National Sanitation Foundation (NSF) 61-2000a. Coupons were tested according to the industry standard leaching procedure using four different test water types and a coupon surface area to volume ratio that mimicked a 4-inch diameter pipe. The recipe met the requirements of AWWA C602, *Cement Mortar Lining of Water Pipelines in Place – 4" and larger*. Cement mortar was prepared from medium grade, washed sand mixed in a 1:1 ratio with Type II Portland cement and tap

water added equal to 0.43 weight fraction of the cement weight. The cement was poured into prepared molds consisting of a plywood base that was covered with a thin Teflon sheet upon which was built a Teflon tape wrapped-wooden frame structure that was one centimeter high and produced fifty 8cm x 8cm x 1cm coupons. The design allowed for easy filling and mortar leveling during the fabrication process. The cement mortar coupons were covered with one layer of polyethylene sheet to preserve moisture and cured in a humid room for 24 hours.

Leaching tests

The leaching protocol was adapted from *ANSI/NSF 61* which evaluates the leaching of organic and inorganic components from materials in contact with drinking water.

Test waters

Four water types were tested so as to investigate the different disinfectants and pH values that are typical of USA drinking waters. The low alkalinity/low hardness reference water contained 3.3×10^{-4} M MgSO_4 , 6.77×10^{-4} M NaHCO_3 , 1.45×10^{-4} CaSO_4 , 1.41×10^{-4} M CaCl_2 , 9.15×10^{-5} M Na_2SiO_3 , 9.9×10^{-5} M KNO_3 , 2×10^{-4} M HCl . This reference water was amended with sodium hypochlorite to produce chlorinated water or sodium hypochlorite plus ammonia to produce chloraminated water and the pH adjusted to prepare these four test waters: Water 1 - pH = 8.0 with no disinfectant; Water 2 - pH = 6.5, 2.0 mg/L chlorine; Water 3 - pH = 8.0, 2.0 mg/L chlorine; and Water 4 - pH = 8.0, 4.0 mg/L monochloramine.

Water quality analyses

Three all glass test vessels of approximately 3 L volume and containing six cement mortar coupons were prepared for each water type. The ratio of water to surface area replicated a 4-inch diameter pipe. A control was prepared for each water type that contained no cement mortar coupons. All vessels were maintained in the dark and headspace-free at 21-23 °C throughout the experiment. Odor intensity levels were obtained using Flavor Profile Analysis (Standard Method 2170) on samples obtained on days 1, 4, 9, and 14; the human subjects protocol was approved by the Virginia Tech Institutional Review Board. The water in each of the sixteen test vessels was sampled at days 1, 2, 4, 8, 10, 14, 15, 19, 24, and 30. The water was completely drained and fresh water was added to each vessel after samples were obtained. The following analyses were performed:

- pH by Standard Method 2310 using an ion selective electrode;
- Alkalinity by Standard Method 2320 using titration;
- Disinfectant residual by Standard Method 2350 by DPD colorimetric method;
- Total organic carbon (TOC) by Standard Method 5310C using a Sievers® 800 Portable TOC analyzer;
- Ammonia (chloraminated samples only) by Standard Method 4500 using colorimetric method;
- Metals (including Ca and Mg for hardness) by Standard Method 3125 using Inductively Couple Plasma-Mass Spectrometry;
- Trihalomethanes by Standard Method 6200 using purge and trap GC with ECD detector.

RESULTS AND DISCUSSION

Water quality parameters

pH

The initial water quality condition – either pH 6.5 or 8 with no disinfectant, chlorine or monochloramines- did not affect the final pH after the cement mortar coupons were placed in the test vessel (Figure 1). The control waters, which were test vessels without cement mortar coupons, were able to maintain their target pH values of 8 or 6.5. This is similar to results reported by Douglas et al. in 1996. As shown in the Figure 1, the pH for the all four test water conditions increased from 6.5-8 to 10.8-12.4 during the leaching tests. pH was measured on days that the water in the vessels was changed; thus, the length of time that the coupons were in contact with the water is reflected by the days between water changes. For sampling days 1, 2, 4 and 9, the pH was consistently about 12. The water change on day 11 (48 hour contact time) and four previous water changes resulted in a pH drop to about 11.6. After a contact time of 72 hours, when the water was changed on day 14, the pH was still about 11.6. When the water was changed 24 hours later on day 15, it declined sharply to about pH 10. This is likely due to a combination of the short contact time, the previous leaching, and curing of the cement.

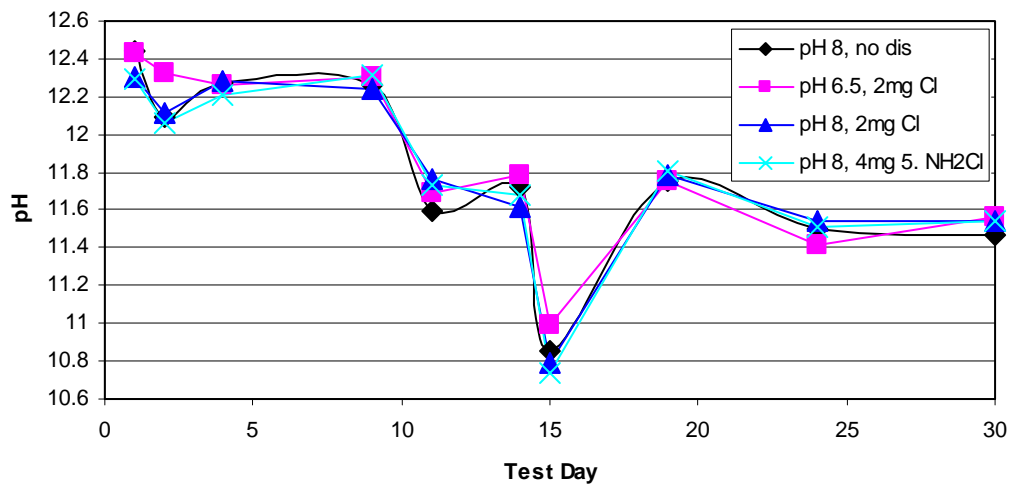


Figure 1. pH as a function of time for all four test waters in contact with cement mortar coupons.

Alkalinity

The target concentration for the reference water was 35 mg/L and the average for the all the controls for all sampling days was 35.5 mg/L, indicating excellent agreement between prepared and measured concentrations (Figure 2). Again, these data mimic that reported by Douglas et al. in 1996. Wispy gray solids initially precipitated in all test waters when cement coupons were present. These solids were predominately composed of calcium and carbonate; they settled onto the bottom of the containers indicating that they were denser than water. The solids were no longer present after the cement's substantial curing (day 9). Similar to the pH data, there were no

impacts from the initial water quality condition – either pH 6.5 or 8 with no disinfectant, chlorine or monochloramines- on the amount of alkalinity released from the cement mortar. All water conditions performed similarly and had alkalinities near 600 mg/L for the first days of the leaching tests. Similar to the pH data, there was a substantial drop in alkalinity between the water changes on days 9 and 11. After this time, the alkalinity leveled off and remained about 100 mg/L for all four test waters.

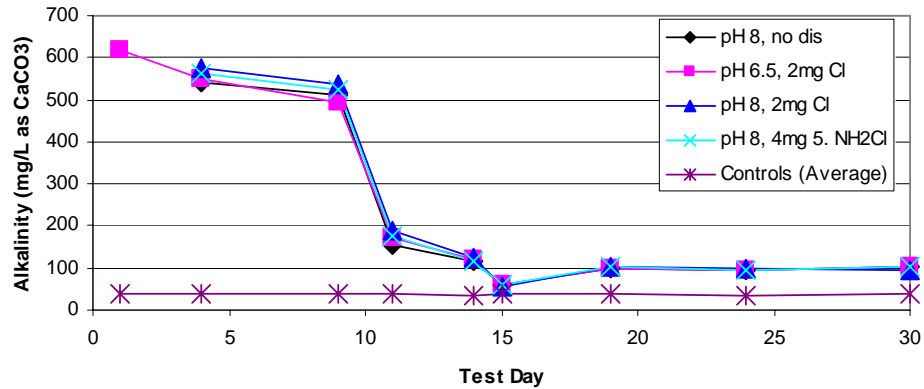


Figure 2. Alkalinity as a function of time for all four test waters and the controls.

Hardness

The data below are typical of those observed for the metals released from new cement mortar coupons for all pH and disinfectant combinations of the experiment (Figure 3). As was seen in Douglas et al., 1996, the concrete mortar contributes a significant amount of calcium and magnesium into the water. The calcium concentration is much greater than the magnesium concentration and in the range of hundreds of mg/L for the first few days. However, after the concrete cured substantially (around day 9), the concrete actually began to sorb calcium, as the values were below that of the controls. The magnesium behaved differently, and is consistently sorbed by the concrete before or after substantial curing. The controls contained calcium and magnesium as these cations were added in the reference water at 12mg/L and 0.8 mg/L, respectively.

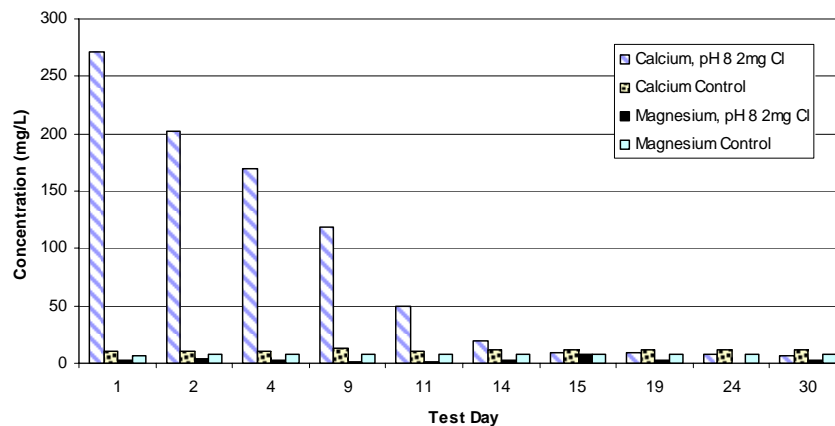


Figure 3. Calcium and Magnesium as a function of time for water with pH 8 with 2 mg/L Cl₂ (water 3).

Metals

The data below are typical of those observed for all test conditions of the experiment (Figure 4). Of the metals tested, chromium and aluminum were consistently higher than their controls relative to the other metals (sodium, phosphorus, sulfur, chlorine, iron, manganese, nickel, zinc, potassium, and copper). Aluminum in the initial period exceeded the EPA secondary standard of 0.2 mg/L (EPA, 2003). As with hardness, there is a significant changeover when the concrete cured substantially (day 9), after which the quantities of aluminum and chromium became negligible and within limits (though still slightly higher than the corresponding controls).

Loss of disinfectant

Figure 5 below shows the disinfectant loss in mg/L-day for Water 3 (pH 8, 2 mg/L Chlorine) and Water 4 (pH 8, 4 mg/L Monochloramine). Water 2 (pH 6.5, 2 mg/l chlorine) demonstrated a trend similar to Water 3 and is not shown. Water 1 contained no disinfectant and thus would produce no relevant data. Data for the controls are shown as both chlorine and monochloramine and are acknowledged to decay spontaneously even without the presence of cement mortar. The decay rate for both chlorine and monochloramine was enhanced by the presence of cement mortar. This higher rate of decay was maintained until test day 9, after which the cement mortar performed more similarly to the controls. This became true across the board, such that there was no longer a significant impact from the cement on the disinfectant. This pattern is similar to that observed for pH and alkalinity. Water 4, which was dosed with 4 mg/L monochloramine at pH 8, demonstrated no difference in ammonia concentrations and thus cement mortar had no noticeable effect on monochloramine decay.

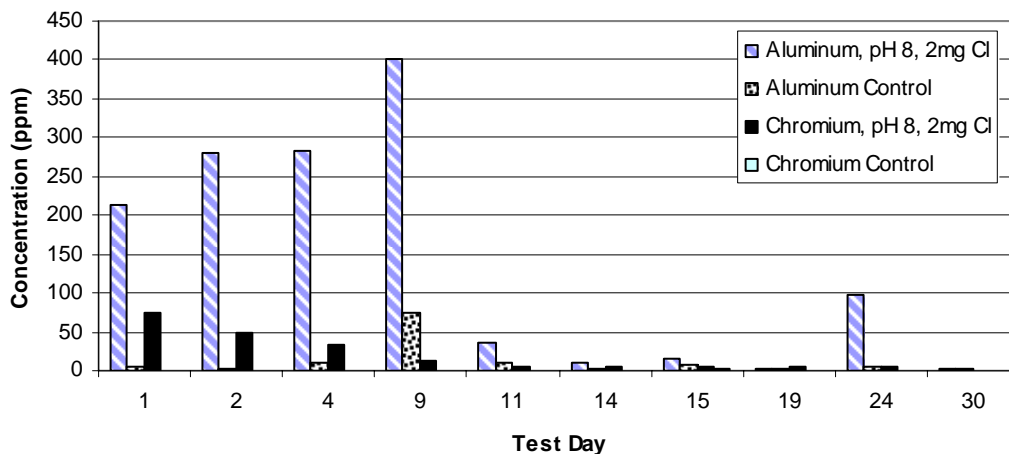


Figure 4. Metals as a function of time for pH 8 with 2 mg/L Cl₂ (water 3) and its control.

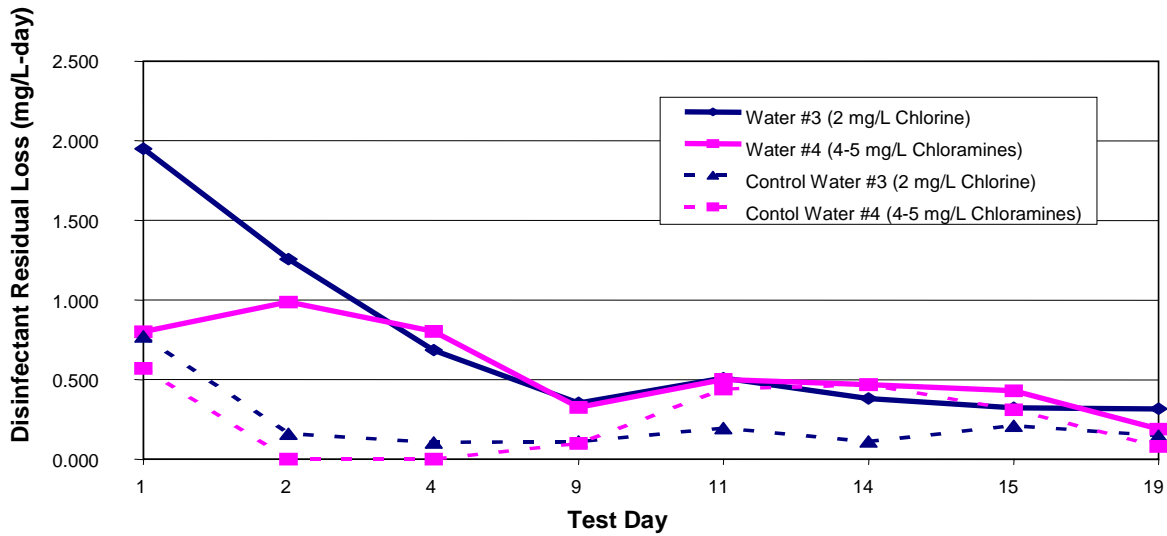


Figure 5. Loss of residual on based on contact time of cement mortar with disinfectant.

Total organic carbon

Data for the TOC are shown in Figure 6. The background level of TOC fluctuated between 0.2 and 0.4 mg/L with a mean of 0.26 mg/L. These concentrations are typical of reagent water produced for laboratory deionization/carbon filtration systems used to produce purified water. The cement mortar coupons initially contributed about 0.5-1 mg/L TOC to the test waters through day 9 of the experiment, and then leveled off to background levels similar to the controls.

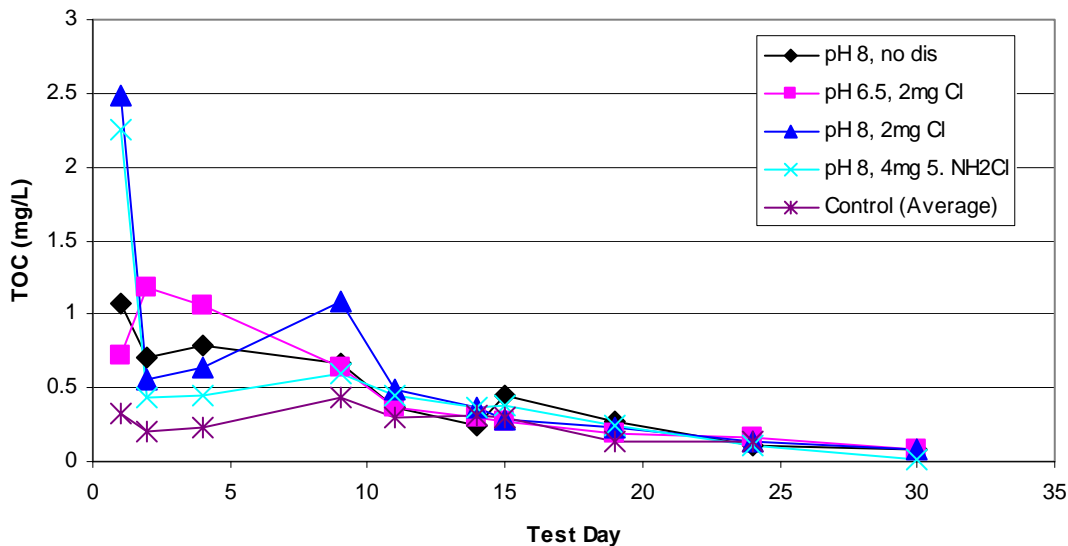


Figure 6. TOC as a function of time for water type in contact with cement mortar.

Odor

Water samples were evaluated for odor by 3-4 members of a human panel trained in Flavor Profile Analysis. A noticeable “cement” odor was imparted to the water by the cement mortar coupons (data not shown). The intensity of this cement odor was in the 3.5 - 4.5 on the FPA scale of 0-12. This is a “weak” intensity (defined as FPA intensity = 4); the weak intensity corresponds to the sweetness of canned fruit (for comparison, FPA intensity = 8 = moderate and corresponds to canned soda and FPA = 12 = strong and corresponds to syrup or jelly). The odor intensity did not diminish over the 14 day period of that data were collected. The control waters were generally odor free with an occasional threshold (1) or 2 rating. The cement odor intensity or description was not impacted by the presences of chlorine or chloramine. On Day 1, nearly all the residual disinfectant had been consumed in the samples. The sensory panelists detected a chlorine odor (data not shown) on Day 14 in Waters 3 and 4, which were dosed with 2 mg/L free chlorine or 4 mg/L monochloramine, respectively. This is consistent with the chlorine residual loss data (Figure 5) which shows that the chlorine and monochloramine were being consumed by the cement mortar early in the test but had leveled off by day 14.

CONCLUSIONS

As stated earlier, there was a marked decrease in water quality effects after the cement substantially cured. Prior to curing substantially, many water quality parameters were higher values than desired, especially with regard to dissolved solids, pH, and aluminum where the EPA secondary standards were exceeded. Upon curing substantially, these effects were reduced, though pH was still well above its EPA guidelines. Other tests, such as TOC, showed a gradual decrease as time progressed independent of curing. All data indicate that the effects on water quality of in-situ relining with cement mortar were most pronounced during the initial days, declining thereafter. However, there was a distinct odor of ‘cement’ that persisted throughout all testing, despite the curing of cement and general decrease in all other water quality parameters. This odor did not diminish or disappear by the time testing had ended, which indicates that in-situ relining would have distinguishable effects on the perceived quality of the water. In-situ relining provides an effective means to extend the use of failing metal pipes in the long term, though it has a significant, immediate impact on the quality of the drinking water.

ACKNOWLEDGEMENTS

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MANAGING MANGANESE IN DRINKING WATER: AN ASSESSMENT FOR MICROBES AND METALS

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ABSTRACT

Manganese can be released and cause aesthetic problems in the drinking water systems worldwide. The relationships between metals and microbes; at the sediment-water interface; have already been established to assess microbiological implications in natural water systems. Studies have been conducted on the oxidation and reduction of manganese by microbes, but their implications in drinking water systems have not been understood thoroughly. Five microbial strains that were recovered from the filtration and sedimentation basins of a water treatment plant in Blacksburg, Virginia were selected to conduct this research. The five selected strains were found to be capable of performing oxidation of Mn (II) to Mn (IV) when inoculated in an Mn-oxidation selective broth medium. Mn (II) present at concentrations equal or less than 0.05 mg/l in the broth medium used for this study had no effect on the growth of the five strains evaluated. Mn (II) inhibited growth of these strains when present at a concentration of 0.08mg/l. The results from this research suggest that microbial manganese oxidation takes place in drinking water systems and that it could have an effect on manganese release and deposition. This research also serves as a model for the proper investigation of other metal contaminants, which would improve the principles of the water treatment systems and ensure quality drinking water to consumers.

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DETERMINATION OF THE TASTE THRESHOLD OF IRON IN WATER

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ABSTRACT

The effect of iron on human health is a relevant issue in the food and beverage, and the drinking water industries. The impact of iron taste is important as many food and beverage products are being fortified with iron. Iron can also be found in drinking water as a naturally occurring mineral in groundwater or resulting from the corrosion of metal piping in the water distribution system. This study determined the taste threshold for ferrous iron at near neutral pH. This is of particular importance for the drinking water industry, which provides water at or near pH 7.0 and attempts to avoid customer complaints from aesthetic problems. A one-of-five test was used to determine a threshold with solutions containing 0.002 – 5 mg/L of Fe^{2+} in Nanopure water. Individual thresholds for iron ranged from 0.003 mg/L to 5 mg/L Fe^{2+} with population thresholds calculated by geometric mean and logistic regression being 0.052 and 0.031 mg/L respectively.



Session VI-C

New Insights into Measuring Streambank Erosion

- Evaluation of Continuous Turbidity Monitors for Measuring Streambank Contribution to Suspended Sediment Load
- Use of Capacitance Sensors for Continuous Suspended Sediment Concentration Monitoring
- Comparison of Methods for Measuring Streambank Retreat
- Urban Channel Erosion Quantification in Upland Coastal Zone Streams of Virginia, USA

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EVALUATION OF CONTINUOUS TURBIDITY MONITORS FOR MEASURING STREAMBANK CONTRIBUTION TO SUSPENDED SEDIMENT LOAD

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ABSTRACT

In their 2000 report on national water quality, the US EPA cited sediment as the leading cause of water quality impairment in rivers. Excessive sediment reduces reservoir capacity, increases demands on water treatment plants, and reduces the quality of benthic stream communities. Turbidity is increasingly used to quantify suspended sediment concentrations in streams. The goal of this study is to determine if turbidity sensors can be used to measure the contribution of streambank retreat to suspended sediment concentrations (SSC) and flux. Two monitoring stations were established at each end of a 450-meter pastoral reach of Stroubles Creek, a small headwater stream that is experiencing rapid bank retreat. At each station, two Eureka Manta multiparameter sondes with McVan wiped turbidity probes were installed. One sonde was suspended 10 cm below the water surface, while the other was fixed 10 cm above the stream bed and included a pressure transducer to record stream stage. Continuous records of discharge and SSC were computed by relating stage and turbidity to field measurements over a range of storm events. Estimates of sediment loading due to streambank retreat will be computed as the difference in sediment loading between the upstream and downstream stations. These estimates will be compared to two direct methods of measuring streambank retreat: erosion pins and topographic surveying. If successful, results of this project will provide a tool for continuously measuring sediment loads from streambank retreat. This increased temporal resolution could lead to new insights on the processes involved in streambank retreat.

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USE OF CAPACITANCE SENSORS FOR CONTINUOUS SUSPENDED SEDIMENT CONCENTRATION MONITORING

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KEY WORDS: sediment transport, suspended sediment concentration, capacitance meter, erosion

ABSTRACT

According to the US Environmental Protection Agency (USEPA) sediment is a leading cause of water quality impairment (US EPA, 2002). The annual costs of sediment pollution in North America alone approach \$16 billion (Osterkamp et al, 1998). Due to the large spatial and temporal variations inherent in sediment transport, the measurement of suspended sediment is challenging. This research is a preliminary evaluation of the use of capacitance sensors to determine instream suspended sediment concentration (SSC). The overall goal of this research is to identify and develop a sensor for continuous suspended sediment monitoring in streams. Optimal sensor geometry will maximize the capacitance resolution of SSC from 0-1,000 mg/L. During the pilot study, in 1500 mL solutions at three different conductivities (0, 250, 500 $\mu\text{S}/\text{cm}$), 11 different sediment concentrations (0 – 1000 mg/L), and four temperatures (4, 10, 20, 30°C) will be prepared and analyzed with the design capacitance sensors. Conductivity and temperature will be monitored independently of the capacitance sensors. Total suspended solids will be analyzed for each SSC using a 100 mL aliquot removed from the area around the probes to determine the exact suspended sediment concentration within the probe measurement volume. Pilot study data will be used to create a calibration equation for the sensor and then tested with 30 randomly generated concentrations. Future research will provide additional laboratory and field testing of the prototype sensor.

INTRODUCTION

Throughout history humans have altered the natural environment, often causing unintended consequences to both society and the natural environment. An example of this conflict is the degradation of stream systems following watershed urbanization. A key component of stream function and ecological health is sediment concentration and transport. Due to the high temporal and spatial variability in suspended sediment transport, continuous measurement in multiple

spatial dimensions is necessary to fully understand stream dynamics and the impact of land use and channel modifications on the integrity of aquatic ecosystems.

Suspended sediment is a global problem; annual worldwide suspended sediment yields are estimated at around 20 billion tons (Wren et al., 2002) and the suspended sediment load makes up 70% of the total sediment transported to the oceans (Leopold, 1994). In North America, the annual costs sediment pollution approach \$16 billion (Osterkamp et al, 1998). The US Environmental Protection Agency (US EPA) identified sediment as a leading cause of water quality impairment in 2002 (US EPA, 2002). A survey of streams throughout the United States determined 46% of streams analyzed suffer from excessive siltation (Berkman and Rabeni, 1987). Excess sediment affects aquatic ecosystems by reducing the diversity and abundance of aquatic organisms, reducing reservoir capacity, increasing drinking water treatment costs, and serving as a carrier for contaminants such as phosphorus, bacteria, heavy metals and pesticides.

Section 303(d) of the Clean Water Act requires states to list impaired waters and establish Total Maximum Daily Load (TMDL) water quality management plans for watersheds drained by impaired streams. Results of these TMDL plans have wide-ranging impacts on local land use, industry, and development. Currently, many of the water quality assessments for sediment are based on a few grab samples; there is little information on natural levels of suspended sediments or the thresholds at which ecological impairment occurs. Few suspended sediment criteria have been established by the states or the US EPA because the connection between suspended sediment concentrations and ecological responses is not clear (Kuhnle and Simon, 2000).

Due to the large spatial and temporal variations inherent in sediment transport, the measurement of suspended sediment is challenging (Wren et al., 2002). The majority of sediment movement occurs infrequently during large rainfall events, leading to the rapid mobilization of field personnel into hazardous flood conditions. Also, a large number of samples may be required to see changes in suspended sediment concentration (SSC) throughout the duration of a storm (Lewis and Rasmussen, 1999).

Suspended sediment concentration can also be monitored with a variety of systems other than the traditional field methods of bottle and pump sampling. Current sensing technologies available on the market today or under evaluation are optical backscatter, acoustic, and laser diffraction. Optical backscatter sensors, commonly called turbidity sensors, have been commercially available for estimating SSC since the 1980's (Gartner et al., 2001). The SSC is determined from the strength of the backscatter from an infra-red or visible light source (Agrawal and Pottsmith, 2001). Turbidity probes have two main disadvantages: 1) particle size dependency; and, 2) biofouling of the optical window (Gartner and Gray, 2002).

Acoustic sensors determine SSC by monitoring the strength of the backscatter from short bursts of high frequency sound. The concentration and size of sediment present and the acoustic frequency are contributing factors to the backscatter voltage amplitude (Thorne et al., 1991). Acoustic systems are a non-intrusive technology and can measure over a large vertical range; however, in some systems changes in measurements may be due to increased concentration or changes in particle size (Gartner and Cheng, 2001; Gartner, 2002). Research is ongoing to distinguish particle size using different acoustic frequencies.

Laser diffraction measures multi-angle scattering of light particles at small angles, which is used to compute sediment size-distribution and concentration. Laser systems are not particle size dependent or affected by particle composition. However, laser diffraction systems are expensive, flow intrusive, provide only point measurements, and require special training for technicians.

Other technologies currently under development include: nuclear, capacitance, fiber optic transmissometer, and photo-optic imaging. This research focuses on applying capacitance principles to measure SSC. A capacitor is an energy storage device, and capacitance is a measure of the ability of a capacitor to store electrical charge. A basic capacitor design includes a pair of facing metallic plates separated by an insulating material (Carr, 1993). Each metal plate is a conductor (O'Malley, 1992). If the potential difference between the two conductors is V volts, where there is a positive charge of Q coulombs on one conductor and an equal and opposite charge on the other, the capacitance is determined as follows:

$$C = \frac{Q}{V}$$

where C is the capacitance in farads (Carr, 1993; O'Malley, 1992).

The main feature of the capacitor is its ability to hold charge on the insulating material, with negative charge on one of its two conductors and positive charge on the other. The capacitor then releases the stored energy. Capacitance is measured in farads (F), where one farad is the capacitance that will store one coulomb (Q) of electrical charge ($6.28 * 10^{18}$ electrons) at an electrical potential of one volt (Carr, 1993). A farad is too large for most electronics applications; therefore, micro- and pico farads are the more common units of measure.

The insulator between the two conductors is called the *dielectric*. The dielectric can be a gas or a solid, and consists of atoms with an electric dipole structure (Diefenderfer, 1998). The dipole structure requires a physical separation between positively and negatively charged entities on an atomic level. These charges are bound by atomic forces and are unable to move. In the case of ideal dielectrics there are no free charges and the material is neutrally charged on a molecular level (Diefenderfer, 1998). In the case of soils the dielectric is a mixture of soil, water, and air (Munoz-Carpena et al., 2005). For a parallel plate capacitor the capacitance is expressed as follows:

$$C = \epsilon_r \epsilon_0 \frac{A}{d} (n - 1)$$

where C is the capacitance in picofarads, A is the area of either plate in square meters, d is the distance between the plates in m, n is the number of plates, ϵ_r is the relative permittivity in farads per meter of the dielectric, and ϵ_0 is the electric constant or permittivity of free space (8.854 pFm^{-1} ; Nelson, 1991; Robinson et al., 2005). Therefore, system capacitance increases with increasing plate surface area or dielectric permittivity of the insulator and decreasing distance between the conductors (Carr, 1993; O'Malley, 1992).

The permittivity or dielectric constant is a property of the insulating material and is related to atomic characteristics of the insulating medium that support electric flux (Carr, 1993). The standard reference for dielectric constant is 1.000 for a perfect vacuum (Carr, 1993). When an electrical field is applied to the conductors, the charges on the conductors distort the surface charge on the atoms of the insulating medium, resulting in a net negative charge on one side of the dielectric surface and a net positive charge on the opposite side. This dipole-effect partially neutralizes the effects of the surface charge on the insulating medium, permitting an increase in electrical charge for the same voltage (O'Malley, 1992). The permittivity of a material has also been described as the extent to which the charge distribution within the materials is polarized in an external electric field (Robinson et al., 1999). Natural soils have an apparent dielectric constant of 2-20 depending on the air and water content within the soil matrix. Therefore, the soil minerals do not polarize as well as water, which has an apparent dielectric around 80 (Hasted, 1973).

Capacitance sensors have also been applied to environmental monitoring of natural systems. When applied in systems, such as a soil environment, the dielectric constant describes the soil permittivity to an electrical field (Munoz-Carpena et al. 2005; Robinson et al., 2005). Capacitance sensors measure the varying capacitance of the soil and relate that change to changes in the dielectric constant of the soil matrix. In a soil system, the value of the dielectric constant is dominated by the presence of water ($\epsilon = 80$), as compared to soil and air ($\epsilon_{\text{soil}} = 2-20$ and $\epsilon_{\text{air}} = 1$). Therefore, capacitance is often used to measure the water content of a soil matrix. The measured capacitance is used to calculate the apparent or relative dielectric constant of the medium based on the known surface area of the conducting plate and the distance between the plates. The dielectric constant increases with increasing proportions of water to soil.

Most capacitance sensors have two or more electrodes in parallel plates, rods or metal rings along a cylindrical configuration. When the probe is inserted into the soil and an electrical field is applied to the conducting plates, the soil in contact with the electrodes forms the dielectric of the capacitor and completes the oscillating circuit (Munoz-Carpena et al. 2005). Advantages of using a capacitance sensor over other soil moisture measurement techniques include the following:

- Can read at high salinity values;
- Can be connected to a data logger;
- Is flexible in application; and,
- Is relatively inexpensive.

Disadvantages of capacitance sensors include a small sensing sphere; the need for good contact with the medium; and, sensor sensitivity to temperature, bulk density, and clay content of the soil, as well as air gaps. Additionally, calibration to specific soil conditions is required for the highest accuracy (Munoz-Carpena et al. 2005).

Li et al. (2005) hypothesized that the relationship between soil and increasing water content could be reversed to measure increases of sediment content in a water sample. Their study tested both parallel and cylindrical capacitor geometries with brass conductors. Li et al. (2005) coated one of the two conductor plates with a thin layer of electric-insulating paint to reduce the

amount of energy lost due to the conductance of water. Li et al. (2005) found the outputs from both sensor geometries increased linearly with increasing sediment concentration up to 70% sediment, and then the outputs started to decrease linearly. The study also considered the effects of salinity, flow velocity, soil texture, and temperature on the accuracy of the capacitance sensor SSC predictions. The study found that temperature and salinity increased the output linearly, while flow velocity (0-2 m/s) and soil texture (sandy loam and loam) did not cause a significant difference in sensor output. Specifically, the error of the unified calibration curves was $\pm 0.3\%$ for non-saline soils ($\text{NaCl} < 0.2\%$) and $\pm 1.1\%$ for saline soils ($\text{NaCl} > 0.2\%$). Li et al. (2005) measured changes in SSC; however, more research on capacitance sensors for suspended sediment concentration is needed to better understand how environmental factors affect capacitance measurements.

This research will focus on the use of capacitance sensors to determine suspended sediment concentration. It is hypothesized that the dielectric constant of the suspension will decrease with increasing sediment concentrations. Sediment therefore decreases the ability of the insulating media to pass an electric current. Also, suspended sediment is present as both individual particles and aggregates, and a surface charge may be present on the sediment before the particle passes within the capacitor. The actual interactions between the complex sediment structure and an electrical field are not fully understood at this time; more is known about how a soil matrix and specific minerals react to an electric field. It is hypothesized that the electric field applied by the capacitor will not greatly change the configuration of the sediment aggregates or the ions adsorbed to the particles, but likely will rearrange the alignment of the particles relative to the plates.

To determine if capacitance principles can be successfully used to continuously record SSC, experiments will be conducted in the Land and Water Resources Laboratory of the Department of Biological Systems Engineering at Virginia Tech. A pilot study will be conducted initially to help design the final experiment. The pilot study will also help evaluate the amount of time needed for each stage of the final study and will evaluate possible interactions between SSC, conductance, and temperature.

GOALS AND OBJECTIVES

The overall goal of this research is to identify and develop a sensor for continuous suspended sediment concentration monitoring in streams. It is hypothesized that the dielectric constant of a sediment–water suspension measured by the capacitance probes will decrease with increasing sediment concentrations. It is also hypothesized that, at a given suspended sediment concentration, the dielectric constant will vary significantly with changes in electrical conductivity and soil texture. Specific objectives include the following:

1. Determine if capacitance sensors can accurately measure suspended sediment concentration;
2. Determine if particle size can be determined using capacitance probes; and,
3. Evaluate the feasibility of using capacitance sensors in the field for long term suspended sediment monitoring.

PROPOSED METHODS

Pilot study

A pilot study will be conducted using 1500-mL solutions at three different conductivities (0, 250, 500 $\mu\text{S}/\text{cm}$), 11 different sediment concentrations (0 – 1000 mg/L), and four temperatures (4, 10, 20, 30°C). The soil will be suspended using a stirring rod. Conductivity and temperature will be monitored independently of the capacitance sensors. Total suspended solids will be analyzed for each SSC using a 100-mL aliquot removed from the area between the plates to determine the exact suspended sediment concentration within the sensor measurement volume.

To reduce environmental errors while the capacitance sensor is being developed it will be tested under conditions typical of the stream environment. For example, conductance, temperature, stream flow, soil texture, and soil mineralogy will all be examined for significant effects on the capacitance of the sample volume. Specifically, the goal is to construct a predictive model for capacitance based suspended solids concentration, conductivity, and temperature. If a successful model can be developed, then inverse regression will allow the suspended solids concentration to be estimated given the capacitance, conductivity, and temperature. The model thus serves as a calibration curve. The hypothesis to be tested is that the capacitance can be predicted to a satisfactory level of accuracy (within ± 1 pF) given SSC, conductivity and temperature.

Because there is no theoretical model for the relationship between capacitance and SSC, conductivity, and temperature, standard model development and testing procedures will be used. Nonlinearities will be examined graphically and will include transformations, polynomial terms, and interaction terms. Tree regression and t-tests from standard linear regression models will be used to determine if the explanatory variables are useful. Scatter plot matrix and other graphical techniques will be used to initially evaluate variable relationships. Regression diagnostics will include residual plots, normality testing for correlations for residuals, leverage and Cook's distance.

The suspended sediment sensor will be calibrated with a static calibration procedure using known concentrations over the required sensor range. The model will be verified by randomly generating 30 concentrations, measuring the capacitance of each suspension, and predicting the SSC with the transfer function. The results will be compared to the known starting concentrations, and tested to see if the difference between the known concentration and the predicted concentration is significantly different than zero.

Extended study

The extended study will continue to enhance the transfer function by analyzing the effects of flow velocity and sediment mineralogy on the sensor predictions. Specifically, during this phase of the study, the effect of five different flow velocities on sensor output will be studied using a 1 m x 0.45 m x 6 m sediment flume.

Sediment size and mineralogy will be varied during the extended experiments to explore the ability of the capacitance sensor to distinguish changes in sediment size distribution. The probes

will be tested with a coarse quartz sand; the fine sand and silt fractions of the Groseclose silt loam used during the pilot study; and, two clay suspensions to determine differences in capacitance due to soil texture and mineralogy. The two clay minerals, kaolinite and montmorillonite, will be used to determine if the capacitance sensors can distinguish between particles within the same size fraction but with different chemical properties. Kaolinite is a 1:1 clay with a low surface charge and does not shrink or swell due to changes in moisture content. Montmorillonite is a 2:1 clay with a high surface charge that shrinks or swells significantly with changes in moisture content. During the soil texture evaluations, 11 sediment concentrations will be tested for each soil type (0-1000 mg/L). Also, the soil types will be tested in both the bench-scale set-up and the sediment flume, if possible. A particle size analysis will be completed for each sediment type.

The extended study will continue to build on the transfer function developed during the pilot study. Because there is no theoretical model for the relationship between capacitance and particle size, mineral type, and stream flow velocity, standard model development and testing procedures will be used, as described above.

Field study

Once the equations are created and tested sufficiently in the lab, the capacitance probes will be deployed in Stroubles Creek where turbidity probes are currently installed to compare measurements from the turbidity and capacitance sensors to sediment concentration and particle size as determined using depth-integrated sediment samples.

EXPECTED OUTCOMES

Currently water is treated as a disjointed resource, separating water quality from quantity, and surface water from ground water (Montgomery et al., 2007). The National Science Foundation has called for a national platform for real time environmental observations (Montgomery et al., 2007). This initiative will require knowledge of the physical, chemical, and biological processes controlling water quantity and quality. Traditionally, studies of this magnitude have been limited by a lack of observations at the necessary temporal frequency and spatial distribution to infer the controlling mechanisms (Montgomery et al., 2007). In freshwater, lotic ecosystems the magnitude and distribution of sediment transport is one of the most difficult characteristics to both measure and predict. In 1998 the US EPA described contaminated sediments as a high priority environmental issue and required by 2000 methods and indicators to assist in setting aquatic eco-criteria (US EPA, 1998). This study will determine the feasibility of using capacitance sensors for continuous SSC and particle size monitoring in field environments. If the study is successful, an inexpensive and simple alternative to acoustic and laser diffraction systems will be available for field studies world-wide. Use of these probes will allow researchers to continuously monitor natural variations in stream sediment loads and apply this knowledge to the development of TMDLs and the determination of sediment thresholds for biological impairment.

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COMPARISON OF METHODS FOR MEASURING STREAMBANK RETREAT

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KEY WORDS: LIDAR, surveying, streambank retreat, erosion, uncertainty

ABSTRACT

Nationwide, numerous public agencies are expending significant funds and effort to control streambank erosion as a means of reducing sediment and nutrient loads. However, data concerning streambank erosion and retreat rates are scarce and range greatly. In addition, the actual measurement of streambank retreat rates over time and space can be extremely time consuming and are often characterized by large measurement uncertainties. In this study, we compare and contrast four techniques for measuring streambank retreat rates: 1) typical surveying methods with an electronic total station; 2) bank erosion pins; 3) ground-based digital camera with three-dimensional photogrammetry; and 4) a portable, laser-based imaging and digitizing system (or LIght Detection and Ranging - LIDAR). We have conducted re-surveys of a retreating streambank on Stroubles Creek in Blacksburg, Virginia using all four techniques. Our results will contrast and compare the four methods in terms of time and effort, cost, spatial and temporal resolution, and uncertainty.

INTRODUCTION

Stream morphology, or the physical structure of streams, is directly related to many ecological and environmental issues. Approximately \$1 billion is spent annually on stream restoration projects (Bernhardt et al., 2005). Bank stabilization, bank retreat, and sediment loading are all issues impacting stream restoration and Total Maximum Daily Load (TMDL) development. Fluvial applications such as these rely on effective modeling of stream topography for scientists and engineers to make decisions. Traditionally, the development of stream topography is performed primarily through methods such as manual surveying with laser levels or electronic total stations. These methods, however, are limited by problems such as time-intensive field surveys, restricted resolution, and difficulties in surveying complex physical structures such as undercut banks. A higher resolution topographic model could lead to more efficient and accurate estimations of streambank retreat, as well as resulting sediment loads.

Streambank retreat is a process that is affected by multiple factors, including subaerial processes (climate-related events), fluvial erosion (direct transport of soil material by stream flow) and bank failure (caused by bank instability). This phenomenon is highly dependent on many variables and complex processes in the stream and the watershed, and so it is difficult to represent physically with a formal equation. Bank retreat rates are also difficult to measure with traditional methods, with measurements scarce and highly variable, ranging from 0 to 1100 m/yr (Simon et al., 2000). Measurement of bank retreat has already been investigated with methods

such as photogrammetry (Pyle et al., 1997) and digital elevation models (Chappell et al., 2003). Fluvial morphology has just recently been measured with ground-based LIDAR (Heritage and Hetherington, 2006), but few efforts have been made to quantify bank retreat over time.

METHODS

For this study, an 11 m bank on Stroubles Creek, located on the Heth Farm in Blacksburg, VA, has been selected to measure retreat over time (Figure 1). Stroubles Creek is a highly impaired stream with both urban and agricultural impacts. The study reach is located on a dairy farm, cattle have complete access to the stream, and the bank face is bare with minimal vegetation. The bank has been surveyed and measured with four different methods: 1) measurements from installed erosion pins; 2) surveying with an electronic total station; 3) 3-D photogrammetry with a digital camera and; 4) scanning with a ground-based, or terrestrial, LIDAR. Temporally the bank has been assessed four times, March 31, 2006, December 15, 2006, May 7, 2007, and August 13, 2007, with additional visits planned in the future. The median lateral streambank retreat and volumetric soil erosion will be calculated between each period of time using each of the surveying methods. The results will aid in comparing the four methods in terms of time and effort, cost, spatial and temporal resolution, and uncertainty.



Figure 1. Target bank at Stroubles Creek.

Erosion pins have been installed along a reach of Stroubles Creek every two meters, with 30 cm vertical spacing between pins. The pins are 6 mm diameter, 50 cm long stainless steel rods. Measurements are taken once a month of the pin length that is exposed out of the bank.

The total station was used to take five cross sections along the stream width. Points were surveyed focusing on representing the target streambank, with other points taken along the stream bed and the opposing streambank.

Setup for 3-D photogrammetry first required the placement of small plastic balls along the streambank to be used as control points. Approximately fifty images were then taken of the bank from multiple angles using a digital camera with a constant focal length. A 3-D point cloud was then generated with the software PhotoModeler from Eos Systems Inc.

The LIDAR system was set up in three locations on the opposing streambank to scan the target bank from different angles. An average point cloud resolution of 1 cm was used for each of the

scans. To aid with the spatial and temporal alignment of the LIDAR scans during post-processing, three square reference targets were fixed in space and marked with rebar to remain in the same location for each resurvey over time. Two targets were placed on top of the target bank at the extremes of the bank length, and one was placed on the opposing bank where the LIDAR scanner was set up. After the data was collected in the field, alignment and analysis were performed using PolyWorks from InnovMetric (Supplied by Direct Dimensions, Inc.).

RESULTS AND DISCUSSION

Topographic data of the target streambank has been collected using each of the four methods during each of the four visits to the site. The data from the later two visits is still in the analysis phase; however, there are some preliminary data from the first two visits. For the erosion pin method, the monthly values recorded were combined to create one value for each pin. This number represents the lateral bank retreat, or difference, measured at each pin location on the eroding bank (Table 1). For each cross section measured with the total station, the area difference was calculated (Figure 2). The median lateral retreats at each cross section are quite dissimilar between the erosion pin and total station survey methods (Table 2). With only this information, it is difficult to determine the actual bank retreat, since the possible range of values between the two results is so large.

Table 1. Retreat measured at each erosion pin (cm) from 03/31/06 to 12/15/06. Blank spaces represent the absence of erosion pins.

Height above base flow (cm)	Cross Section						
	1	2	3	4	5	6	7
120					4.0		
90	7.6	12.8	12.4	12.8	5.8	4.5	5.2
60	3.3	3.3	6.1	6.4	6.2	7.9	4.7
30	10.3	1.4	6.0	3.9	7.9	5.2	7.4
0	2.0	2.8	0.9	3.3	4.1	5.5	

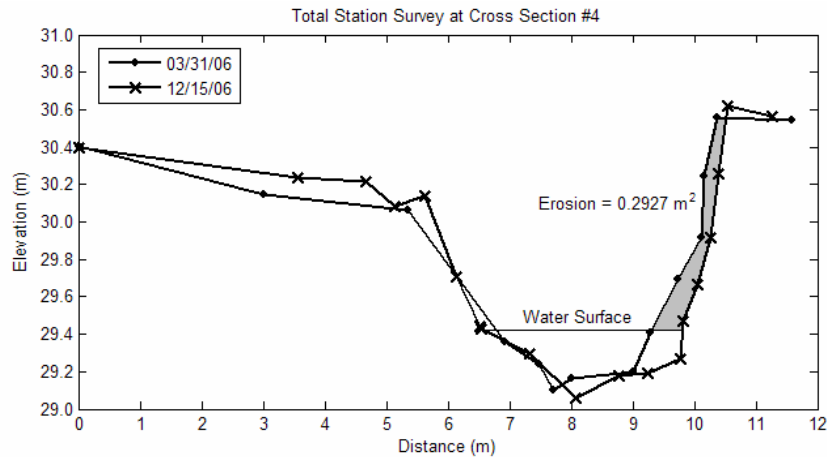


Figure 2. Example of measured cross section change using the total station.

Table 2. Median retreat values at each cross section (cm) from 03/31/06 to 12/15/06.

Method	Cross Section			
	2	4	5	7
Erosion Pins	3.1	5.1	5.8	5.2
Survey	52.1	22.5	16.4	15.9

The 3-D photogrammetry method was only used during the first two bank visits. The results from PhotoModeler (Eos Systems Inc.) are completely dependent on the position of the control points. Since control points were placed primarily on the erosion pins, there is no gain in resolution by using the digital camera. Also, aligning the control points between each of the digital images must be performed manually and is very time consuming. For these reasons, the method of using a digital camera for photogrammetry was abandoned for all future field visits.

Lateral retreat measurements using the LIDAR scans from March 31, 2006 and December 15, 2006 are shown in Figure 3. In PolyWorks (InnovMetric), the difference algorithm was set to calculate the shortest distance between the two datasets. This is only a partial section of the bank, ranging from approximately cross-section 3 to cross-section 6. The LIDAR data provides a better spatial picture for the erosion along the streambank at a much higher resolution than any of the other methods. The figure indicates that there is less than about 10 cm of retreat along most of the bank, with some areas of deposition. However, one important note is that for these first two field dates, targets were not used for spatial referencing of the LIDAR, so the point clouds required manual alignment. This likely introduced error to the LIDAR data, and was corrected during future field visits using stable targets.

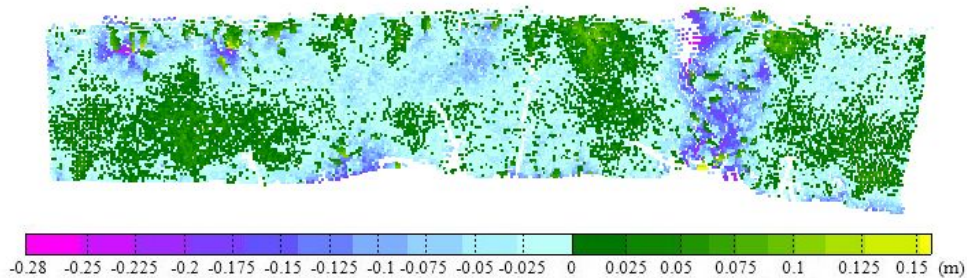


Figure 3. Lateral difference using LIDAR along the bank between 03/31/06 and 12/15/06.

For the last two visits, May 7, 2007 and August 13, 2007, reference targets were used to aid with alignment issues. The bank difference between these two times, shown in Figure 4, indicates a more uniform retreat across the bank. There are a few locations near the top of the bank with higher retreat values, indicating bank failure. The figure also shows that there are areas of soil deposition at the bottom of the bank below the bank failure. A comparison between the surveying methods for this time period will be performed in the future once the data for the other two methods has been processed.

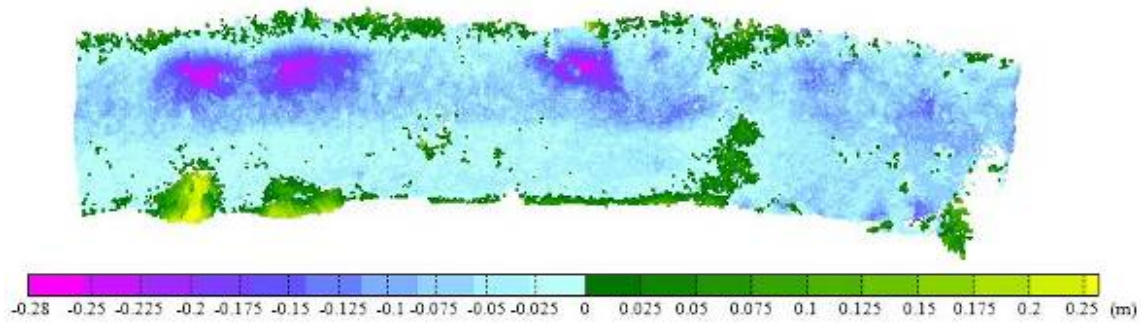


Figure 4. Lateral difference using LIDAR along the bank between 05/07/07 and 08/13/07.

It is interesting to look at some other spatial and temporal information summarized in Table 3. The ground-based LIDAR scanner was able to record far more points than either of the other tools. Erosion pins are limited to a set, coarse resolution and total station surveys are limited by the field time required to survey and by complex structures like undercut banks. The LIDAR scanner was able to produce a point cloud with an average resolution of 1 cm in less field time than required for surveying at a much lower resolution. However, the downside of the terrestrial LIDAR data is that currently, the post-processing of the data is not well defined and in a learning phase. While methods of processing and analyzing erosion pin and survey data is straight forward and has been utilized for decades, the LIDAR data requires more work involving data editing and scan alignment. As methods improve and automated programs are made to aid in the post-processing of LIDAR datasets, we expect that this disadvantage of LIDAR will improve.

Table 3. Approximate data sizes and time required for each method.

	Total Data Points	Field Time	Post-Processing
Erosion Pins	28	1 hr/month	1.5 hr
Survey	87	3 hr	1 hr
LIDAR	~ 1 Million	1.5 hr	7 hr

A volumetric difference between the different LIDAR datasets has not been performed yet because an additional module for PolyWorks (InnovMetric) is needed to create 3-D water-tight surfaces, as shown in the example in Figure 5. This add-on has been ordered and the analysis for comparing the volumetric soil erosion between the different surveying methods will proceed when the software is obtained. We expect the LIDAR data will produce dramatically different, and much more accurate, results for volume when compared to erosion pins and total station surveying. This is because the triangulated irregular network (TIN) model produced by the LIDAR will be much more refined than the other datasets as a result of the higher resolution. The simplified spatial datasets from the erosion pin and total station methods will lose much of the bank detail. As a result, we anticipate that the LIDAR will be able to create a TIN that is more representative of the actual streambank structure, as well as bank retreat measurements over time.

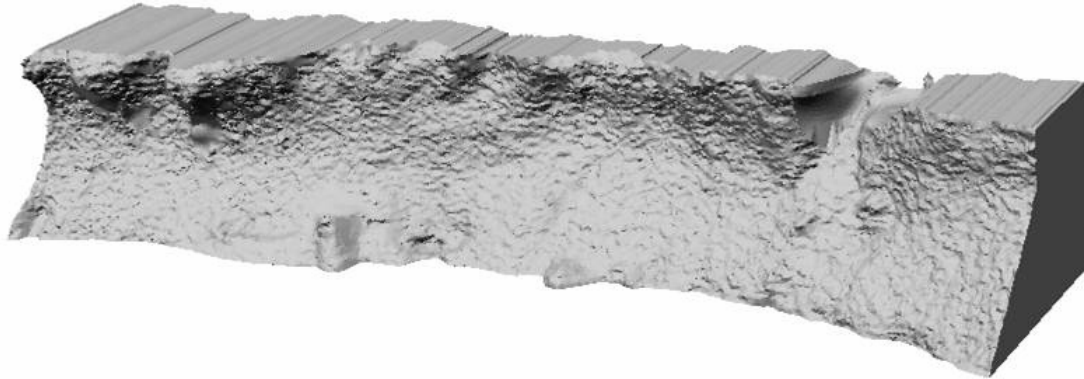


Figure 5. 3-D polygonal surface model of the bank created with PolyWorks (InnovMetric).

CONCLUSIONS

Preliminary results have shown that the LIDAR scanner is indeed an effective tool for measuring bank retreat. LIDAR is able to produce a much higher resolution spatial data set with less field time than traditional surveying. Currently, the post-processing stage for LIDAR data is more complex and time-consuming than for total station surveying. However, post-processing is expected to improve as the methods become more refined.

Continuing work involves processing the bank retreat data for each of the methods. Future visits to the streambank at Stroubles Creek are planned to gather more temporal data with both the total station and terrestrial LIDAR methods. The volumetric comparison of the data will also be performed in the near future once the proper software is obtained. Finally, the uncertainty in the datasets will be examined, including investigating using the LIDAR point cloud as an exact measurement for estimating the uncertainty in the total station survey.

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URBAN CHANNEL EROSION QUANTIFICATION IN UPLAND COASTAL ZONE STREAMS OF VIRGINIA, USA

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ABSTRACT

While considerable effort has been directed toward reducing erosion from agricultural and urban lands, a major source of sediment - streambank erosion - has received little attention. Streambank erosion is a complex cyclic process involving erosion of the bank toe followed by geotechnical bank failure. Because of the process complexity and the lack of physically-based algorithms to describe these processes, quantification of this source is often underestimated. The proposed research will identify the most widely applicable approach for estimating sediment loading from channel degradation and the most appropriate field-based monitoring techniques for future model calibration and verification.

The coastal zone management area (CZMA) in Virginia consists of three general categories of land use – highly urbanized areas (urban), an urbanizing fringe (developing), and a mix of relatively undeveloped forest and agricultural areas (rural). The relative dominance of channel erosion processes and the state of channel evolution varies among these three categories of land use. Because of the increasing complexity and source interaction in large watersheds, the relatively higher contributions of channel erosion from the urbanizing fringe, and the extension of this fringe into headwater areas, the assessment of field-based research will focus on watersheds less than 10-100 km², which also corresponds to the lower range of watersheds in the new NWBD watershed delineation scheme, making results from this research potentially useful to other statewide programs.



Session VII-A

Case Studies of Water Conflicts and Collaborations

- Multiple Partners Provide the Key to Watershed Restoration Success in West Virginia
- Wastewater Treatment in the Upper Guyandotte River Watershed: A Creative Strategy to Meet Community Needs
- Regional Approaches to Complying with Virginia's Water Supply Management Regulations

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MULTIPLE PARTNERS PROVIDE THE KEY TO WATERSHED RESTORATION SUCCESS IN WEST VIRGINIA

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ABSTRACT

Historically, flooding has affected each of the 32 major watersheds and 55 counties within the state of West Virginia. Federally declared flood disasters are far too common in the Mountain State. Thousands of West Virginians have suffered through damaged homes and businesses, declining communities, and deteriorating water quality.

The West Virginia Conservation Agency has developed an integrated watershed management program to address the need for restoration of streams that have been impaired from mining, residential and commercial development, and floodplain encroachment. This program is a cooperative effort by multiple agencies and stakeholders with the intent to improve the health and sustainability of watersheds throughout West Virginia. It addresses problems within a watershed that do not qualify for any of the traditional federal programs.

Projects are identified by watershed groups, individual citizens, and local government. Conservation Districts also play a key role in prioritizing identified projects within their area and administering the project funding. This program is funded with state dollars that are used to leverage additional local funding (i.e. watershed groups and city government) and other state funding. Partnerships with local, state and federal government as well as non-profit groups and landowners have allowed WVCA through the Conservation Districts to implement more projects. Permits are obtained through the appropriate agencies, utilizing Army Corps of Engineers Regional Permits where applicable. Projects completed in coal mining areas of southern West Virginia often utilize mitigation funds for implementation. This program is also

being used in conjunction with the Conservation Reserve Enhancement Program (CREP) on agricultural land. Inspection and monitoring of these projects have shown great benefits to the environment and surrounding communities. These benefits include stable streambanks, reduced sediment load, improved sediment transport, improved water quality, enhanced flood plain area, and increased fish habitat.

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**WASTEWATER TREATMENT IN THE UPPER GUYANDOTTE RIVER
WATERSHED: A CREATIVE STRATEGY TO MEET COMMUNITY NEEDS**

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ABSTRACT

66% of the 3,200 households in the Upper Guyandotte watershed in southern West Virginia are without access to adequate household wastewater treatment; many rely on “straight pipes” that discharge raw sewage directly into the river. Consequently, local streams are severely contaminated by fecal coliform bacteria. This presentation will describe the grassroots efforts of the Upper Guyandotte Watershed Association to develop a sustainable solution to this social and environmental problem. Through the development of a comprehensive wastewater treatment plan, UGWA has advocated for the use of decentralized wastewater treatment systems in order to provide cost-effective service to over 60 rural, low-income watershed communities. Implementation of the top priority project identified in the plan—a decentralized system to serve 50 homes in one southern Raleigh County community—is currently under way.

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REGIONAL APPROACHES TO COMPLYING WITH VIRGINIA'S WATER SUPPLY MANAGEMENT REGULATIONS

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ABSTRACT

In response to the extended drought of 2001 through 2003, the State Water Control Board developed Regulation 9 VAC 25-780, Local and Regional Water Supply Planning. The purpose of this regulation is to establish a comprehensive water supply planning process for the development of local, regional, and state water supply plans. The plans will evaluate existing water resources, propose management strategies to protect the quantity and quality of those resources, evaluate the adequacy of those resources to meet current and future demand, and identify potential water supply alternatives.

A regional approach allows local governments to discuss individual and regional water supply needs and work together to find solutions to the challenges they encounter during the planning process. One of those challenges is evaluating future demand for the next 30 to 50 years. Demand projections have traditionally been very conservative often resulting in unrealistic projections. One of the goals of this planning process has been to develop a more realistic demand projection methodology based on water use in disaggregate amounts (i.e., residential, commercial, industrial) and compare the resulting projections to the more traditional population-based and land use methods. There have been many challenges which include obtaining accurate current demand data from utility billing systems, being able to clearly define disaggregated categories, addressing local government concerns with a more realistic method, and addressing interbasin transfer outside the region and in some cases interstate transfer. This presentation will discuss the demand projection methodology and address the challenges that have been encountered.



Session VII-B

Research Experiences of Undergraduates (Part II)

- Ecological Stoichiometry of *Pycnopsyche gentilis* and Its Resources
- A Study of Energy Consumption by Water Supplies and Wastewater Infrastructure in Blacksburg, Virginia
- Analysis of Water Consumption for Developing Various Energy Sources

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ECOLOGICAL STOICHIOMETRY OF *PYCNOPSYCHE GENTILIS* AND ITS RESOURCES

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ABSTRACT

Ecological stoichiometry is a branch of ecological study which traces elements along the various components of an ecosystem. If applying this concept on an organismal level, a flow of nutrients can be traced between an organism and its resource. Oftentimes, an organism eats matter that is not an exact fit to the chemical composition of its body (i.e. the nutritional requirements of that organism). This study was formed to explore how that consumer-resource imbalance affects the physiological processes of that organism such as excretion and assimilation. While data indicated distinct relationships between an organism and nutrient content of its food, sample sizes were simply not adequate for providing statistically reliable answers. However, this study provokes many new questions and provides a foundation for further studies of this fairly unexplored area.

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A STUDY OF ENERGY CONSUMPTION BY WATER SUPPLIES AND WASTEWATER INFRASTRUCTURE IN BLACKSBURG, VIRGINIA

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ABSTRACT

Understanding the energy and water relationship is a fundamental step toward developing technology and strategy that will help sustain the supply of both resources. Rising energy costs and increased water demand due to population growth will affect availability of safe and adequate water to meet societal needs. Information on energy consumption through water infrastructure is scarce. The objective of this study was to estimate energy consumption associated with treating drinking water, wastewater treatment, and water distribution using Blacksburg, Virginia as a case study site. Data was collected from water and wastewater treatment plants, Town of Blacksburg, and Virginia Tech. Energy consumption (kWh) for various systems components were estimated for the years 2000, 2003, and 2006. Results show fairly consistent seasonal and annual energy consumption for water and wastewater treatment plants with some noticeable peaks and valleys. Per volume basis, slightly more energy is needed for wastewater treatment compared to water treatment for drinking water supplies. For drinking water supplies, more than 60% of energy use is attributed to pumping and distribution. Paper is concluded with future research needs.

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ANALYSIS OF WATER CONSUMPTION FOR DEVELOPING VARIOUS ENERGY SOURCES

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ABSTRACT

Water serves as a basic ingredient for the extraction of energy resources and for production of useful energy forms and therefore is a major consumer of freshwater resources. For example, coal requires water for scrubbing; electricity generation from coal requires water to produce steam; nuclear power plants require water for cooling; hydroelectricity uses water stored in reservoirs; and oil and gas production may rely on injection of water into wells. Some alternative energy sources such as ethanol also require large volumes of water. The Energy Information Administration has projected that the American electricity demand would jump by approximately fifty percent in the next 25 years. Significant amounts of water will be needed to meet energy demand. Decreased water availability during drought conditions, overdrawn groundwater aquifers, and competition to meet other water demands such as ecosystem services and irrigation will impact energy production and cost. The objective of this paper is to determine the water consumption efficiency of various energy production systems. Several existing documents on energy production systems and water use were reviewed. Water use efficiency was calculated by determining water volume (gallons) used to produce one unit of energy. The energy is expressed in British thermal unit (BTU), a global energy unit for power generation. Using BTU in analysis facilitates comparison between various types of energy production systems. Results show that nuclear power plants are the least water intensive power generation systems with a range of water consumption of 2.62×10^{-6} to 4.58×10^{-6} gallons per BTU. Fossil fueled thermoelectric plants are the most water intensive power generation systems with a range of 1.15×10^{-3} gallons per BTU to 2.24×10^{-3} gallons per BTU. In respect to fuel sources, liquid natural gas is the most water efficient, using only 1.72×10^{-7} gallons per BTU. Biodiesel is the least efficient since it requires 1.4×10^{-2} gallons per BTU. The presentation will discuss water consumption for various types of energy production systems and its implications for meeting future water and energy demand.



Session VII-C

Stream Sediments and Acid Mine Drainage: Connecting Terrestrial and Aquatic Systems

- Near Bank Shear Stress Along Vegetated Streambanks
- Changes to In-Stream Turbidity Following Construction of a Forest Road in a Forested Watershed in West Virginia
- An Economic Benefits Analysis for Acid Mine Drainage Remediation in the West Branch Susquehanna River Watershed, Pennsylvania
- Externality Effects of Acid Mine Drainage on Residential Property Values in Clearfield County, PA

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NEAR BANK SHEAR STRESS ALONG VEGETATED STREAMBANKS

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KEY WORDS: boundary shear stress, streambank retreat, fluvial erosion

ABSTRACT

Sediment, a leading cause of water quality impairment, damages aquatic ecosystems and interferes with recreational uses and water treatment processes. Streambank retreat can contribute as much as 85% of watershed sediment yield. Vegetation is an important component of stream restoration designs, but vegetation effects on streambank boundary shear stress (SBSS) need to be quantified.

The overall goal of this experiment is to predict SBSS along vegetated streambanks. This goal will be met by determining a method for measuring SBSS in the field along hydraulically rough streambanks; evaluating the effects of streambank vegetation on SBSS in the field; and developing predictive methods based on measurable vegetation properties. First, a prototype stream will be modeled in a flume study to examine both SBSS measurement theory and instruments for field use. An appropriate method (law of the wall, Reynold's stresses, TKE, or average wall shear stress) and field instrument (ADV, propeller, or Pitot tube) will be selected, resulting in a field technique to measure SBSS. Predictive methods for estimating streambank boundary shear stress based on common vegetation measures will be developed in the flume study and validated with field data.

This research is intended to evaluate the effects of vegetation on fluvial entrainment of streambank soils, providing insight to the type and density of vegetation that offers protection against streambank erosion. The results will also aid in quantifying sediment inputs from streambanks, providing quantitative information for stream restoration projects and watershed management planning.

INTRODUCTION

Sediment is a leading cause of water quality impairment for rivers, polluting 31% of impaired stream miles (USEPA, 2002). Sediment damages aquatic ecosystems and interferes with recreational uses and water treatment processes. Damages associated with sediment pollution in North America cost roughly \$16 billion annually (ARS, 2003). While agriculture, urban runoff, construction, and silviculture are significant sources of sediment (USEPA, 2002), streambank erosion can contribute as much as 85% of the total watershed sediment yield (Wallbrink et al., 1998; Trimble, 1997; Prosser et al., 2000). In response to the habitat and water quality impacts of streambank erosion, approximately \$1 billion has been spent annually over the past decade to restore degraded streams (Bernhardt et al., 2005). Stream restoration typically involves regrading the stream channel and establishing woody riparian vegetation on the bank face and floodplain. While design information exists for traditional bank stabilization materials, such as riprap, information on the effects of woody vegetation on boundary shear stress and bank erosion is incomplete. Determination of streambank boundary shear stress (SBSS) is critical for correctly predicting streambank erosion and stable channel design. This study will examine the changes in boundary shear stress resulting from three forms of riparian vegetation.

While bank retreat occurs from a combination of subaerial processes, fluvial erosion, and mass failure (Thorne, 1982), this research will focus on fluvial entrainment. Fluvial entrainment initiates bank retreat by scouring soil from the bank toe. Stream flow generates shear stress along the streambank proportional to the near bank velocity gradient. When the bank material does not exert an equal and opposite force (lift and drag forces overcome the forces holding the grain or aggregate in place), fluvial entrainment of bank material occurs (Thorne, 1982; Lawler et al., 1997). The excess shear stress equation, which relates the streambank erosion rate to the difference between the applied grain shear stress and the soil critical shear stress, predicts the erosion rate of fine grain soils due to stream channel scour (Osman and Thorne, 1988; Hanson and Simon, 2001). Hydraulic shear stress on the channel boundary is commonly calculated based on flow depth and slope (Chang, 2002), but this method can cause major errors in soil detachment calculations when resistance (e.g. vegetation) is substantial (Thompson et al., 2004). A key component of erosion models is the determination of the applied bank shear stress. While models exist to determine SBSS (e.g. law of the wall, Reynold's stresses, turbulent kinetic energy, FST-hemispheres, Prandtl's mixing length theory, and average boundary shear stress), those models do not consider the impact of vegetation; therefore, research is needed to quantify the influence of riparian vegetation on SBSS so the effects of stream restoration can be assessed at the watershed scale.

The overall goal of this experiment is to develop a method to predict SBSS for vegetated streambanks to improve stream restoration design and to assess the impacts of riparian management and landuse change on channel stability. This goal will be met by:

1. Evaluating the effects of streambank vegetation on SBSS;
2. Determining a method for measuring SBSS in the field along hydraulically rough streambanks; and
3. Developing methods for SBSS based on easily measured vegetation properties.

METHODS

Objective 1. Evaluate the effects of streambank vegetation on SBSS

A second order prototype stream, Tom's Creek (watershed area = 2070 ha) in Blacksburg, VA, will be modeled within a research flume (1-m wide; 6-m long; 40-cm deep). The stream has three distinct vegetation types: forested, mixed, and herbaceous. An experimental reach was identified within each vegetation type, resulting in three prototype reaches. The reaches met the following criteria to minimize variability: (1) relatively straight to decrease secondary flow effects; (2) grouped to ensure similar watershed characteristics (Hession et al., 2003); (3) minimal tributary input for constant discharge among reaches; and, (4) at least four bankfull widths in length. Each reach consisted of a riffle and pool so the bedforms were similar among the reaches. The main difference among the research reaches was vegetation density and type. The forest, mixed, and herbaceous reaches were dominated by rhododendron, briars, and tall fescue grass, respectively (fig. 1).



Figure 1. Experimental stream reaches along Tom's Creek in Blacksburg, VA, with forested (a), mixed (b), and herbaceous (c) vegetation cover.

For each experimental field reach (forested, mixed, and herbaceous), topography, vegetation, discharge, and bed and bank sediment characteristics were measured. Scaling followed a fixed-bed Froude scale modeling (FSM) technique to achieve kinematic and dynamic similarity between the prototype and model streambank (Peakall et al., 1996). Median particle size for the model bed and bank were calculated as 10 mm and 1 mm, respectively (Wolman, 1954). The model streambank (49-cm long) will be 35.5° from horizontal, which represents an average bank angle among the reaches (fig. 2). One model streambank of the prototype stream will be constructed for each experimental field reach, resulting in three model streambanks. Only half of the channel width will be constructed to minimize sidewall effects. Simulated vegetation (wooden dowels, plastic strips, etc.) will be attached in the locations identified in the field survey.

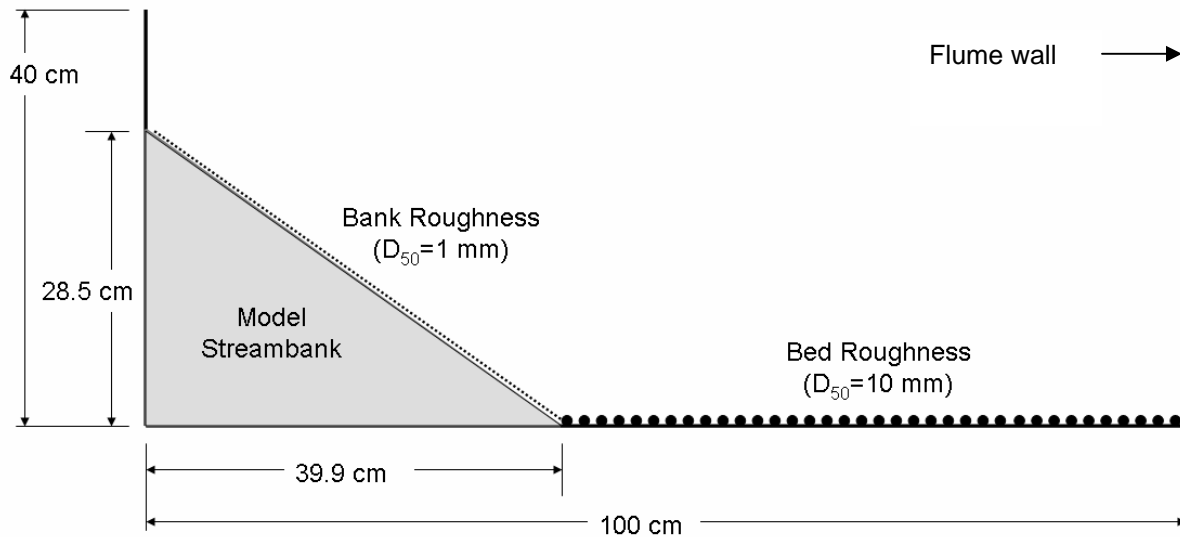


Figure 2. Dimensions for constructed flume streambank without vegetation; bank dimensions, bed roughness, bank roughness, and bed slope will remain consistent among all vegetation types.

Several methods of calculating near bank shear stress will be evaluated, including the law of the wall, Reynold's stresses, and TKE method (table 1). Because it is typically used in channel design, the average boundary shear stress will also be calculated. The law of the wall will be evaluated following the three procedures outlined in Wilcock (1996), requiring time-averaged velocity measurements. Reynold's stresses will be evaluated by point and extrapolation methods (Biron et al., 2004) and require velocity fluctuation values in the streamwise and vertical directions. The TKE method will be applied following the procedures of Biron et al. (2004) and Daniels and Rhoads (2004), requiring velocity fluctuations in multiple directions.

To determine the boundary shear stress using the methods discussed above, a velocity profile and 3D velocity fluctuations near the boundary are necessary (table 1). A Sontek Flowtracker ADV will be utilized to provide 3D velocity profiles (20 hz for 30 s) and fluctuations for the Reynold's stresses and TKE methods. A miniature propeller (Armfield H33) will be used to measure time-averaged velocities in the streamwise and vertical directions. A Pitot tube will be used to take velocity measurements for the law of the wall, as it only provides measurements in the streamwise direction. Velocity profiles (at 0.5-cm intervals; Biron et al., 1998) perpendicular to the bank and point velocity measurements at a height of 10% of the flow depth will be measured with each instrument.

Table 1. Velocity instruments, procedure, and measurements used to test each near bank shear stress theory for the flume studies.

Theory	Procedure	Measurements	Velocity Instrument
Law of the Wall	(Wilcock, 1996)	1. Velocity profile at 0.5-cm intervals perpendicular to the bank 2. Point velocity at 0.1 depth from the bank	ADV* Miniature Propeller Pitot tube
Reynolds Stresses	(Biron et al., 2004)	1. Velocity profile fluctuations at 0.5-cm intervals perpendicular to the bank 2. Point velocity fluctuations at 0.1 depth from the bank	ADV Miniature Propeller
TKE**	(Biron et al., 2004; Daniels and Rhoads, 2004)	1. 3D velocity profile and fluctuations at 0.5-cm intervals perpendicular to the bank 2. Point velocity and fluctuations at 0.1 depth from the bank	ADV
Average Wall Shear Stress		Hydraulic radius and slope	NA

* Acoustic Doppler velocimeter

** Turbulent kinetic energy

A Dantec MiniCTA system with a flush mounted hot-film probe (55R46) will be used to measure shear stress at the streambank wall. For this study, measurements made by the hot-film anemometer will be considered representative of the actual boundary shear stress. Results from the 13 combinations of experimental theories and instruments (table 1) will be compared to the hot-film anemometer measurements.

A measurement grid will be established in the flume test section (1.5-m long, fig. 3). Measurements will be taken along four cross sections on the upper, middle, lower, and toe of the bank, and on the bed (20 nodes total). Measurements will be conducted for each vegetation type and location at a discharge corresponding to the prototype stream bankfull flow. To determine the most accurate combination of estimation method and instrumentation, a randomized complete block approach will be used to test for differences among local shear stress estimates. Each vegetation type will be examined separately. The node location will be treated as the block (b=20; fig. 3) and the theory/instrument combination will be evaluated as the treatment. Velocity profiles perpendicular to the flume model streambank will also be evaluated. Velocity versus perpendicular distance from the streambank will be graphed to qualitatively evaluate velocity profiles. Average profiles for each point on the cross section (upper, middle, lower, toe, and bed) for each vegetation type will be plotted. The profiles will be compared among vegetation types to see if profiles are similar along the bank face. The analysis will evaluate whether the velocity profiles follow a logarithmic, S-shaped, wedge (Hoover and Ackerman, 2004), or other shape. The profiles will be compared along the bank for each separate vegetation type to determine if a single shape characterizes the profile for the entire bank. To determine whether there are differences in SBSS as a function of vegetation type, the hot-film anemometer measurements will be examined using a randomized design approach.

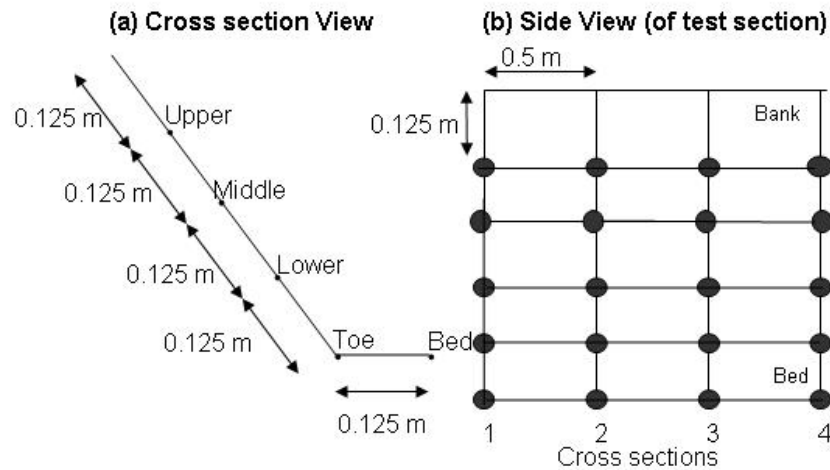


Figure 3. Schematic of grid where SBSS measurements will be taken in the flume study along the cross sections (a) and streambank (b); the side view shows the placement of the four cross sections in the test section of the flume.

Objective 2: Determine a method for measuring SBSS along hydraulically rough streambanks

From the flume experiments both a model and a field instrument will be selected as the appropriate method for use in the field. The field method will be selected based on its similarity with the hot-film anemometer estimates and its ease of use in the field. It will be a combination of a near boundary shear stress theory (law of the wall, Reynold's stresses, or TKE) and a velocity instrument (ADV, miniature propeller, or Pitot tube) and will be used to measure SBSS along the prototype stream (Tom's Creek) during flood flows. Measurements will be taken at each experimental reach (herbaceous, mixed, and forest). The grid intervals will be determined from the survey data. Velocity profiles (0.05 m from the bank at 0.10 m intervals perpendicular to the bank; Pizzuto and Meckelnburg, 1989) as well as point measurements (0.1 of flow depth perpendicular to the bank) will be measured. Even if the selected field method requires point measurements, velocity profiles will be measured so the profiles can be analyzed in the same manner as in the flume study.

Objective 3: Develop predictive methods for estimating SBSS based on easily measured vegetation properties

Finally, a predictive model for boundary shear stress will be developed from the shear stress measurements and vegetation properties (e.g. roughness density, deflected height, relative submergence, blockage factor, frontal density, and stem density; table 2), using regression analysis. The predictive model will be developed using the flume experiments and tested in the field. The analysis will be extended through the use of dimensionless numbers and a shear stress partitioning approach. The grain shear to average shear stress ratio recommended for each vegetation type will be calculated.

Table 2. Vegetation measurements for the development of a predictive model.

Vegetation properties	Reference
Roughness density (m^2/m^2)	Raupach, 1992; Nepf 1999
Frontal density	Garcia et al., 2004; Bennett et al., 2002, 2004
Deflected height (m)	Samani and Kouwen, 2002;
Relative submergence (m/m)	Jarvela, 2002
Blockage factor (%)	Green, 2005

EXPECTED OUTCOMES

This research is intended to determine the role of riparian vegetation in stream morphology, providing insight to the type and density of vegetation that offers the greatest protection against streambank erosion. Streambank boundary shear stress values for three types of riparian vegetation will be calculated for use in watershed modeling and stream restoration design, contributing quantitative information for stream restoration projects. A method for measuring SBSS in the field will be developed in a flume study and will quantify SBSS for three distinct vegetation forms. A predictive equation to determine SBSS based on easily measurable vegetation properties will be developed, providing insight into the influence of vegetation on SBSS and the role of vegetation in stream restoration. The results will also will aide in quantifying sediment load from streambanks, providing water quality information for use in watershed management planning.

Specific likely outcomes of the research include the following:

- 1) Development of a field method to measure streambank boundary shear stress for three distinct vegetation types: herbaceous, mixed, and forested;
- 2) Quantification of SBSS for field situations for three distinct vegetation types;
- 3) Evaluation of velocity profiles perpendicular to the bank along hydraulically rough streambanks;
- 4) Insight into observed differences in stream width as a function of riparian vegetation type;
- 5) Determination of significant measurable vegetation properties to describe the influence of vegetation on SBSS; and,
- 6) SBSS prediction method based on easily measurable vegetation properties.

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CHANGES TO IN-STREAM TURBIDITY FOLLOWING CONSTRUCTION OF A FOREST ROAD IN A FORESTED WATERSHED IN WEST VIRGINIA

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KEY WORDS: best management practices, forest roads, erosion and sediment, stream quality.

ABSTRACT

Two watersheds were sampled to assess changes to in-stream suspended sediment and turbidity following the construction of a forest haul road. Water control features were installed on the haul road after a delayed period of time to assess runoff from the road. The catchments are ephemeral/intermittent tributaries of the Left Fork of Clover Run in the Cheat River watershed. Turbidity, suspended sediment concentrations (SSC), and streamflow were measured throughout May 2005. Silt fence was installed on these catchments to prevent natural soil movement from entering the stream channel. The objectives were to assess turbidity before and after road improvement and assess in-stream turbidity levels. Reductions in the levels of turbidity and suspended sediment were found after the road conditions were improved. The primary focus of this study was to measure changes to in-stream suspended sediment. One-hundred and thirty-four storms were sampled during post-treatment. By comparison, the reference watershed routine and storm samples average SSC were 0.7 and 0.5 times the pretreatment levels post-treatment while average turbidity were 1.0 and 1.2 times the pretreatment levels post-treatment, respectively. The highest turbidity value recorded in the treatment watershed (2,352 Nephelometric turbidity units (NTU)) was 6.4 times larger than the highest turbidity sampled in the reference watershed, it was sampled during low streamflow ($<1.4 \text{ L s}^{-1}$ or $<0.05 \text{ ft}^3 \text{ s}^{-1}$ (CFS)). Fourteen post-treatment samples exceeded 100 NTU at discharges greater than 56.5 L s^{-1} (2.0 CFS) when the treatment watersheds average streamflow was 5.5 L s^{-1} (0.20 CFS). Significant increases in turbidity can be explained by the construction of FS 973, inadequate cross drainages, and a poorly constructed stream crossing.

INTRODUCTION

Road construction and use are recognized as the primary sources of sediment production during forest operations (Hornbeck and Reinhart 1964). Roads accelerate erosion, affect run-off, and increase effective channel lengths in headwater watersheds (Reinhart 1964, Binkly and Brown 1993, Jones and Grant 1996, Wemple et al. 1996). One year after road construction in north

central West Virginia, treatment watershed maximum turbidity exceeded maximum reference watershed turbidity by 3,700 JTU (Jackson turbidity units) (Hornbeck and Reinhart 1964).

The largest increases to in-stream suspended sediment and turbidity occur during road construction and maintenance (Hornbeck and Reinhart 1964, Swift 1988). Turbidity and sediment tend to decrease most rapidly within the first couple of years post-treatment (Rice and Wallis 1962, Hornbeck and Reinhart 1964, Megahan and Kidd 1972). After the first couple of years recovery rates decrease and elevated turbidity and sediment continue to persist for years (Hornbeck and Reinhart 1964).

The adverse effects caused by increasing in-stream sediment have initiated the use of better construction practices. For example, best management practices (BMPs) are mandated by the Clean Water Act 1977 and state law during forest operations. Water quality degradation following forest operations decreases with the use of better BMP construction practices (Kochenderfer and Hornbeck 1999). Although, these methods do decrease to-stream sediment transport, inadequate background sampling can mischaracterize BMP effectiveness (Edwards et al. 2004). Storm sampling is required to characterize sediment and turbidity in steep headwater stream channels, as variation between storm exports can be as large, or larger than variation between annually exported sediment (Kochenderfer et al. 1997).

Therefore, the objectives of this study were to: 1) describe turbidity before and after haul road construction, 2) determine if or when in-stream turbidity levels decreased after construction of a haul road in the treatment watershed, and 3) if possible, given the short pre and post treatment periods, evaluate if recovery was linear, exponential, or if turbidity levels off at a level higher than pretreatment at some point in time.

METHODS

In-stream suspended sediment, turbidity, and streamflow in two headwater streams were measured since 1999. Both streams are located within the Clover Run Watershed, Monongahela National Forest, north central West Virginia. This design adopts the typical paired watershed design (e.g. reference and treatment watersheds) to evaluate the effects of road construction on water quality (i.e. turbidity and suspended sediment).

When measured from the stream monitoring stations, the treatment and reference watersheds are 32.7 ha (80.8 ac) and 20.2 ha (49.9 ac) in area respectively. Hillside slopes average 42.6 percent and 38.7 percent in the treatment and reference watersheds respectively. Stormflow sampling started November 2, 1999 and lasted until June 4, 2002 in both watersheds. One-hundred and fifty-three storms were sampled during pretreatment. Of these 70 were paired storms – that is, they were sampled on both the treatment and reference watershed. Stormflow sampling in the reference watershed started again on November 1, 2002 and lasted until April 30, 2005. Treatment watershed storm sampling started again on October 15, 2002 and lasted until April 30, 2005. One-hundred and thirty-four storms were sampled during post-treatment. Of these forty-two were paired storms.

Stream velocity and stage measurements were made on 5-minute intervals since October 1, 1999. Water samples were processed for turbidity at the US Forest Service's Timber and Watershed Laboratory in Parsons, West Virginia. Turbidity, in nephelometric turbidity units (NTU), was determined using a Hach Ratio Turbidimeter, which was calibrated using formazin standards (Edwards, P.J. Submitted).

Statistical Analysis Systems (SAS 1988) was used to analyze these data. Nonparametric methods primarily were used because the data were not normally distributed. Median scores were used to test for differences between watersheds turbidity. The relationship between turbidity and SSC (TS ratio) was created to compare the turbidity of a sample to the suspended sediment concentration. This ratio compares two different types of water clarity measurements and samples between watersheds were of different volumes.

RESULTS

Pretreatment reference watershed routine samples averaged 4.0 NTU, with a standard deviation of 6.5 NTU (Table 1). Pretreatment treatment watershed routine samples averaged 1.7 NTU, with a standard deviation of 5.2 NTU. Pretreatment reference watershed stormflow samples averaged 9.0 NTU, with a 21.1 NTU standard deviation. Forty samples exceeded 25 NTU. Reference watersheds stormflow turbidities averaged 5.8 NTU, with a standard deviation of 8.1 NTU during post-treatment. The treatment watershed's turbidities averaged 34.2 NTU, with a standard deviation of 109.7 NTU during post-treatment (Table 1). The treatment watershed's average stormflow turbidity (3.5 NTU) was not statistically different ($P=0.16$) than the reference watersheds average stormflow turbidity (4.4 NTU) for paired storms. The treatment watershed's median turbidity (2.1 NTU) for paired storms was statistically lower ($P=0.02$) than the reference watershed's median turbidity (3.8 NTU). Post-treatment stormflow turbidities exceeded 100 NTU on the treatment watershed 100 times and exceeded 100 NTU on the reference watershed one time. The treatment watershed's average turbidity (33.2) was 5.4 times the reference watersheds average turbidity (6.2 NTU) ($P<0.01$). The largest recorded turbidity (2352 NTU) occurred at the beginning of a July 2003 storm.

Table 1. Nonparametric statistical tests between and within watersheds and similar sample types.

	Samples	Watershed	n	Average (NTU)	StDev (NTU)	Median (NTU)	Ranks (NTU)
Pretreatment	All	Treatment	2680	2.8(a,1)	11.3	1.2(a,1)	2366(a,1)
		Reference	3059	4.3 (b,1)	6.6	2.5 (b,1)	3311(b,1)
	Routine	Treatment	1019	1.7 (a,1)	5.2	1.0 (a,1)	870 (a,1)
		Reference	1120	4.0 (b,1)	6.5	1.8 (b,1)	1251(b,1)
	Storms	Treatment	1661	3.5 (a,1)	13.8	1.4 (a,1)	1500(a,1)
		Reference	1939	4.5 (b,1)	6.6	2.9 (b,1)	2057(b,1)
Post-treatment	All	Treatment	2037	28.4 (a, 2)	97.2	7.7 (a, 2)	2304 (a, 2)
		Reference	1753	6.7 (b,2)	13.4	3.6 (b,2)	1419 (b,2)
	Routine	Treatment	461	7.2(a, 2)	12.1	4.6 (a, 2)	528 (a, 2)
		Reference	512	9.0(a, 2)	21.1	3.4 (b,2)	449 (b,2)
	Storms	Treatment	1576	34.6 (a, 2)	110.4	9.8 (a, 2)	1763 (a, 2)
		Reference	1241	5.7(b,2)	8.1	3.6(b,2)	960(b,2)

* Letters are used to denote statistical significance within watersheds, numerals are used to denote statistical significance between watersheds, and different numerals or letters indicate statistical differences between similar sample types.

The ratio between turbidity and suspended sediment concentrations indicates that even though routine turbidities were statistically lower in the treatment watershed during pretreatment, the treatment watershed exhibited less sediment per turbidity value. The treatment and reference watershed's routine samples average and median TS ratio were not statistically different within watersheds classified by pretreatment and post-treatment periods (P=0.32, 0.26, 0.81, and 0.46, respectively).

Slopes were statistically significant within both watersheds. The R^2 values between turbidity and SSC were larger than the R^2 values between streamflow and SSC and ranged between 0.03 and 0.72. The largest correlation occurred in the treatment watershed post-treatment.

Prior to road construction both the reference watershed and treatment watersheds' turbidities peaked prior to peak streamflow then receded to pre-storm turbidity quicker than streamflow (Figure 1). Peak turbidity occurred prior to peak streamflow then would remain elevated up to and after streamflow returned to low flow levels (Edwards, P.J Submitted). After treatment, the treatment watershed's average stormflow turbidities declined toward pretreatment values. Slope (-0.06) and intercept (56.6) were both statistically significant ($P < 0.01$) and $R^2 = 0.12$. This equation yields, on average, a 21.9 NTU decrease per year from the intercept. A logarithmic decrease to average stormflow turbidity explained 5 percent more variation than a linear decrease.

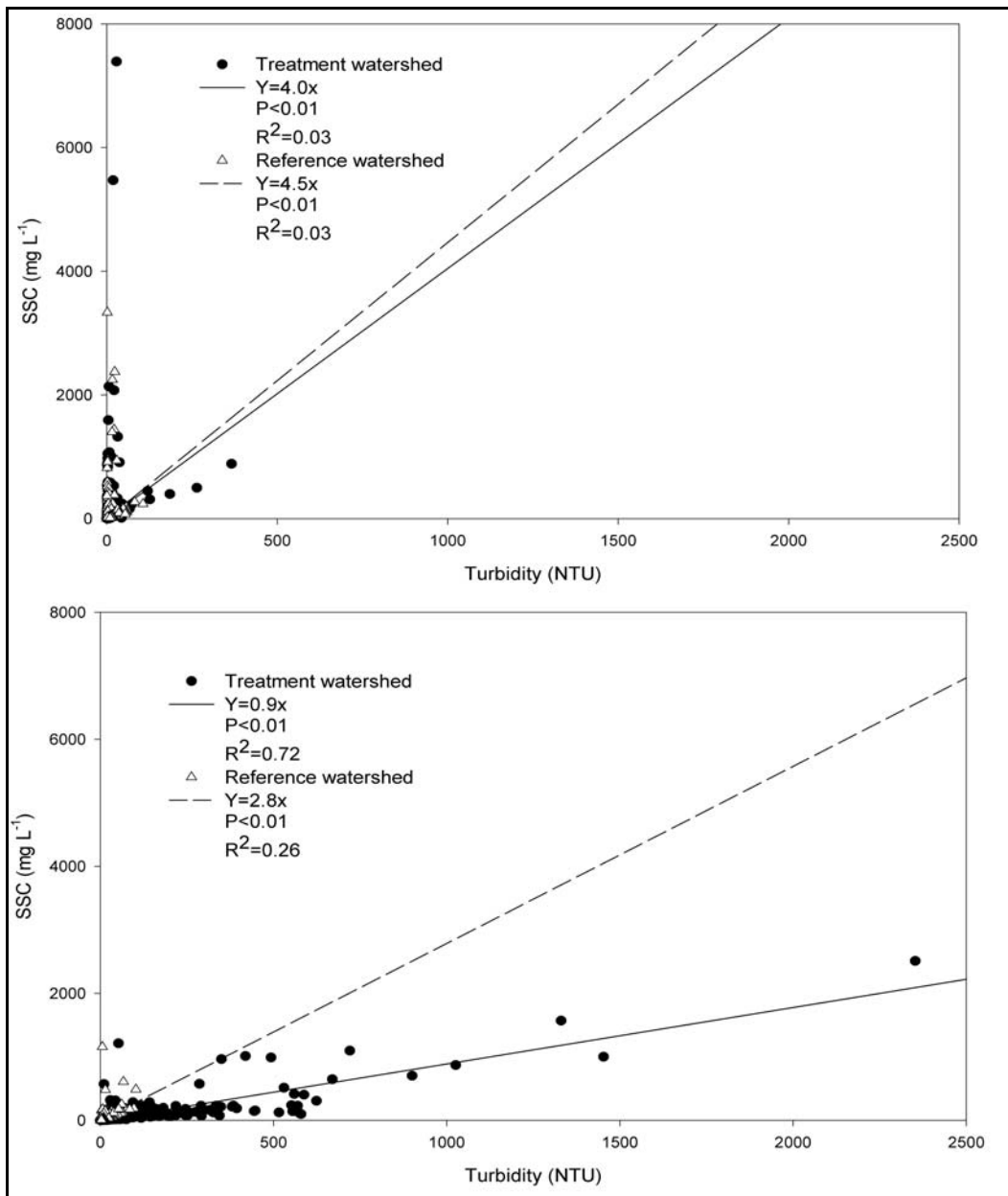


Figure 1. Storm samples SSC (y) measurement versus the turbidity (x) measurement.

DISCUSSION AND CONCLUSIONS

The reference watershed's storm and routine samples prior to construction were statistically more turbid than the treatment watershed's. The reference watershed's routine samples contained more sediment by weight relative to its turbidity index. The reference watershed produced less turbidity per sediment than the treatment watershed.

Average and median turbidities for these watersheds were below 5 NTU during pretreatment. Turbidity is noticeable around 5 NTU (Edwards Submitted) therefore; these streams normally have clear water. Prior to treatment, the treatment watershed's stream samples (2,680) exceeded 25 NTU 29 times or 1 percent of the time and the reference watersheds samples (3,059) exceeded 25 NTU 55 times or 2 percent of the time.

Maximum pretreatment turbidities were less than 400 NTU in both watersheds. They occurred during the largest storm events or during summer thunderstorms. Turbidities were elevated throughout the summer months during pretreatment. Stormflows that produced larger turbidities were relatively short-lived and storm samples overall produced clockwise hysteresis. Clockwise hysteresis is an indicator of a sediment supply limitation.

The reference watershed stayed within normal background levels after treatment even though the treatment watershed's average and median turbidities were above 5 NTU. Fourteen percent of the turbidities exceeded 25 NTU in the treatment watershed. Elevated turbidities were the result of stream crossing construction. Areas in stream crossings were less than 1 percent of the treatment watershed using 10 m aerial photographs.

Maximum turbidity in the treatment watershed following treatment reached 2,352 NTU and occurred during the initiation of a storm event. The treatment watershed's turbidities were less seasonally dependent, that is, the largest average monthly turbidity, occurred more so in late fall and during the winter months. The treatment watershed's stormflow turbidities were substantially elevated during the initiation of all storm events and are believed to be the result of precipitation impact remobilizing easily suspended channel sediment. Stormflows produced larger peak, average, and median turbidity values. Stormflow turbidities were relatively longer-lived and even maintained and increased after peak stormflow. Several storms produced counter-clockwise hysteresis towards the end of the first year post-treatment. Counter-clockwise hysteresis is an indicator of an energy limited situation and an abundance of sediment in the stream channel.

The TS ratio indicated that the treatment watersheds turbidities were significantly lower during pretreatment although less sediment per weight produced them. The reference watershed was transporting relatively more sediment with less turbidity. After treatment, the TS Ratio increased to 1.4 as the majority of turbidity values were larger than the SSC values. Towards the end of the post-treatment sampling period the TS ratio fell to around 0.5 as the majority of the turbidity values were half the SSC values. The TS ratio went from the highest levels to the lowest levels relative to pretreatment levels in 2 years or by the third year post-treatment. Although, turbidity is a better predictor of SSC than streamflow, the relationship between SSC and turbidity changed substantially between sample types, pretreatment and post-treatment periods, and levels of turbidity to warrant the use of several different regression relationships.

This study illustrates that significant increases to average turbidity during forest operations are not exclusively the result of similar increases to average SSC. For example, the treatment watershed routine and storm samples average SSC was 1.4 and 1.0 times the pretreatment levels post-treatment while average turbidity was 4.5 and 9.9 times the pretreatment levels post-treatment, respectively. By comparison, the reference watershed routine and storm samples average SSC were 0.7 and 0.5 times the pretreatment levels post-treatment while average turbidity were 1.0 and 1.2 times the pretreatment levels post-treatment, respectively. SSC measurements are an inadequate indicator of water quality as decreases to water clarity were probably the result of smaller inorganic and organic sediment that weighed less than average pretreatment sediments.

Stream crossings have to be constructed with better soil conservation practices. Time study analysis may be useful to help contractors increase road production and efficiency while decreasing costs associated with road construction while increasing soil conservation. Although, these crossings are legally defined as non-point sources of pollution, this study illustrates that very specific points along the road network were mainly responsible for water quality degradation. Preharvest planning for forest roads is necessary to reduce the amount of soil movement that is created by these disturbances.

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**AN ECONOMIC BENEFITS ANALYSIS FOR ACID MINE DRAINAGE
REMEDICATION IN THE WEST BRANCH SUSQUEHANNA RIVER WATERSHED,
PENNSYLVANIA**

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Julie Svetlik and Alan Collins
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ABSTRACT

This study comprehensively examines acid mine drainage (AMD) in the West Branch Susquehanna River, and attempts to estimate the potential economic benefits to be derived from AMD remediation in the watershed. The project includes five analyses: 1) contingent valuation survey to determine willingness-to-pay for stream restoration; 2) property price analysis to determine increases in property value after remediation; 3) local economic benefits of remediation project spending; 4) local economic benefits of increased recreational spending; and 5) a drinking water supply study to determine how drinking water sources and associated treatment costs are related to AMD in the watershed. Preliminary results indicate that AMD negatively impacts recreation and property prices in the area, and users of the watershed are willing to pay a positive amount to restore impacted streams. Benefit estimates can then be compared to remediation costs to determine feasibility of remediation. When this study is completed, we expect the final results to illuminate the complete cross section of economic impacts from AMD, justifying remediation dollars for all economic sectors.

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EXTERNALITY EFFECTS OF ACID MINE DRAINAGE ON RESIDENTIAL PROPERTY VALUES IN CLEARFIELD COUNTY, PA

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Agriculture & Resource Economics, West Virginia University

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ABSTRACT

This paper estimates the value of the negative externality created by acid mine drainage (AMD) on land prices in the West Branch Susquehanna (WBS) river basin. AMD is a stream quality issue prevalent in areas with a history of coal mining; in these regions, large numbers of abandoned mines discharge metal-laden groundwater into nearby streams. Streams impacted by AMD support little or no aquatic life, and in most cases metal concentrations in the water are so high that stream channels, rocks, and vegetation become discolored; this environmental problem compounds the existing economic stagnation faced by communities in historic coal-mining regions. Here we use a spatial lag hedonic model to estimate the coefficients of variables influencing residential property value in Clearfield County, PA, located within the WBS watershed. Parcel price is analyzed with respect to its structural, locational, and landscape characteristics, with particular attention paid to parcels located within 200ft of streams. Initial results indicate that proximity to AMD-affected streams has a negative and significant effect on land price per acre. Other factors affecting property price continue to be examined in detail in order to calculate the willingness-to-pay of homebuyers in the WBS for property near clean streams, and ultimately to calculate the benefits of restoring AMD-impacted streams in the WBS to viable trout fisheries.



Abstracts from workshops

- Introduction to Fluvial Geomorphology
- An Overview of the TMDL Process
 - An Overview of Virginia's TMDL Process
 - TMDL Issues of Special Interest
 - Stressor Identification and TMDL Development in Biologically Impaired Streams
 - Implementation Plans: Do They Work?
 - Issues with Sediment TMDLs
 - Living with TMDLs, an Industrial Perspective of Permitting Issues
 - Use Attainability Analysis – A Case Study
- Tools for Watershed Modeling and Assessment

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Workshop 1 INTRODUCTION TO FLUVIAL GEOMORPHOLOGY

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ABSTRACT

This two day workshop introduced attendants to issues such as watershed hydrology, erosion and sediment fate and transport, stream classification, hydraulic geometry and regional curves, and linkages between fluvial morphology and aquatic ecosystems. Two field trips were included in the workshop's program.



Workshop 2
AN OVERVIEW OF THE TMDL PROCESS

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AN OVERVIEW OF VIRGINIA'S TMDL PROCESS

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KEY WORDS: progress, challenges, and opportunity

ABSTRACT

The TMDL program is a watershed-based process required by §303(d) of the Clean Water Act and §62.1-44.19:4, et seq. Code of Virginia. EPA and the states are using the TMDL process as the primary program for the restoration of impaired waters. Also, the 2006 General Assembly directed the Office of the Secretary of Natural Resources to develop a plan to clean up Virginia's impaired waters. Virginia's TMDL restoration efforts are summarized in The TMDL Progress Report (2000 – 2006) posted at <http://www.deq.virginia.gov/tmdl/>.

In addition to meeting an ambitious Consent Decree schedule, Virginia's TMDL process has addressed and resolved numerous technical and legal issues. However, some issues remain and new ones emerge. The TMDL program has integrated with all other water quality programs, especially permits and Water Quality Standards.

One of our technical challenges has been developing allocations for those pollutants without promulgated numeric criteria. There are emerging issues with PCB and mercury TMDLs, atmospheric deposition, and the role the TMDL process has for impaired waters that cannot be fully restored.

TMDL WLAs are placing pollutant caps on regulated stakeholders. There is need to identify and document acceptable offsets and tracking procedures for TMDL WLA compliance.

The extensive public outreach component of the TMDL process has provided stakeholders a forum and an opportunity to participate in the impaired waters restoration process. Also, the public outreach effort provides ample opportunities for the stakeholders to participate in development of the procedures and guidance used in the TMDL process.

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Workshop 2
TMDL ISSUES OF SPECIAL INTEREST

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KEY WORDS: TMDL, daily loads, permitting

ABSTRACT

The rules governing the national total maximum daily load (“TMDL”) program are over twenty years old, have been successfully revised only once, and have been the subject of considerable controversy ever since EPA’s failed effort to overhaul them in 2000. Absent up-to-date rules, the national TMDL program has been shaped largely through guidance and, more recently, litigation.

In 2006, the D.C. Circuit Court of Appeals invalidated EPA’s long-standing interpretation that TMDLs could be developed using non-daily loads where appropriate to implement the applicable water quality standards. Although limited to one particular stream segment in the District of Columbia, this decision resulted in new national policy and a deluge of technical guidance from EPA on how to develop “defensible” daily loads.

In 2007, the Ninth Circuit Court of Appeals invalidated another long-standing interpretation about post-TMDL permitting for new sources. Although EPA has not yet officially reacted to this decision (e.g., seek re-hearing, seek Supreme Court review or accept as binding), the initial fall-out from the decision has called into question TMDL and permitting approaches across the country, including in Virginia.

As a delegated program, Virginia is heavily influenced by these national developments. However, Virginia is also governed by state-specific laws on TMDL implementation, use attainability analyses and permitting.

Unlike other environmental areas that are fairly mature, the TMDL program continues to evolve, with the potential for radical change in response to litigation. Unfortunately, many specific TMDL issues remain to be interpreted, either by the agencies or by the courts. This presentation covered the latest developments and forecast emerging issues.

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Workshop 2
**STRESSOR IDENTIFICATION AND TMDL DEVELOPMENT IN BIOLOGICALLY
IMPAIRED STREAMS**

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ABSTRACT

In development of TMDLs for biologically impaired streams, the West Virginia Department of Environmental Protection's Watershed Assessment Branch uses a weight-of-evidence approach to designate stressors to benthic macroinvertebrate communities. This process, accompanied by Tetra Tech (TMDL contractor), involves examination of monthly water quality data collected prior to TMDL development, which is contrasted to both water quality standards and developed biological thresholds. Quantitative evaluations of in-stream and riparian habitats are considered in stressor assignments as well as numeric ratings of potential impacts from various sources. For statistical classification of potential stressors to benthic macroinvertebrate communities, the WVDEP and Tetra Tech rely upon a null model, which references assemblages impacted by a sole stressor. Stressed communities, termed "dirty references," are compared to unknown assemblages in regard to statistical similarity and probability. Categories of biological stress designated through the weight-of-evidence approach include organic enrichment, ionic stress, sedimentation, acidic pH/dissolved metals.

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Workshop 2
IMPLEMENTATION PLANS: DO THEY WORK?

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KEY WORDS: TMDL implementation, NPDES permit, nonpoint source reduction, watershed restoration

ABSTRACT

Federal regulations adopted by the U.S. Environmental Protection Agency in 1985 continue to govern the TMDL process nationally. Those rules do not require development of implementation plans. Nationally, TMDL implementation is driven largely by the need for NPDES permits to comply with waste load allocations contained in TMDLs, and by various voluntary or state or federally-financed nonpoint source programs.

Virginia chose to require the development of implementation plans for TMDLs. The 1997 Water Quality Monitoring, Information and Restoration Act requires the development of implementation plans for TMDLs developed for Virginia waters. However, the development of the plan and the actual implementation are both dependent upon available resources and are not subject to federal deadlines familiar to TMDL practitioners. It is too soon to tell how implementation will work in Virginia.

It remains to be seen whether the requirement of an implementation plan will lead to faster and more complete recovery of water quality in Virginia than in states without such a requirement. The primary tool for implementing TMDL recovery plans in Virginia may remain tightening down on NPDES permits, at least in watersheds with point sources contributing to impairments. Increased state funding for point source and nonpoint source reductions should contribute to effective implementation of TMDLs in Virginia.

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Workshop 2
ISSUES WITH SEDIMENT TMDLs

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KEY WORDS: monitoring, sediment, TMDL

ABSTRACT

Sediment transport is a natural function of stream systems. Too much, or too little, sediment will create problems for the biological components that have come to depend on the natural supply, rhythm, and movement of sediment. Background rates and loads and, therefore, target TMDL loads undoubtedly will vary by physiographic, geomorphic, climatic, and geologic factors.

Sediment impacts the biological community by disrupting and degrading the spawning habitat of organisms that occupy the interstitial spaces in the stream bottom, by reducing the flow of oxygen through gravel beds, by interrupting the feeding mechanisms of filter feeders, and by reducing the visibility necessary to avoid predators.

The issues discussed included

- Generally available ambient monitoring data reveals little about sediment impacts.
- Runoff monitoring data, if available, generally verifies that storms produce sediment loads.
- There are no state sediment standards, and target loads corresponding to non-impaired conditions are likely to be a function of multiple factors.
- Reference watersheds are often used to set target loads.
- Channel erosion is not well quantified.
- The primary evidence for sediment as a stressor (in our area) is comprised of visual observations of available sources, poor habitat metrics, and low %haptobenthos metric values.
- In mining areas, the presumption that use of best practical technology (sediment detention ponds) will result in compliance with a maximum daily permit concentration of 70 mg/L is not consistent either with simulated loads or DMME monitoring.

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Workshop 2
LIVING WITH TMDLS, AN INDUSTRIAL PERSPECTIVE OF PERMITTING ISSUES

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KEY WORDS: TMDL, permitting, compliance

ABSTRACT

Birthing pains develop during the implementation of any new, or significantly revised, regulatory program. While TMDLs have been a reality within the provisions of the Clean Water Act since its inception, it has only been since 1999 that the State of Virginia began the process of seriously developing and implementing TMDLs. The first group of TMDLs for the coalfields of Southwestern Virginia was approved by the State Water Control Board (SWCB) in 2004, and implementation plans (IP) were developed for two of these watersheds. Of the two implementation plans, one has been approved by the SWCB.

The coal industry is one of the more heavily regulated industries. We are constantly forced to adapt to a changing regulatory climate, and the TMDL process is another step in the evolution of our industry. Some of the issues facing our industry under the TMDL program are:

- Obtaining permits within TMDL watersheds for continued or new operations.
- Meeting load allocations as defined within the TMDL(s)
- Maintaining compliance with NPDES discharges and overall allocations within the watershed.
- Addressing offset impacts to reduce or offset loading.
- Better defining the use, or function, of the watersheds.

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Workshop 2
USE ATTAINABILITY ANALYSIS – A CASE STUDY

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KEY WORDS: use attainability analysis, designated use, water quality standards, benthic impairment

ABSTRACT

The U.S. Environmental Protection Agency believes that it is important for States to have the right uses designated to each waterbody segment. States need to invest in putting in place more refined use designations along with differentiated criteria to protect those uses. A Use Attainability Analysis (UAA) is a structured scientific assessment of the factors affecting the attainment of the use which may include physical, chemical, biological and economic factors. In July 2006 Virginia implemented legislation facilitating the conduct of a UAA. The Virginia Coalfields TMDL Group subsequently requested permission to conduct a UAA for the Aquatic Life Designated Use on Straight Creek in Lee County, Virginia. The Virginia State Water Control Board approved this request in March 2007. This paper presented an overview of the legal and technical issues associated with the conduct of this UAA.



Workshop 3
TOOLS FOR WATERSHED MODELING AND ASSESSMENT

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ABSTRACT

This workshop demonstrated the capabilities and potential of the Watershed Modeling and Characterization System (WCMS), developed by the Natural Resource Analysis Center at West Virginia University. The WCMS runs within ESRI's ArcGIS 9.x GIS software platform, and includes several hydrological and water quality modeling tools designed to aid in water resources management and assessment.

This full morning workshop included presentations and discussion detailing the WCMS, as well as hands-on computer experience in using WCMS tools and functions. For participants, familiarity with ArcGIS 9.x was helpful, but not required.

The presentations for this workshop included an overview of the overall system goals, practical applications of the WCMS to management of water resources, and a description of spatial datasets used with the WCMS. Following each short presentation section, individual computers were available for participants to use the WCMS with sample datasets.

Highlighted functions of the WCMS included stream flow estimation, low flow estimation, watershed delineation, land cover tabulation, flow path modeling, and other related tools. Additional water quality modeling related functions demonstrated included expected mean concentrations modeling for stream pollutant loadings, fate transport modeling, and identification of potentially affected streams. Finally, the workshop also included discussion of newly added functionality, including incorporation of externally linked water quality databases and HSPF modeling.

