Ohio Water Resources Center
Annual Technical Report
FY 2015

Introduction

Pursuant to the Water Resources Research Act of 1964, the Ohio Water Resources Center (WRC) is the federally-authorized and state-designated Water Resources Research Institute for the State of Ohio. The Ohio WRC was originally established at The Ohio State University in 1959, as part of the College of Engineering's Experiment Station. The Ohio WRC continues to be administered through the College of Engineering, in the Department of Civil, Environmental, and Geodetic Engineering.

The Ohio Water Resources Center promotes innovative, water-related research through research grant competitions, coordination of interdisciplinary research proposals, and educational outreach activities. Ohio WRC forges key relationships by connecting researchers and stakeholders. In Ohio – when there is a crisis, such as the detection of microcystin in drinking water- the Ohio WRC teams up with the Ohio Sea Grant Program, Stone Lab, the National Water Quality Research Center, the Lake Erie Research Center and other key organizations and research universities to provide critical support and operational inputs to federal, state and local policy makers, local operators, key professional associations, and civic and educational groups.

Ohio WRC sponsored researchers enable ecologically and socially sound water management by investigating the sources of nutrients and algal blooms in our environment, developing novel methods and technologies to reduce nutrients and other pollutants in water, and characterizing and monitoring the effects of energy development on water resources. By funding researchers early in their careers and developing powerful alliances with partner institutions, Ohio WRC seeds innovative approaches that foster impactful outcomes.

Ohio WRC reaches out to water professionals, educators, and citizens to ensure current and future citizens are water smart. Ohio WRC leaders are active in local and national water research, education and policy organizations such as the Ohio Water Resources Council, Water Management Association of Ohio, National Institutes of Water Resources and University Council on Water Resources.

Research Program Introduction

The Ohio Water Resources Center consistently invests in water related research in the State, growing the number of principal investigators involved in Ohio's water issues, and educating the next generation of water professionals by funding student work on water research projects. Over this past year's reporting period, we sponsored seven new projects and administered five ongoing research projects conducted at four different Ohio universities that totaled $626,447 in research funding (direct and cost share). The PI's for these projects are seven Assistant Professors, two Research Scientists and three Associate Professors. In total, this research helped support directly and indirectly thirty-three students majoring in environmental engineering, biological sciences, environmental studies, restoration ecology, geology, chemistry, public health, public policy, natural resources and other water related fields.

The new funded research projects entail studies of important Ohio water resources problems. For example, Dr. Gajan Sivandran from the Ohio State University applied advanced hydrological modelling techniques to assess spatial and temporal dynamics of non-point source pollution in Ohio. Nine projects were finalized during this fiscal year, three projects (funded via non-federal sources) will be continued into next year. These include Dr. Cheng's project on field-based nutrient treatment, Dr. Bohrer's project on methane emissions from lakes and Dr. Bohrerova's citizen's science project called "Adopt Your Waterway".

In summary, Ohio WRC administered 12 research projects this reporting period, 7 of which were funded or co-funded by USGS 104(b) base grants. This resulted in the training of 34 students, 10 manuscripts in development, submitted or accepted in peer-review journals and 31 presentations or posters at local, national or international conferences. In this reporting period our PI's have been able to secure an additional $503,082 in research awards using data generated with Ohio WRC funding.

Assessment of a Novel Application of Biochar to Improve Runoff Water Quality from Vegetated Roofs

Basic Information

Title:	Assessment of a Novel Application of Biochar to Improve Runoff Water Quality from Vegetated Roofs
Project Number:	2013OH297B
Start Date:	3/1/2013
End Date:	2/28/2016
Funding Source:	104B
Congressional District:	1
Research Category:	Water Quality
Focus Category:	Hydrogeochemistry, Nutrients, Water Quality
Descriptors:	
Principal Investigators:	Ishi Buffam, Dominic L Boccelli

Publications

1. *Hochwalt, P. and I. Buffam. April 2014. “Can biochar increase nutrient binding and water holding capacity in vegetated roof growing medium?” University of Cincinnati 2014 Undergraduate Research Symposium, Cincinnati OH, USA (poster presentation).
2. *Divelbiss, D., P. Hochwalt, M. Mitchell, D. Boccelli, and I. Buffam. May 2014. “Black Is The New Green - Enhancing Green Roof Performance With Novel Substrate”. Confluence Water Technology Innovation Cluster, 2014 Water Symposium, Covington, KY, USA (oral presentation).
3. Wright, P., P. Hochwalt and I. Buffam. April 2014. “A batch study of the sorption kinetic properties of biochar amendment to vegetated roof growing medium” University of Cincinnati 2014 Undergraduate Research Symposium, Cincinnati OH, USA (poster presentation).
4. Hochwalt, P., M.E. Mitchell, D. Boccelli, D. Divelbiss, and I. Buffam. November 2014. Biochar enhances water and nitrogen retention in green roof substrate. Cities Alive: 12th annual Green Roofs and Walls Conference, Nashville, TN, USA (poster presented by I. Buffam)
5. Buffam, I. and M.E. Mitchell. 2015. Nutrient cycling in green roof ecosystems. Chapter 5 in R. Sutton, ed. Green Roof Ecosystems, Springer, New York (May 2015; Research from this project is mentioned in the paper, though not the focus of the entire chapter)
6. Buffam, I. and *M.E. Mitchell. October 2015. Is black the new green? Impact of biochar amendments on water and nutrient retention in vegetated roof substrate. Cities Alive: 13th annual Green Roofs and Walls Conference, New York City, NY, USA.
7. *Shaw, C., *A. Kosielski, *M.E. Mitchell and I. Buffam. October 2015. Impact of Biochar- Amended Substrate on Water Holding Capacity and Evapotranspiration Rate in Green Roof Test Plots. Cities Alive: 13th annual Green Roofs and Walls Conference, New York City, NY, USA (poster, presented by I. Buffam).

Final Report

Assessment of a Novel Application of Biochar to Improve Runoff Water Quality from Vegetated Roofs

Project G101643/G200355: 3/1/2013 – 5/31/2016

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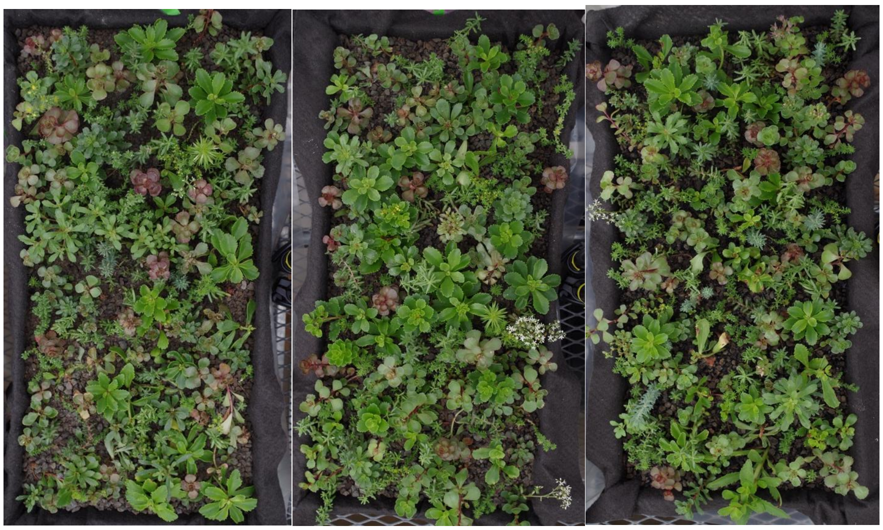
University of Cincinnati



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1. Summary

Vegetated (green) roofs – a type of urban green infrastructure – have been demonstrated to improve stormwater retention and provide energy savings, but can also serve as a source of inorganic nutrients nitrate (NO_3^-), ammonium (NH_4^+) and phosphate (PO_4^{3-}) through runoff. An early study using biochar – a type of activated carbon – within the green roof substrate (soil mix) suggested that an amended soil mix could improve the effluent water quality from vegetated roofs. The *overarching objective of this project* is to improve our understanding of the water quality benefits associated with the use of a biochar-amended substrate mix within vegetated roof technology. Our *central hypothesis* is that biochar can enhance the ability of vegetated roofs, which can already reduce the quantity of stormwater runoff, to decrease nutrient loading by binding nutrients as the runoff passes through the amended green roof substrate. In the first year of the project (2013-14), using column experiments in the lab we demonstrated that the incorporation of biochar substantially increased the water holding capacity of the substrate, reduced and delayed the efflux of NH_4^+ and slightly delayed the passage of NO_3^-, but had little effect on PO_4^{3-}. We also carried out experiments on the dynamics of sorption kinetics, which suggest a two-phase sorption mechanism onto biochar, the slower process taking several days to reach equilibrium. In the second phase of the project (2014-2015), our research group carried out a pilot study on green roof test plots and developed a method for making continuous measurements of evapotranspiration on the plots. In the final phase of the project (2015-2016), we scaled the experiment up and measured the effect of biochar on nutrient retention, water retention, and plant vitality in green roof plots, carried out detailed batch experiments on sorption/desorption equilibria of standard and biochar-amended green roof substrate, and developed a mathematical model to describe the nutrient sorption dynamics. Our results indicate that biochar has excellent potential as a low-cost amendment to green roof substrate to improve downstream surface water quality by increasing water retention and by strongly binding NH_4^+, one of the nutrients of concern.

2. Problem and Research Objectives

A significant issue many urban centers face is the direct discharge of untreated sewage into receiving waters due to overburdened and antiquated combined sanitary and stormwater sewers. While conventional grey infrastructure approaches to mitigating combined sewer overflows (CSOs) tend to be disruptive and costly, the use of urban green infrastructure (UGI) – generally defined as implementations (often vegetated) to reduce stormwater surface runoff by increasing infiltration and evapotranspiration – can mitigate overflow events and the associated deleterious impacts, while contributing other co-benefits. Green infrastructure techniques have been gaining traction for reducing urban stormwater runoff and improving water quality (e.g., NYC Department of Environmental Protection, 2011), and are one of the best management practices for CSO reduction recommended as part of the integrated green infrastructure program by Cincinnati's Metropolitan Sewer District (www.projectgroundwork.org). Vegetated (green) roofs – a type of UGI – are becoming increasingly popular with for example >20% coverage of the flat roof area in cities like Stuttgart, Germany. These installations are expected to continue to proliferate in the near future with stated goals of 20% coverage of large buildings in Washington, D.C. by the year 2025 (Deutsch et al. 2005) and 50-70% coverage of city owned buildings in Toronto and Portland (Carter and Laurie 2008). Vegetated roofs have been demonstrated to improve stormwater retention (Bliss et al, 2009; Getter et al, 2007; Carter and Rasmussen, 2006, Mentens et al, 2006; Van Woert et al, 2005) and provide energy savings (van Woert et al, 2005), but also serve as a source of organic carbon, nutrients, and metals through runoff (Berndtsson et al., 2010). There is concern that the water quality benefits of green roofs related to reduced CSO events, may be offset by the direct contribution of organic carbon and nutrients in runoff from these systems. Runoff from vegetated roofs often contains very high concentrations of nutrients, particularly phosphorus but also organic carbon and sometimes inorganic nitrogen (Berndtsson et al. 2009; Berndtsson et al. 2010; Oberndorfer et al. 2007; Buffam and Mitchell 2015; Gregoire and Clausen 2010). Thus, there is a need for further study into the potential water quality benefits and, potential, negative impacts associated with nutrient release. These are at high enough levels to contribute to eutrophication in downstream waterways, and to date no clear solution to this ecosystem disservice has been found and tested.

A novel potential solution to this problem has been identified: the integration of biochar into the vegetated roof substrate. Biochar is the term given to biomass, such as wood, which has undergone pyrolysis to generate a carbon-rich product. The production of biochar is similar to the process which creates charcoal but is distinct in that the end product is meant to be used as a soil amendment. The purpose of this soil amendment is to increase soil productivity, sequester carbon, and filter percolating water (Lehmann, 2009). Adding biochar to soils can improve the ability of the soil to absorb phosphorous (Lehmann, 2002; Beaton, 1960; Downie, 2007), absorb metals by increasing cation exchange rate (Lehmann, 2009), and increase water holding capacity (Piccolo, 1996).

Biochar is a proven technology to improve water quality but it has not been extensively challenged in the treatment of green roof effluent. The multifaceted claims of biochar, specifically, improved soil fertility, carbon sequestration, and improved effluent water quality (Lehmann, 2009), suggest the technology could reduce threats to ecosystems receiving runoff, create cost savings due to reduced green roof maintenance through nutrient retention, and increase effectiveness of green roofs to retain water. Evaluation of this application is necessary to determine the true effectiveness of this possible game changing technology for green roofs. One previous study (Beck et al., 2011) carried out in Portland, OR has shown qualitatively that biochar is capable of improving effluent water quality (i.e. phosphorous, nitrate, organic carbon) and reducing runoff volume in green roofs. However, this study did not examine the temporal dynamics of sorption, nor the changes in performance by varying biochar concentrations in the media, instead using a fixed proportion of 7% biochar.

The *overarching objective of this project* is to improve our understanding of the water quality benefits associated with the utilization of a biochar-amended soil mix within vegetated roof technology. Our *central hypothesis* is that biochar can enhance the ability of vegetated roofs to improve water quality by binding nutrients (N and P) as the runoff passes through the amended green roof medium. While vegetated roof technology has been demonstrated to reduce rainfall runoff, additional research has demonstrated a potential degradation in the effluent water quality (Oberndorfer et al. 2007; Berndtsson et al. 2010). Recent research has shown a net leaching of dissolved nitrogen and exceptionally high levels of inorganic phosphorous in green roof runoff, both from full-scale green roofs and small-scale experimental plots (Buffam and Mitchell 2015). The use of biochar – an inexpensive activated carbon – is expected to improve the ability of vegetated roofs to retain nutrients. The *rationale* for understanding the water quality impacts of a biochar-amended soil medium is to evaluate the benefits for use within vegetated roofs an part of an integrated stormwater management plan, which would benefit designers and planners in assessing the potential impact to water quality conditions within a regional design setting.

Our project had three associated objectives: 1) evaluating the abiotic capabilities for nutrient and stormwater runoff retention due to enhanced sorption properties of biochar-amended soil medium via column reactors; 2) evaluating biotic and abiotic capabilities for nutrient and stormwater runoff retention due to enhanced sorption properties of a biochar-amended soil medium in vegetated green roof plots; and 3) developing a model for representing the hydraulic and water quality performance of vegetated roofs with and without biochar. In the first year of the project we carried out the first objective, while the second and third objectives were carried out during years 2-3.

3. Column Studies on biochar-amended vegetated roof substrate (2013-2014)

3.1 Methodology

3.1.1 Column experiment design. We designed and carried out column studies in order to determine the nutrient holding capacity and water-holding capacity of biochar-amended vegetated roof substrate. Fixed bed column reactors (7 cm diameter, Figure 1) were packed with four different treatments in duplicate of combined biochar and commercial green roof media (with biochar proportions of 0%, 2%, 7%, and 14% of total weight) at 10 cm of media depth to conform to common extensive green roof construction. Biochar samples used were derived from a wood-based feedstock from chips or grounds, 3 mm minus, >70% carbon sorption >8% butane ash up to 23% but with low buffering at 500°C. The growing medium used was a proprietary aggregate based extensive blend from Tremco Roofing Inc, (Cincinnati, OH), sieved through a 2 mm sieve.

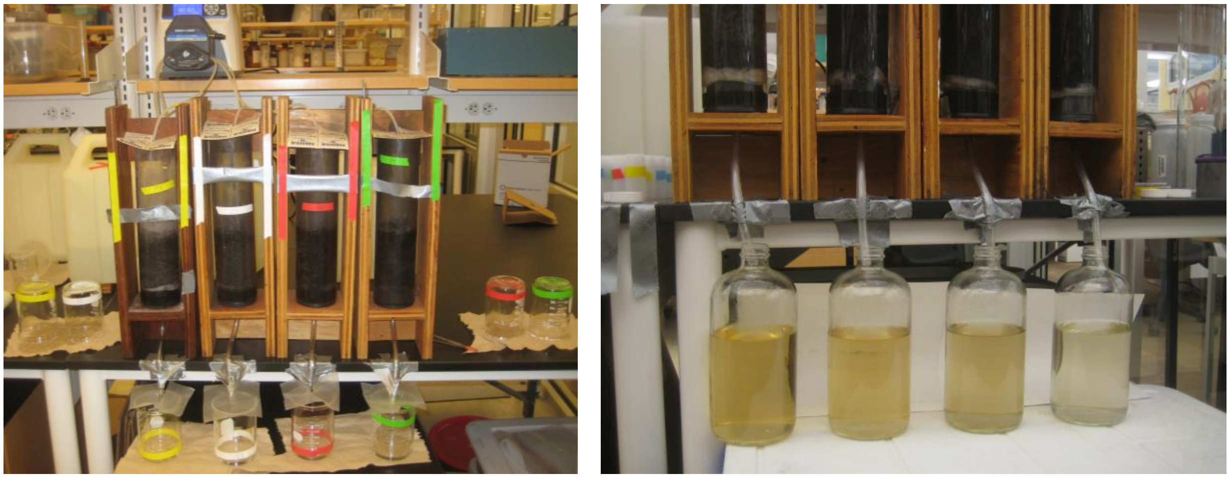


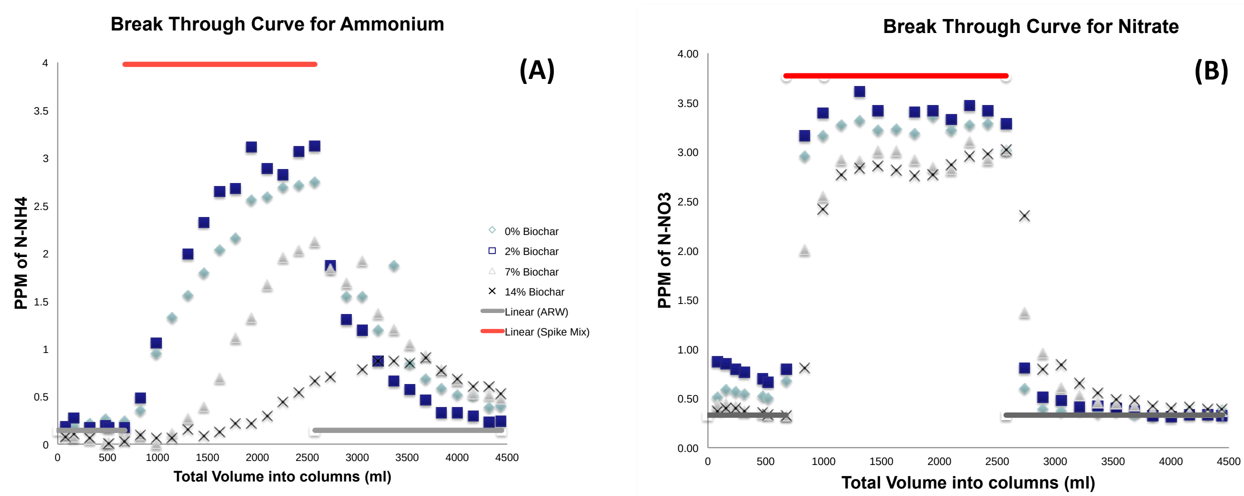
Figure 1. Left panel: Column setup showing pump, four columns varying in % biochar integrated into substrate, and beakers collecting continuous runoff water samples. **Right panel:** Collection bottles showing, from left to right, runoff water from 0%, 2%,7%, & 14% by mass biochar/ substrate mixtures. Note the decreasing brown color for the high biochar mixtures, indicating removal of dissolved humics.

The columns were flushed during 24 hours with 1 year's worth of artificial rain water (ARW), equivalent to 1 meter depth, immediately prior to the beginning of the experiment. This was done so that the experiment would not be dominated by the initial release of nutrients from the substrate and would instead behave as a more established green roof. The concentrations of compounds in the ARW were determined by using the NADP precipitation averages from 2007 to present for the Oxford, Ohio station (OH09) (http://nadp.sws.uiuc.edu/NTN/ntnData.aspx). The flow rate was then reduced to 1cm/h to simulate a heavy, yet realistic, rain event. This treatment was continued for 24 hours, followed by a "heavy nutrient" mix containing ARW enriched with approximately 3 mg/L each of NH4-N, NO3-N, and PO4-P. This heavy nutrient mix was added at the same flow rate for 48 hours. This was determined to be long enough to observe the breakthrough of nutrients. This was followed by a 48h flush out using ARW. Thus, the total experiment length was 120 hours.

3.1.2 Water Quality Collection and Analysis. Water samples were collected at 1 hour intervals throughout the entire experiment, and enough samples analyzed to adequately construct the breakthrough curves (in practice, typically one sample every 4 hours). We collected a total of 120 per treatment (n=4) per trial (n=2) for a total of 960 samples. Column effluent and batch samples were collected in acid-washed high-density polyethylene containers, filtered at 0.45 μm, frozen to preserve and subsequently analyzed for the concentrations of the inorganic nutrients ammonium, nitrate and phosphate. Colorimetric techniques were used at a microplate scale (Ringuet et al. 2010) for nitrate (Doane and Horwath, 2003), ammonium (Weatherburn, 1967), and phosphate (Lajtha et al., 1999). Initial pilot study samples for the concentration of metals common in the urban environment including copper (Cu) and zinc (Zn), revealed that the green roof substrate was a sink for these metals when loading occurred at a level to be expected in urban environments. Even in the absence of biochar, effluent concentrations of these metals were below the detection limit (Atomic Absorption spectroscopy). Therefore, detailed breakthrough curves were not generated for these elements, and focus was rather placed on inorganic nutrients, which have been shown to be a pervasive issue in green roof effluent.

3.2 Principal Findings

In summary, we found that the incorporation of biochar into vegetated roof substrate substantially reduced and delayed the efflux of NH_4^+ and slightly reduced and delayed the passage of NO_3^-, but had little effect on PO_4^{3-} (Figure 2). Biochar also increased the water-holding capacity of the substrate (Figure 3), which has important implications for the stormwater runoff reduction potential of green roofs. When averaged over the entire 5-day experiment, the volume-averaged mean concentrations (directly proportional to total flux) were reduced in the high-biochar treatment by up to 75% for ammonium and 17% for nitrate, while all columns were a slight net source of phosphate regardless of biochar amendment (Figure 4).



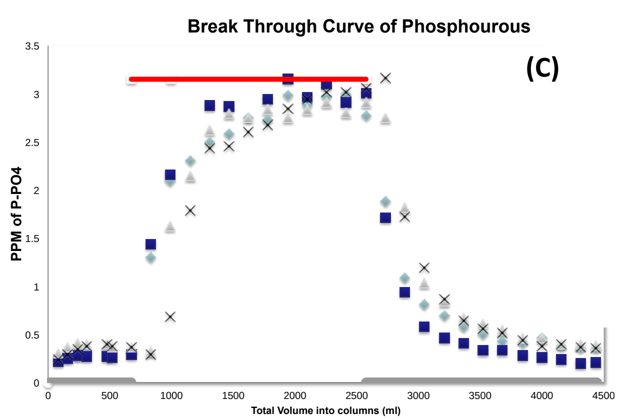


Figure 2. Nutrient break through curve of (A) ammonium, (B) nitrate, and (C) phosphate for vegetated roof substrate columns amended with 0%, 2%, 7%, & 14% w/w biochar. Each point represents an effluent sample taken every 4 hours, which was then analyzed for nutrient concentration. The hard lines (grey and red) represent the concentration of the input water over the length of the line (1 day with ARW, 2 days with ARW+high nutrients, and 2 days with ARW again).

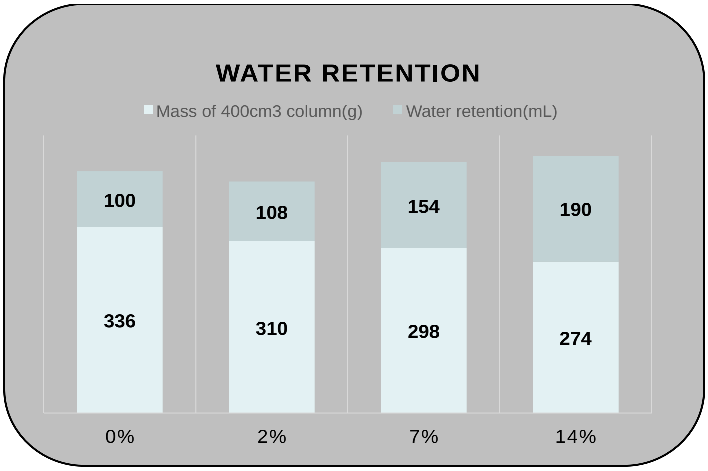
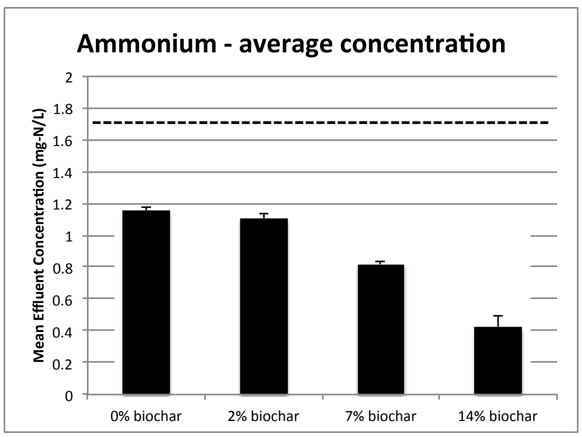
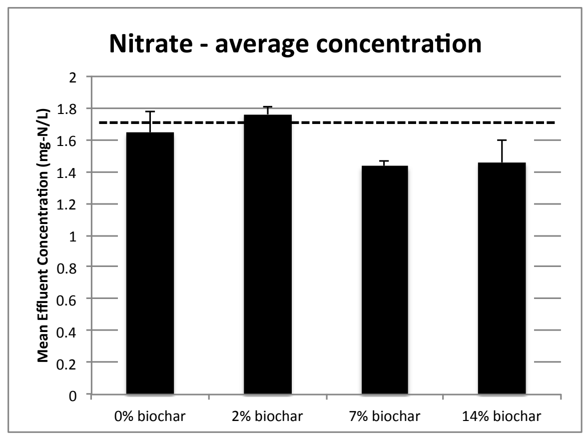


Figure 3. The water holding capacity was measured by the wet weight of the columns at the finish of the experiment. Increasing biochar had the effect of increasing water retention, with the 14% biochar treatment nearly doubling the water retention relative to the biochar-free control – this in spite of a lower initial mass for the high biochar column.





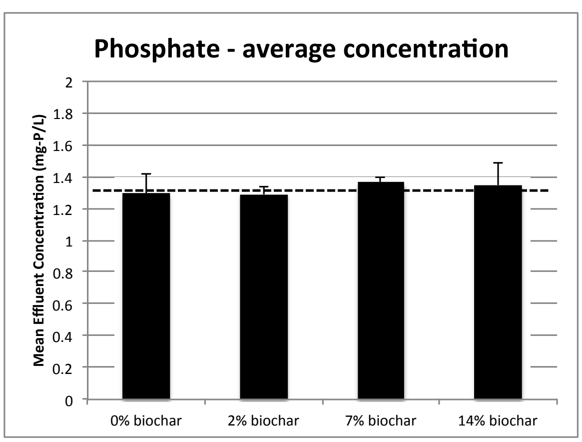


Figure 4. Volume-averaged mean effluent nutrient concentrations for the entire 5-day experiment, for columns varying in biochar % integrated into green roof substrate. Error bars represent standard error of the mean for duplicate trials. The horizontal dotted lines represent the volume-averaged influent "precipitation" concentration in the experiment. Substrate alone resulted in a 34% reduction in NH4-N, with higher biochar resulting in a reduction of up to 76% of NH4-N. NO3-N fluxes were not affected by substrate alone, but the higher biochar treatments resulted in a 17% decrease in NO3. PO4-P fluxes were not affected significantly by either substrate alone, or biochar amendments; though all columns were a slight source of PO4.

4. Batch experiments to determine sorption kinetics (Spring 2014)

4.1 Methodology

Pilot batch studies were carried out to determine the sorption kinetics for nutrients in vegetated roof substrate with and without biochar. Batch experiments were run with a known ratio of biochar to growing medium in solutions with a known concentration of three nutrients - ammonium, nitrate, and phosphate. The solutions contained either artificial rainwater (ARW) containing nutrients at levels observed in local precipitation, or ARW + 20 mg/L of NH4-N, NO3-N, and PO4-P. Samples were taken periodically and analyzed for nutrient concentration.

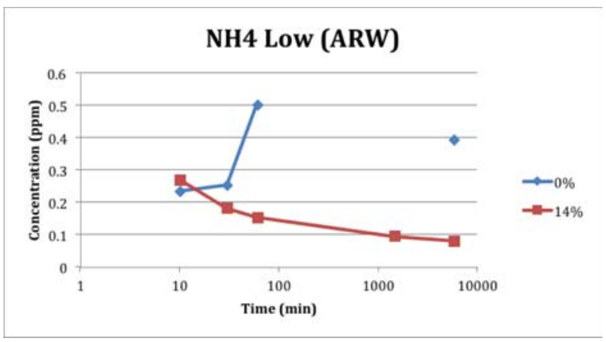
Two sample levels were prepared per treatment, 0% biochar and 14% biochar, measured by mass. The 0% contained no biochar and 1.00 g of growing medium. The 14% contained 0.14 g biochar and 0.86 g growing medium. The mixtures were placed in 50 mL centrifuge tubes along with 40 mL of either ARW or the 20 ppm nutrient spike, depending on the treatment. Five centrifuge tubes were used per sample per treatment (20 total). Once the treatment water was added to the mixtures, the tubes were immediately placed on a shaker table shaking 100 rpm at 25°C. The table was covered to ensure no light penetrated the tubes to control for any photocatalytic activity that may ensue. Samples were taken at five time points- 10 min, 30 min, 1hr, 24 hr, and 96 hr- and vacuum filtered through a 0.45 μl filter then immediately frozen for further analysis. A second iteration of the same experiment also included a 100% biochar treatment, and added 12 hr, 48 hr, and 120 hr timepoints as well as additional replicates (triplicates), but was otherwise identical to the pilot study. Nutrient concentrations of the effluent were determined using a spectrophotometer (Figure 5) as described in *Section 3.1.2* above.

Figure 5. Left panel: Undergraduate research assistant Pat Wright pipetting out standards for use in nutrient analysis. **Right panel:** Side view of five 96-well microplates showing color development used in nitrate analysis.

4.2 Principal Findings

The objective of the batch study was to determine the kinetics and equilibrium concentrations for standard and biochar (14%)-amended substrate, subjected to two different initial nutrient concentrations in solution (ARW and 20 ppm). For the small-scale batch study, the hypothesis was that that for ammonium, nitrate, and phosphate, the concentration of nutrients in the effluent will be reduced with a higher ratio of biochar to growing medium. This was corroborated for ammonium (Figure 6), low concentrations of nitrate, and high concentrations of phosphate but rejected for low concentrations of phosphate because final concentrations were higher than initial values (data not shown). The 14% biochar amendment was quite effective at binding and reducing concentrations of ammonium from ARW- though equilibrium was not reached after 96 hours as well as reducing concentrations from a 20 ppm spike and coming to equilibrium in 1 hour (Figure 6). This conclusion is congruent to Yao et al. (2011) finding in that biochar does have the ability to bind contaminants from water. This is also supported by the effect biochar had in reducing nitrate concentrations from ARW (data not shown). With phosphate, the opposite occurred and phosphate was leached out of the substrate into the ARW, as was seen by Buccola (2008). However, when initial phosphate concentrations are increased to 20 ppm, biochar did show the capability to bind phosphate with equilibrium reached at 1 hour (data not shown).

In the batch study, the biochar appeared to come to an initial equilibrium with the water for several hours then followed by a later decrease in concentration. This may indicate that multiple equilibria exist that could include binding sites on the surface of the biochar as well as deeper within the particle. This would entail a fast and slow cycle, where the slow cycle is the binding of nutrients to the surface sites, and the slow cycle is the nutrients diffusing further into the biochar particles to inner binding sites. This would seem plausible because the diffusion process through solids takes longer than surface binding and would account for the ~23 hour period of equilibrium. As a consequence of the complicated kinetics, estimated time to equilibrium varied substantially among treatments (Table 1; Table 2), but an initial equilibrium was generally reached within 24 hours for all analytes regardless of starting concentration.



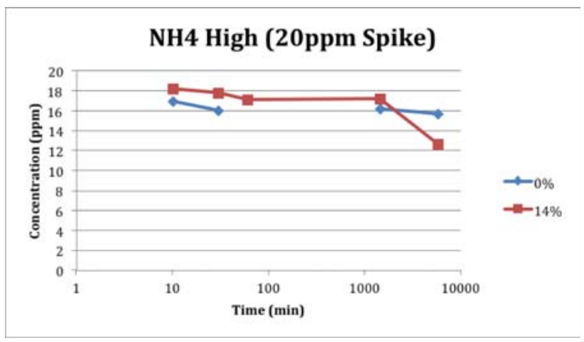


Figure 6. Left panel: The addition of 14% biochar to substrate decreases the concentration of NH_4 in ARW while substrate alone is a source of NH_4. **Right panel:** For the high ammonium spike solution, both substrate alone and substrate + 14% biochar result in a gradual decrease in ammonium over time, presumably due to sorption. A fast initial drop (within 1 hour) followed by continued gradual decline in both experiments suggests multiple mechanisms of sorption. Missing points represent missing analytical samples in this pilot study.

Table 1. Approximate time of equilibrium reached for each analyte and treatment, in the pilot batch kinetic study.

Analyte	0% ARW	14% ARW	0% 20ppm	14% 20ppm
NH_4	≈ 2 hrs	N/A	30 min	>24 hr
NO_3	1 hr	>24 hr	N/A	N/A
PO_4	N/A	24 hr	>30 min	1 hr

Table 2. Approximate time of equilibrium reached for each analyte and treatment, in the second batch kinetic study.

Analyte	0% ARW	14% ARW	100% ARW	0% 20ppm	14% 20ppm	100% 20ppm
NH_4^+	30 min	1 hr	1 hr	12 hr	12 hr	1 hr
NO_3^-	24 hr	48 hr	12 hr	1 hr	1 hr	1 hr
PO_4^{3-}	12 hr	24 hr	24 hr	12 hr	12 hr	12 hr

5. Effect of biochar pretreatment on water holding capacity (Spring 2015)

5.1 Methodology

We designed and carried out a column study of the water retention capacity of green roof substrate vs. biochar of differing particle size distributions. Fixed bed column reactors (7 cm diameter, Figure 1) were packed with four different treatments, at 5cm depth: commercial green roof substrate, and three different treatments with 100% biochar, differing only in the size distribution of the biochar particles. One column was filled with raw biochar including some larger pieces, one was filled with biochar that had been sifted to have particles smaller than 2mm and one with biochar that had been blended in a blender on high for 3 bursts of 15 seconds. All columns were treated with a 2.5 cm/hour rainfall by using a Cole Palmer Masterflex L/S vacuum pump for approximately 3.5 hours. Biochar samples used were derived from a wood-based feedstock from chips or grounds, 3 mm minus, >70% carbon sorption >8% butane ash up to 23% but with low buffering at 500°C. The growing medium used was a proprietary aggregate based extensive blend from Tremco Roofing Inc, (Cincinnati, OH), sieved through a 2 mm sieve.

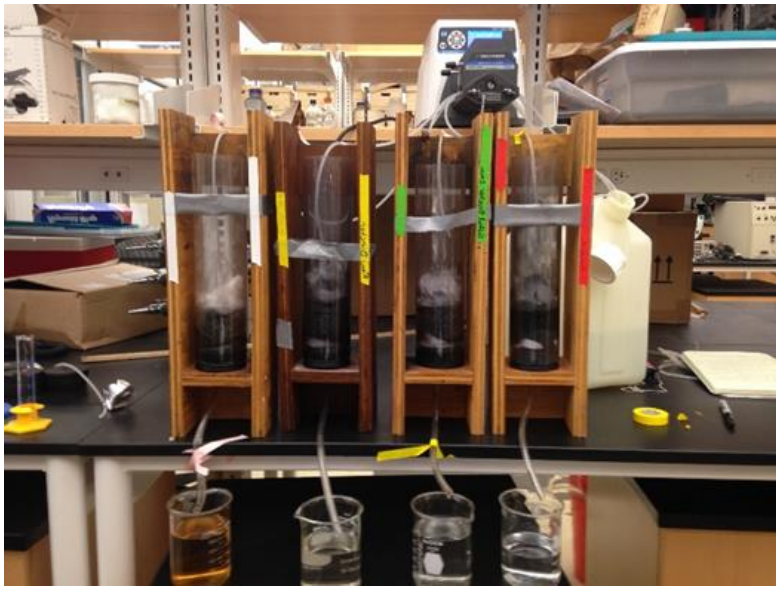


Figure 7. Column setup showing pump, four columns comparing the water retention capacity of green roof substrate (far left) vs. biochar varying in the particle size distribution. Note the brown color of the water coming from the substrate only column, indicating high dissolved humic material content.

5.2 Principal Findings

Biochar is much less dense than green roof substrate, but has a higher water holding capacity (Figure 8). We found that when we scaled the amount of water retained to the total mass of the material, biochar held about three times as much water as the vegetated roof substrate, and the three types were relatively equal to one another in their water holding capacity (Figure 8). With pure biochar mixes, when saturated 70+/-3% of the weight of the columns was water, while the remaining 30% of the weight was biochar. In contrast, saturated vegetated roof substrate was 25% water and 75% substrate by mass. We also found that blending resulted in fine particles of biochar which packed into the column more densely, resulting in a density of 0.29 g/ cm^3 when dry, and 1.01 g/ cm^3 when fully saturated. Vegetated roof substrate had a dry density of 1.00 g/ cm^3, and a wet density of 1.33 g/ cm^3. Raw and sifted biochar were lighter still, with densities of 0.14-0.18 g/cm^3, and wet density of 0.40-0.69 g/ cm^3. As a consequences of these differences, the overall mass of the 192 cm^3 (5 cm depth) columns varied substantially among the treatments. This has implications for construction of full-scale green roofs, which ideally have a high moisture holding capacity but are relatively lightweight. We chose to continue using raw biochar for future plot-scale experiments, since the pretreatment of the biochar (sifting, blending) was labor intensive and did not substantially affect the moisture-holding capacity per unit mass.

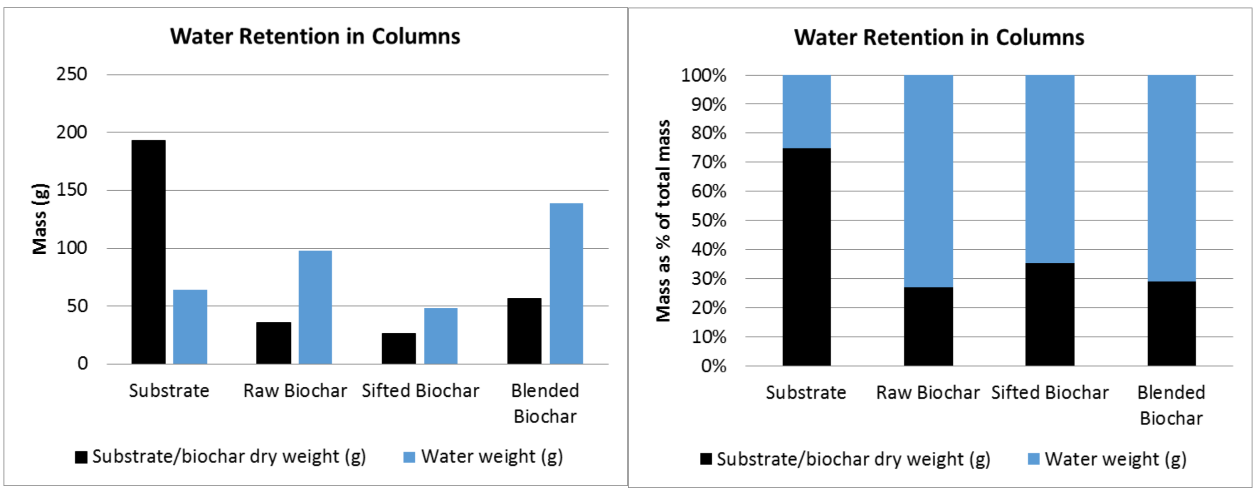


Figure 8. Substrate/biochar mass and water retention in 192 cm^3 substrate or biochar (raw, sifted, blended) columns. (Left panel) Mass in grams; (Right panel) Mass as a % of total. Although blending and sifting affected biochar density, the water holding capacity as a proportion of the total weight was essentially identical for all of the biochar columns, which all held much more water per unit weight than did the substrate alone.

6. Effect of biochar-amended substrate on hydrodynamics in plots (Spring 2015)

6.1 Methodology

We carried out a pilot study using small-scale green roof test plots (Figure 9). The purpose of the pilot study was to develop a method for continuous measurement of evapotranspiration on green roof test plots, as well as determine the differences among evaporation rates among three plots with differing proportions of biochar incorporated into the substrate. Three small (30.5 cm x 61 cm) green roof test plots were constructed. Each plot was comprised of two Eco-Roof green roof trays, with one placed on top of the other. The substrate was held by the top tray and the bottom was empty and lined with 6 millimeter UV resistant polyethylene greenhouse sheeting (Greenhouse Mega Store). This bottom tray served the purpose of making the setup leak resistant and included a spigot with plastic tubing which allowed for easy collection of runoff. The outflow was collected in 10 liter HDPE carboys. The top tray was lined with a single layer of geotextile and held the Tremco extensive green roof growing media being used as a substrate mix and two plots also included biochar from Bluegrass Biochar, identical to that used in the experiments described in Sections 4 and 5 above. The substrate mix in the top tray reached a uniform depth of 8cm throughout the tray. Once two trays were fitted together and lined with plastic and textile they were positioned at a 4% slope atop of an Adam CPW Plus-35 scale, linked to the Adam DU program on a PC, which logged masses for all plots at a 10 minute interval. This made it possible for us to view water loss due to evaporation in real time.

Figure 9. Caitlin Shaw overseeing plot and scale setup for continuous recording of weight changes due to evaporation from biochar-amended vegetated roof substrate.

Once the empty plots were secured atop their individual Adam CPW Plus-35 scale with data collection using the ADAM DU program, each of the three plots was filled with a substrate mix of 0, 5 or 10% biochar by mass. All plots were treated with 4 liters of tap water using a watering can at a slow pace (approximately equivalent to a 1" rainfall). The mass of the plots when water finished dripping is the saturated weight and was used to calculate water holding capacity. We then incubated the plots undisturbed for 11 days in the lab with ambient light and temperature, during which time the change in weight was recorded every 10 minutes as an indication of evaporation rate. After 11 days we oven dried each plot at 70 degrees Celsius for 48 hours. The weight after being oven dried provided us with a dry weight. The dry weight was subtracted from the saturated weight to yield the water holding capacity of each plot.

6.2 Principal Findings

Note, given the exploratory nature of this preliminary study which was primarily devoted to methods development, we only conducted a single replication of each treatment. Therefore, no statistical difference can be inferred until further tests can be completed.

1. The plot with 10% biochar substrate held the most water (Figure 10), retaining 3.29 liters (22.1 % v/v) despite having the lowest mass The 5% biochar plot retained 3.02 liters (20.3 % v/v) and the control (non-amended) substrate retained the least with 2.89 liters (19.4% v/v).

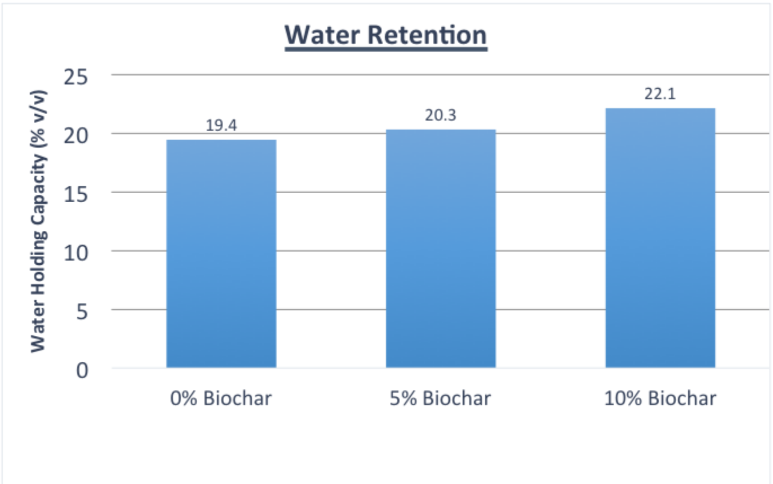


Figure 10. Water holding capacity as a percentage of total volume, for three vegetated roof plots with substrate and amended with either 0%, 5%, or 10% biochar by mass. All three plots were the same volume; the high biochar plots were lighter due to the low density of biochar.

2. The plot with 10% biochar substrate had the fastest average rate of evaporation, losing 7.20 ml/hour on average over 11 days (Figure 11). The non-amended substrate and the plot with 5% are almost even averaging a loss of 6.73 ml/hour and 6.74 ml/hour respectively.

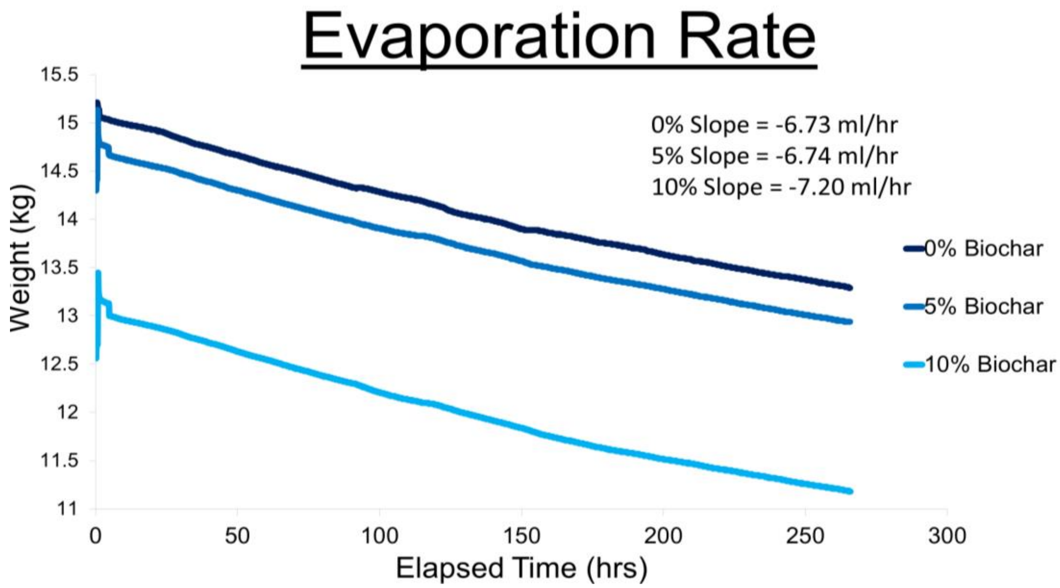


Figure 11. Weight change of each green roof plot (vegetated roof substrate alone, 5% biochar mix, 10% biochar mix) during 11 day incubation period. Evaporation rates were similar for all three treatments.

Our results demonstrate that biochar positively impacts the water holding capacity and that evaporation rates were not strongly affected by the use of biochar. Note, the experimental plots were tested indoors in climate controlled (air conditioning/heat) were the average temperature was approximately 20 degrees Celsius. These conditions are much different than the normal outdoor conditions a green roof would typically experience. Variability of the evaporation rate will occur as a function of precipitation input, wind speed, wind direction, sunlight exposure, ambient relative humidity and temperature for outdoor applications.

Biochar increased the effectiveness of the green roof substrate by increasing the substrate's water retention capacity, but no significant impact on the rate of evaporation or the length of time the substrate can retain water could be inferred. All three plots (with identical volume) lost water to evaporation at very similar rates and retained water for approximately the same length of time. In another study, non-vegetated biochar amended plots retained 2-7% more water than the control plots, but the volume of water retention actually decreased by 1-3% when the plots include vegetation (Beck et al., 2011). Our study saw an increase in the volume of water retained, consistently, upon addition of biochar.

The main purpose of this study was development of a method for continuous measurement and recording of changes in plot weight, to get at evaporation and ultimately evapotranspiration rates. This succeeded and a working method was developed, but given the exploratory nature of this preliminary study, we conducted just a single replication of each treatment. Therefore, no statistical difference can be inferred until further tests can be completed. The scales are fieldwork scales and were covered in water resistant contact paper for our study. This will make it possible for the setup to be taken outside for ambient environmental exposure during subsequent experimental trials. In order to have more certainty and to better evaluate biochar's ability to retain water more replications would be required.

Future studies will examine the effects of biochar on plant vitality, particularly if biochar has an effect on the transpiration rate of plants by increasing water availability. Further plot studies must be conducted in order to continue to evaluate different combinations of substrates that could enhance green roof effectiveness and performance. We predict, due to biochar's ability to retain water, the plots with increasing biochar will produce the densest vegetation because plants will not need to compete for water , and possibly nutrients, as much as in other plots. Based upon the increase biochar can provide to water retention, designers of vegetative roofs should give serious consideration to the amending substrate with biochar. Its ability to increase the water retention of substrate would increase the effectiveness of the roof and may reduce the frequency of when the vegetation experiences water stress.

7. Isotherm Sorption Experiment (2015-2016)

7.1 Methodology

We carried out a set of batch experiments in the lab to generate data for isotherms to describe sorption by green roof substrate, with and without biochar amendment. A typical Freundlich isotherm is an empirical relation between the concentration of nutrient adsorbed by an adsorbent, and the equilibrium concentration of the nutrient in solution. To find the average time to reach equilibrium concentrations of NH4-N, NO3-N & PO4-P, a preliminary batch study was conducted (See Section 4 above). From that study, the average equilibrium time for all the nutrients were ca. 24hr or less with few exceptions. Based on these kinetic studies and a search of the literature for similar experiments, we chose a 24-hour time interval for our isotherm experiments. This time interval was selected to achieve a balance between reaching near-equilibrium conditions with respect to sorption/desorption, vs. allowing minimal biological (microbial) impacts on nutrient concentrations, which become more important as the experiments are run for longer.

To evaluate the potential for biochar, growing medium and biochar-growing medium mixture to remove nutrients from solutions of different concentrations, batch experiments were set up in 50mL centrifuge tubes. Each tube received either 1g of growing medium ("0% biochar"); 1g of biochar ("100% biochar"); 0.95g of growing medium & 0.05g of biochar ("5% biochar"); or 0.86g of growing medium & 0.14g of biochar ("14% biochar"). Then 40ml of various concentration solutions were added: 0ppm, 1ppm, 2ppm, 5ppm, 10ppm, 15ppm, 20ppm, 30ppm, 50ppm, 100ppm of NH_4-N, NO_3-N & PO_4-P. To prepare the solutions, 0ppm solution with no nutrients and 100ppm solution with NH_4NO_3, KH_2PO_4, K_2HPO_4 salts were made and serial dilutions were performed by mixing the 0ppm & 100ppm solutions in the appropriate ratio. Before they were diluted, pH and conductivity were adjusted so that pH and conductivity value approximate the pH and conductivity value of effluent from a vegetated roof. Commonly the pH value of green roof effluent is ca. 7 and conductivity is in the range 150-300µS/cm for green roof runoff originating from rainwater (e.g., Buffam et al. 2016). To adjust the pH and maintain relatively low specific conductivity, $NaHCO_3$ and HCl were added in appropriate amounts calculated from chemical equilibrium equations. Then, all of the centrifuge tubes were placed horizontally in a fully darkened shaker table (to control for any photocatalytic activity) for 24 hours at 100 rpm at 25°C.

Following the incubation, samples were syringe filtered through a 0.45µm Millipore HA filter (Figure 12) and then immediately frozen for further analysis. The tubes with 100% biochar were pre-filtered using a paper filter and then syringe filtered through a 0.45µm Millipore HA filter. Nutrient concentrations of the effluent were determined using a spectrophotometer as described in Section 3.1.2. The adsorbed amount of nutrient (q) in adsorbent has been calculated from the difference of initial and final equilibrium concentration.

Figure 12. Environmental Engineering MS Student Nabila Farah Tasneem filtering batch isotherm samples after incubation, in preparation for nutrient analysis.

For each treatment and nutrient, the adsorbed amount vs equilibrium concentration was then plotted in logarithmic scale and fitted with a best fit line corresponding to the Freudlich isotherm analysis (Eq. 1):

$$q = KC_e^{\frac{1}{n}} \qquad \text{(Eq. 1)}$$

where q = the mass of species absorbed/mass of adsorbent; and C_e = the equilibrium concentration of adsorbable species in solution. This equation gives us the value of Freundlich isotherm parameters K and 1/n which were then used in sorption modeling (Section 8).

7.2 Principal Findings

As expected from the column experiments, different nutrients behaved differently in terms of adsorption. NH4-N showed considerable absorption at a range of initial concentrations, PO4-P showed little interaction with biochar or substrate at any concentration, while NO3-N had intermediate behavior: net adsorption up to 5ppm but little effect at higher concentrations. The amount of adsorption increased with the initial concentration of nutrient as expected, but also varied by treatment (Fig. 13). The behavior of 5% and 14% mixture was generally intermediate between the pure substrate and pure biochar treatments.

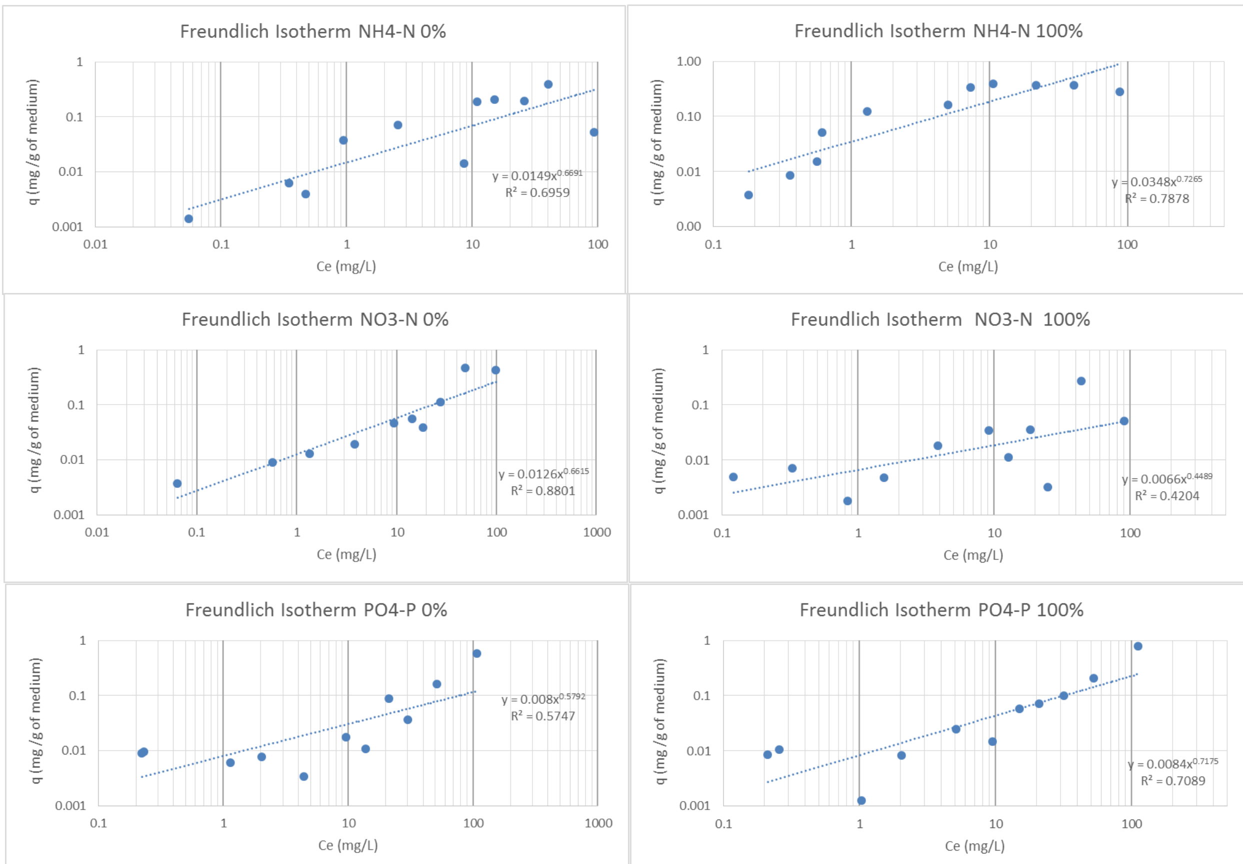


Figure 13. Example isotherm plots for calculating Freundlich isotherms for sorption/desorption of ammonium (top), nitrate (middle) and phosphate (bottom). The pure substrate (left column) and pure biochar (right column) results are shown; the mixtures of 5% and 14% biochar showed results generally intermediate of the two pure substances. The linear fit to the data gives values of the Freundlich isotherm parameters K and 1/n for use in sorption modeling.

8. Sorption Modeling (2015-2016)

8.1 Methodology

A mathematical model was developed in MatLAB to represent the adsorption behavior of NH4-N, NO3-N, and PO4-P respectively, in the presence of extensive green roof growing medium alone, as well as growing roof medium amended with varying concentrations of biochar. For modeling sorption, we chose the Homogenous Surface Diffusion Model (HSDM) based on Fick's second law of diffusion. This is the most general form of mathematical model for representing a fixed bed column with single adsorbent. The assumptions of this model are: (1) a plug flow system; (2) the column is saturated with constant hydraulic loading and no backwashing; (3) radial concentration gradient is ignored; (4) the composite adsorbent mix is homogeneous; (5) no solute interactions during the diffusion process. The model requires the estimation of four parameters: K, 1/n, D_s and K_f, and is fit using data from the fixed-bed column studies (Section 3). The Freundlich isotherm parameters K and 1/n were determined from the batch isotherm experiments as described in Section 7, while the remaining two parameters (D_s and K_f) are determined with a nonlinear least square fitting function using the Levenberg Marquardt Algorithm, appropriate when there are multiple parameters (Traegner and Suidan, 1989). The process is based on using the difference between solution of the model equations and the experimental data to continuously improve on an initial guess of the values of unknown parameters.

8.2 Principal Findings

Initial model runs suggest that the sorption/desorption behavior of the studied nutrients can be successfully represented with the HSDM modeling approach (Figure 14). This modeling work continues as part of Nabila Tasneem's MS thesis, which will be completed in 2016.

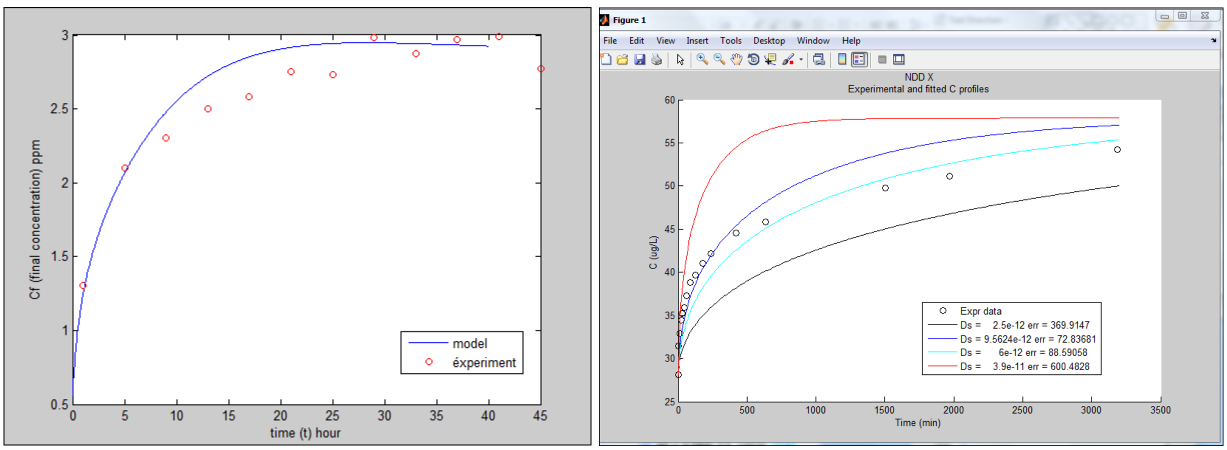


Figure 14. Example sorption model fit for the column experiment, with (Left panel) initial estimates for all four parameters (K, 1/n, Ds, Kf) to represent phosphate dynamics in the 0% biochar treatment, and (Right panel) exploration of sensitivity to variation in the Ds parameter.

9. Plot-scale evaluation of impact of biochar on water and nutrient retention

9.1 Methodology

A plot-scale field study was established in May 2015 to test the effect of biochar amended substrate in green roof plots on water and nutrient retention. Twelve experimental plots were established side by side (Figure 15). The plots contained varying quantities of biochar mixed into the substrate to a depth of 7 cm. Two replicates of each of the following treatments were assembled: vegetated plots containing substrate without biochar (control plots); vegetated plots containing substrate amended with 5% biochar by weight; and vegetated plots containing substrate amended with 10% biochar by weight. In addition, duplicate plots of the same design were constructed without vegetation (non-vegetated plots). Each plot consisted of two HDPE plastic trays with the dimensions of 60 cm by 29.2 cm (EcoRoof Inc.). These plots contained basic green roof components: a filter layer in the form of geotextile fabric (DeWitt Filter Fabric, Forestry Supplies, Jackson, MS), a drainage board in the form of corrugated plastic and a waterproof layer in the form of plastic sheeting to assist in the drainage of water from the plots. Substrate was a proprietary aggregate-based blend (Tremco Roofing Inc., Cincinnati, OH).

Biochar was obtained from Bluegrass Biochar, identical to that used in the earlier experiments described in this report.

The vegetation was established with cuttings of mixed *Sedum* species from Emory Knoll Farms, MD (Figure 15). Species included *Phedimus takesimensis, Sedum kamtchaticum, S album, S. album var. murale, S. aizoon, S. spurium 'John Creech', S. spurium 'Roseum', S. spurium 'Schorbuser Blut', S. spurium 'Fuldaglut', S. middendorfianum diffusm, S. rupestre 'Angelina', S. reflexum 'Blue Spruce', S. kam. floriferum var. 'Weihenstephaner Gold', S. hybridum 'Immergrunchen', S. sexangulare*, and *S. acre*. The vegetated plots contained an even proportion of each Sedum species at a density of 600 g m^{-2} total. Sedum species are especially successful, in terms of plant coverage and survival, in green roof installations in the American northeast and Midwest (Durham and Rowe, 2007; Butler and Orians, 2011, Starry, 2013). During plant establishment, the plots were located at the University of Cincinnati's greenhouse (Rieveschl Hall, Cincinnati, OH). After plant establishment, the plots were moved to a nearby rooftop (Rieveschl Hall, UC campus).

Figure 15. Biology MS Student Alicia Kosielski planting a mixture of *Sedum* cuttings for the plot-scale experiment. Plots were initially established in a greenhouse setting.

Flushes with known volumes of deionized water were performed periodically. Each plot was outfitted with a spigot and tygon tubing to allow for the collection of water into a collection bucket. Water retention was measured as the amount of water that was collected from each plot subtracted from the amount of water added to each plot. Effluent was collected, then filtered using 0.45 μm Millipore HA membrane filters, and pH and conductivity were measured as well as the concentrations of inorganic N and P as described in Section 3.1.2.

9.2 Significant Findings

The Sedum mixtures established successfully in all plots, with no obvious effect of biochar on plant growth or health after the first growing season (Figure 16).

Figure 16. Example overhead photos of plots, four weeks after propagation from cuttings. From left to right: 0%, 5%, and 10% biochar treatments containing a diverse assemblage of *Sedum* spp.

Below are average results from flushes after 3 and 5 months of growth (Fig. 17), representative of the results we have seen so far during the first growing season in the greenhouse. The effect of biochar showed an increase in water retention (30%) and increase of pH (+0.5 units), while the presence of plants had the effect of decreasing specific conductivity (salt concentration) in the effluent water. The greater water retention (thus lower runoff) is in agreement with the results from our column and other lab-based studies. During this initial period, biochar has not had a measurable effect on effluent conductivity or concentrations of phosphate, nitrate, or ammonium. This is an interesting result because it seems to contrast with the results from the column study and sorption batch study, where we clearly saw greater binding of ammonium by biochar than by growing medium alone. However, in the plot study all treatments have fairly low ammonium in the effluent, even those with growing medium alone; this may explain why we don't see a treatment effect due to the addition of biochar. Note, this study had a relatively small sample size (two replicates of each treatment and two flushes analyzed so far) and it may be that differences will develop over time. Graduate student Alicia Kosielski has proposed a follow-up to this study for her MS thesis, building on these results to carry out a similar study but increase the number of replicates to 6 for each treatment, and to carry out the experiment over at least a 1-year period under natural climate and precipitation conditions.

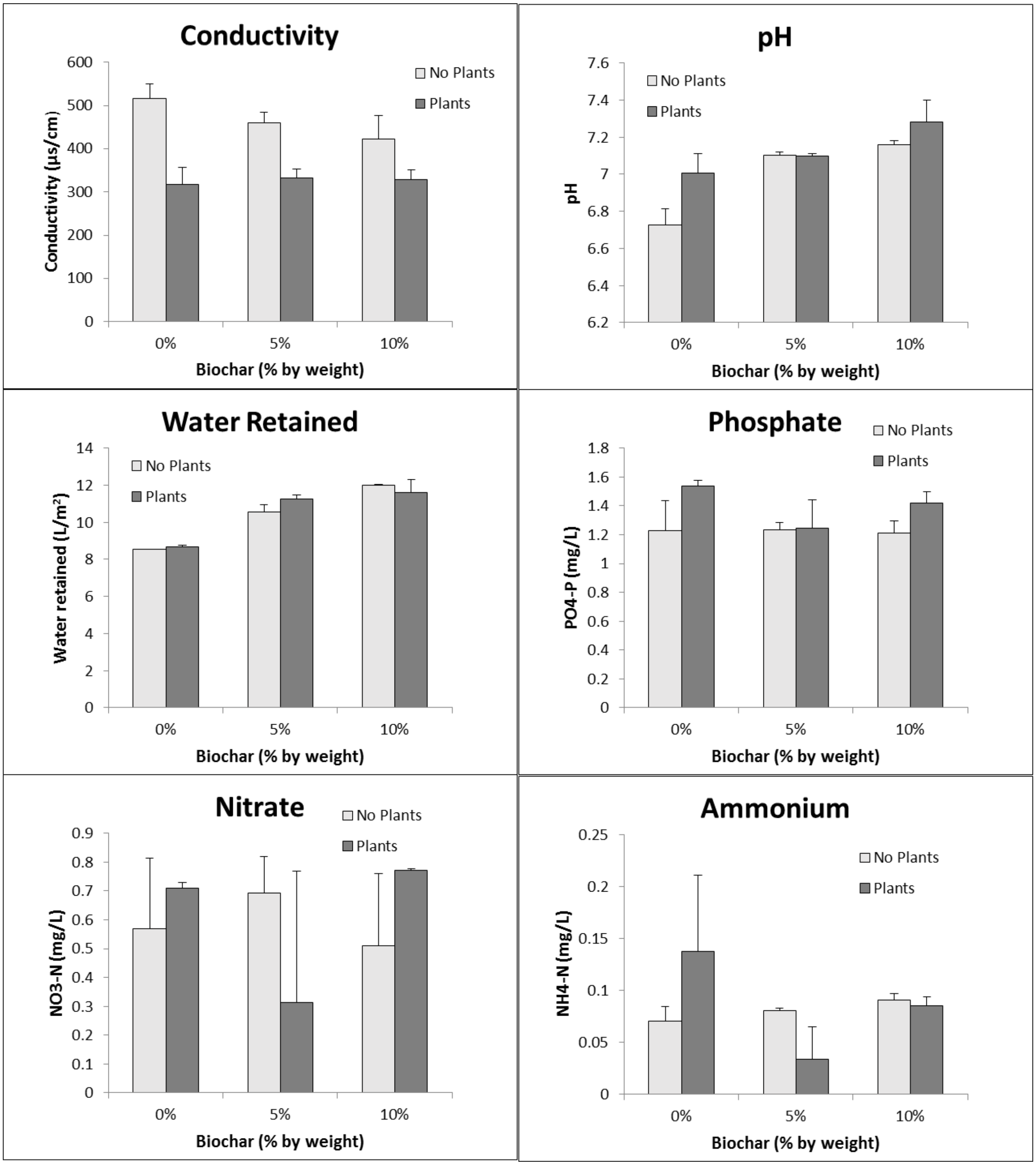


Figure 17. Effect of biochar % and plant presence/absence on runoff water quality and amount, in plot experiments.

10. Significance

The integration of biochar is a potential breakthrough in reducing water quality degradation by green roof runoff, but very little is known about the sensitivity to variation in the proportion of the biochar amendment, or the dynamics of sorption kinetics or equilibria. Our project has demonstrated that a biochar amendment has the potential to substantially decrease ammonium leaching from green roofs, by up to 75% for the high biochar (14% w/w) treatment, in the presence of high ammonium load. The high biochar treatment also doubled water holding capacity of the substrate, a finding with great significance for green roof design for stormwater runoff reduction. A qualitatively similar result was observed in field plots, with a 30% reduction in runoff volume on average, during the first growing season. This is of particular note because on a per-mass basis, biochar is no more expensive than typical commercially available green roof substrate mixes. The patterns of breakthrough curves also give insight into likely physicochemical mechanisms of nutrient binding. Specifically, the inflections in the curve suggest a dual-layer sorption mechanism for the biochar for ammonium and phosphate, with initial surficial sorption occurring within a few hours followed by a slow sorption process taking a few days, perhaps limited by diffusion into the interior of biochar particles. Follow-up work using different sorption breakthrough models and isotherms are underway, to further explore the sorption/desorption dynamics.

This study evaluates a low-cost option for improving the effluent water quality of vegetated roof technology, which is becoming increasingly more important as part of green-engineered solutions for stormwater management. The research demonstrates the water quality improvements associated with a biochar-amended green roof, but also includes a modeling component that will provide a tool for use within an integrated assessment framework both within and beyond the Ohio River Valley. As a result, the positive impact of this project will be a significant step forward in developing a more integrated infrastructure solution for storm water management by illustrating the potential impacts of biochar-amended vegetated roofs on CSO and nutrient management in urban environments.

15. References

Beaton, J.D., H.B. Peterson, and N. Bauer. (1960). "Some aspects of phosphate adsorption to charcoal." *Soil Science Society of America Proceedings*. 24:340-346.

Beck, D.A, G.R. Johnson, G.A. Spolek. (2011) "Amending greenroof soil with biochar to affect runoff water quantity and quality" *Ecological Engineering* 159:2111-2118.

Berndtsson, J.C., Bengtsson, L., and Jinno, K. (2009) "Runoff water quality from intensive and extensive vegetated roofs" *Ecological Engineering* 35: 369-380.

Berndtsson JC (2010) Green roof performance towards management of runoff water quantity and quality: A review. *Ecological Engineering*, 36, 351-360.

Bliss, D. J., Neufeld, R. D., and Ries, R. J. (2009) "Storm Water Runoff Mitigation Using a Green Roof" *Environmental Engineering Science*, 26 (2), 407-417.

Buccola, N. 2008. "A laboratory comparison of green roof runoff." A thesis submitted in partial fulfillment of the requirements for the degree of Masters of Science in Civil and Evnironmental Engineering, Portland State University, Portland, OR.

Buffam, I. and M.E. Mitchell. 2015. Nutrient cycling in green roof ecosystems. Chapter 5 in R. Sutton, ed. Green Roof Ecosystems, Springer, New York.

Buffam, I., Mitchell, M.E., and R.D. Durtsche. 2016. Environmental drivers of seasonal variation in green roof runoff water quality. Ecological Engineering, 91, 506-514, doi: 10.1016/j.ecoleng.2016.02.044.

Butler, C., Orians, M. (2011) Sedum cools soil and can improve neighboring plant performance during water deficit on a green roof. Ecological Engineering 37, 1796-1803.

Carter, T. L. and Rasmussen, T. C. (2006) "Hydrologic Behavior of Vegetated Roofs" *Journal of the American Water Resources Association*, 42 (5) 1261-1274.

Carter, T.; Laurie, F., Establishing green roof infrastructure through environmental policy instruments. *Environmental Management* **2008,** *42*, 151-164.

Deutsch, B.; Whitlow, H.; Sullivan, M.; Saveneau, A. *Re-Greening Washington, DC: A Green Roof Vision Based on Quantifying Storm Water and Air Quality Benefits*; 2005; p 15.

Doane, T.A., and W.R. Horwath. (2003) "Spectrophotometric determination of nitrate with a single reagent" *Analytical Letters* 36: 2713-2722.

Downie, A., V.C. Lima, G. Seehann, J. Donath, V.R. Monntoia, and T. Schwarz. (2007) "Retention properties of wood residues and their potential for soil amelioration." *Wood Science and Technology*. 41:169-189.

Durham, A.K. And B. Rowe. (2007) Effect of Substrate Depth on Initial Growth, Coverage, and Survival of 25 Succulent Green Roof Plant Taxa. HortScience 42(3): 588-595.

Getter, K. L., Rowe, D. B., Andersen, J. A. (2007) "Quantifying the Effect of Slope on Extensive Green Roof Stormwater Retention" *Ecological Engineering*, 31 (4), 225-231.

Gregoire, B. G. and Clausen, J. C. (2011) "Effect of a modular extensive green roof on stormwater runoff and water quality" *Ecological Engineering*, 37, 963-969.

Lajtha, K., Driscoll C.T., Jarell W.M., and Elliott E.T. (1999) "Soil phosphorus: characterization and total element analysis" Pages 115-142 in G.P. Robertson, D.C. Colemna, C.S. Bledsoe, and P. Sollins, editors. Standard Soil Methods for Long-Term Ecological Research. Oxford University Press, New York.

Lehmann, J. and S. Joseph. (2009) *Biochar for environmental management: Science and technology*. Stirling, VA: Earthscan.

Lehmann, J., J.P. da Silva Jr., M. Rondon, M.S. Carvo, J. Greenwood, T. Nehls, C. Stiener, and B. Glaser. (2002) "Slash-and-char – a feasible alternative for soil fertility management in the central Amazon?" *Proceedings of the 17th World Congress of Soil Science*. Bangkok, Thailand. Paper no 449.

Mentens, J., Raes, D., and Hermy, M. (2006) "Green Roofs as a Tool for Solving the Rainwater Runoff Problem in the Urbanized 21st Century" Landscape and Urban Planning, 77 (3), 217-226.

NYC Department of Environmental Protection. (2011) "NYC Green Infrastructure Plan: A Sustainable Strategy for Clean Waterways", 154 pp.

Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R., Doshi, H., Dunnett, N., Gaffin, S., Kohler, M., Liu, K. K. Y., and Rowe, B. (2007) "Green roofs as urban ecosystems: Ecological structures, functions, and services" *Bioscience,* 57, 823-833.

Piccolo, A., G. Pietramellara, and J.S.C Mbagwu. (1996) "Effects of coal-derived humic substances on water retention and structural stability of Mediterranean soils." *Soil Use and Management*. 12:209-213.

Ringuet, S., Sassano, L., & Johnson, Z.I. (2010) A suite of microplate reader-based colorimetric methods to quantify ammonium, nitrate, orthophosphate, and silicate concentrations for aquatic nutrient monitoring. *Journal of Environmental Monitoring*, **13**, 370-376.

Starry, O., J.D. Lea-Cox, J. Kim, and M.W. van Iersel. (2014) Photosynthesis and water use by two Sedum species in green roof substrate. Environmental and Experimental Botany 107, 105-112.

Traegner, U.K. and M. T. Suidan. (1989) Parameter evaluation for carbon adsorption. *Journal of Environmental Engineering*, **115**:109-128.

Van Woert, N. D., Rowe, D. B., Andersen, J. A., Rugh, C. L., Fernandez, R. T., and Xiao, L. (2005) "Green Roof Stormwater Retention: Effects of Roof Surface, Slope and Media Depth" *Journal of Environmental Quality*, 34 (3), 1036-1044.

Weatherburn, M.W. (1967) "Phenol-hypochlorite reaction for determination of ammonia" *Analytical Chemistry* 39: 971-974.

Yao; Gao; Inyang; Zimmerman; Cao; Yang. 2011. "Removal of phosphates from aqueous solutions by biochar derived from anerobically digested sugar beet tailings." *Journal of Hazardous Materials.* 190: 501-507.

SEPARATION OF PHOSPHORUS- AND NITROGEN-NUTRIENTS FROM AGRICULTURAL DEGRADED WATERS USING PERVIOUS FILTER MATERIAL DEVELOPED FROM INDUSTRIAL BY-PRODUCTS

Basic Information

Title:	SEPARATION OF PHOSPHORUS- AND NITROGEN-NUTRIENTS FROM AGRICULTURAL DEGRADED WATERS USING PERVIOUS FILTER MATERIAL DEVELOPED FROM INDUSTRIAL BY-PRODUCTS
Project Number:	2013OH436O
Start Date:	9/1/2015
End Date:	2/28/2017
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Congressional District:	OH-12
Research Category:	Engineering
Focus Category:	Treatment, Agriculture, Water Quality
Descriptors:	Industrial by-rpoducts, filter material, nutrients
Principal Investigators:	Linda Kay Weavers, ChinMin Cheng

Publications

There are no publications.

SEPARATION OF PHOSPHORUS- AND NITROGEN-NUTRIENTS FROM AGRICULTURALLY DEGRADED WATERS USING PERVIOUS FILTER MATERIAL DEVELOPED FROM INDUSTRIAL BY-PRODUCTS

Progress Report

Submitted to:

Ohio Water Resources Center

Submitted by:

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ABSTRACT

End-of-tail filtration has been suggested as a more aggressive and effective approach to reduce losses of nutrients from crop lands compared to current best management practices (BMPs) focusing on source reduction and minimizing transportation. A number of industrial by-products, e.g., coal combustion by-products and bauxite leaching residual, have been proven chemically effective in trapping P- and/or N-nutrients, and therefore, are potential low-cost nutrient sorbents for the end-of-tail filtration approach. However, the application of these industrial by-products as the filtration media is limited due to unfavorable hydraulic properties, as well as unknown associated environmental impacts. In this proposed study, pervious filter materials owning both reactivity to nutrients and adequate hydraulic properties are developed using fly ash, stabilized FGD materials, and bauxite leaching residual as the feedstock. By modifying the composition of these industrial by-products, the pervious materials are expected to have selective nutrient-sequestrating capabilities, which can be used to separate and recycle phosphorus- and nitrogen-nutrients from agricultural drainage waters (ADWs). This study is carried out in three tasks to (1) investigate the adsorption efficiency and service lifetime of selected pervious materials with synthetic ADW; (2) evaluate the physical and chemical integrity of the pervious materials before and after service; and (3) study the interactions between the prepared filter materials and emerging pollutants commonly found in ADW (e.g., estrone). The goal of this study is to demonstrate the feasibility of applying a low-cost and environmentally-sustainable approach to ADW handling and treatment. This alternative to current BMPs is able to convert agricultural and industrial wastes to value-added products containing concentrated and specific nutrients. Currently, the project is still on going. Results obtained from this study will be used to develop a competitive proposal for external funding.

1. **Introduction**

Eutrophication of water bodies, a result of release of excessive phosphorous (P) and nitrogen (N) from soil to drainages[1], has been an increasing environmental issue in the US, especially in the Midwest, northeast, and Gulf coast area where the watersheds of major freshwater bodies involve rapid growth and intensification of crop and livestock farming[2]. Not only eutrophication posts unpleasant aesthetic characteristics to water bodies, accumulation of toxic, volatile chemicals produced by algae can cause neurological damage in people and animals being exposed to them. Consequently, eutrophication of water resources results in losses of biodiversity, as well as their amenities and services[3]. For example, the recent outbreaks of Cyanobacteria, or blue-green algae, in the Grand Lake at St. Mary's area in Ohio has led to state officials to issue water contact and fish consumption advisories.

The major cause of many eutrophication incidents can be directly correlated to fertilizer application[4]. To prevent accumulation of nutrients in surface waters, reduction of nutrients present in the agricultural degraded waters (ADW, i.e., livestock wastewater overflow, subsurface drainages, and surface runoffs from cropland) is perceived as necessary approach[5]. Although many best management practices (BMPs) focusing on source reduction and minimizing transportation have been implemented to reduce losses of nutrients from crop lands, these approaches have shown no control on dissolved phosphorus losses[6,7], which is the most readily available form of phosphorus to aquatic organisms[8]. Instead, end-of-tail filtration has been suggested as a more aggressive and effective approach[6]. However, the application is limited. Ideal filter materials, i.e., material with both favorable nutrient-sequestrating capability and hydraulic property, have yet been identified[9].

In this study, low-cost pervious sorption materials prepared from a self-geopolymerization process using agricultural wastes and industrial by-products are tested for their potential as an alternative to current BMPs. The self-geopolymerization process enchains agricultural wastes with chemically-effective, nutrient-sorbing industrial by-products (e.g., coal ash, flue gas desulfurization materials, and bauxite residual) and forms pervious materials. By modifying the composition, the pervious materials are expected to have selective sorption capabilities to nitrogen (N-) and phosphorus (P-) nutrients with adjustable hydraulic properties, which can be used to separate and recycle nutrients from ADWs.

2. **Objectives**

In this study, a geopolymerization procedure is developed to convert coal combustion by-products (i.e., fly ash and flue gas desulfurization (FGD) material) and alkaline bauxite leaching residual (bauxite red mud) to pervious filter materials. The materials are tested in a bench-scale setting for their effectiveness and capacity on removing nutrients from simulated agricultural drainage waters. The specific objectives of this proposed project are to:

(1) Assess the performance of the industrial by-product-derived pervious filter materials with respect to their nutrient removal efficiencies, service lifetime, and hydraulic properties;
(2) Evaluate the chemical and physical integrity of the materials; and
(3) Study the interactions between the prepared filter materials and other pollutants contained in ADWs (i.e., estrogens).

3. Materials and Method

The work of this proposed study is divided into three tasks. In summary, the first task focuses on preparing and characterizing the pervious filter materials. At least three sets of P-type (i.e., materials selectively adsorb P-nutrients) and N-type (i.e., materials adsorbed nitrate and/or other N-nutrients) are prepared. In the second task, a series of column experiments are setup to (1) evaluate the adsorption efficiency and capacity of the selected pervious materials with a simulated ADW and (2) study the interactions between estrogens and filtration materials. In addition, the physical and chemical integrities of the pervious filter material during and after service are evaluated. The release of metals and metaloids (e.g., mercury, arsenic, selenium, thallium, and boron), as well as sulfate, from the filter materials during filtration are monitored. In addition, surface characterization techniques, such as X-ray diffraction (XRD) and scanning electron microscopy (SEM), are applied to investigate the transformations of mineral composition and surface morphology before and after the filtration materials are exhausted.

Pervious Filter Material Preparation and Characterization

Coal combustion by-products (i.e., fly ash and stabilized FGD materials) and bauxite leaching residue (i.e., red mud) are used in the preparation of the nutrient-selective pervious filtration materials (Figure 1). Two different types of pervious filtration materials (i.e., P- and N-types) are prepared using a method modified from Cheng et al.[10] and Jin[11]. Class F fly ash and sulfite-rich stabilized FGD material provided by coal combustion power plants located in eastern Ohio are used to prepare the phosphorous-capture (P-type) filtration materials. Quick lime

(Carmeus USA, Pittsburg, PA), CaO, is added to provide required alkalinity. The nitrogen-capture materials are prepared from red mud, fly ash, and stabilized FGD material. No quick lime is used in the preparation of N-type filter materials. The bauxite red mud provided by a bauxite processing plant located at southeast Texas is oven-dried before use. In one batch, manganese oxide (MnO_2) is also added in the preparation of N-type material. Woodchip is used in the preparation of both N and P-type filter mixtures to modify the hydraulic properties. The prepared mixtures are then cured in a humidity chamber.

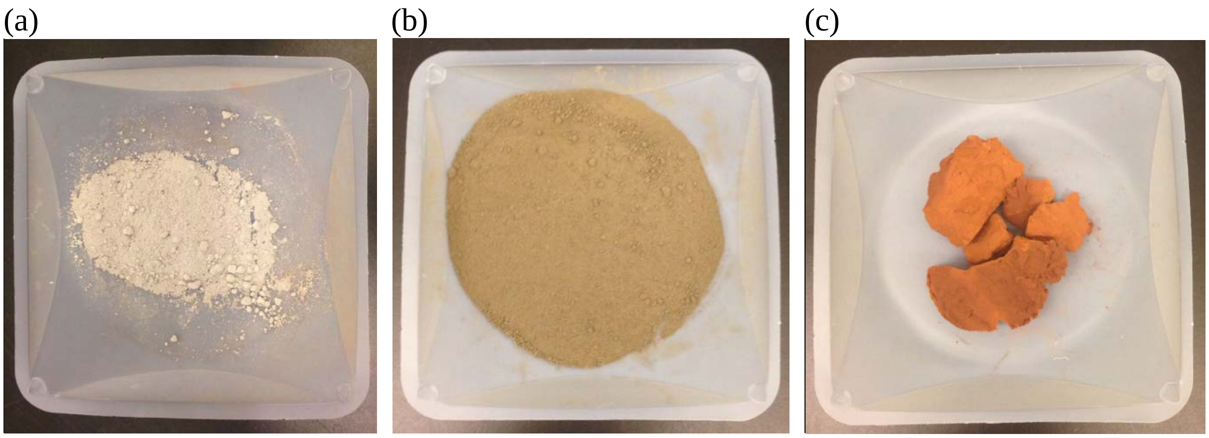

Figure 1. (a) Stabilized FGD material, (b) fly ash, and (c) bauxite red mud used in the preparation of pervious filtration materials.

The cured filter materials are tested for their chemical (i.e., elemental and mineral compositions), physical (density and surface morphology), and engineering (i.e., permeability (k) and/or hydroconductivity (K)) properties as per standard testing protocols. Details on the chemical and physical characterizations of the filter materials are described in the "*Physical and Chemical properties Integrity Evaluation*" section.

Bench-Scale Column Test

A series of column tests are carried out to measure the adsorption capacity and efficiency of prepared pervious materials for P- and N-nutrients with a simulated ADW. In addition to the prepared filter materials, two reference columns, packed separately with granular activated carbon (GAC) and top soil from the OSU's Waterman Farm Complex, are also included in the column study. A control column, i.e., without packing medium, is included to evaluate the adsorption of nutrients and compounds on the experimental apparatus.

The setup of the column test is illustrated in Figure 2. The ADW used in the column test is synthesized based on formula listed in Table 1. In addition to the constituents listed in the table, one estrogen, e.g., estrone (E1) or 17α-Estradiol (17α-E2), commonly found in dairy wastewater[12] is added in selected experimental batches. A peristaltic pump delivers the synthetic ADW to the inlet of a series of two vertically-oriented columns at a constant feed rate (Figure 2). The ADW sequentially passes through the column containing P-type filter material (P-type column) and then the N-Type column. For a given set of filter materials, the column test is carried out under a saturation condition demonstrated in Figure 2.

Table 1. Composition of synthetic dairy wastewater used in this study

Component	Amount (mg/L)
Urea	115.7
NH_4Cl	250.0
$Na_2PO_4 \cdot 12H_2O$	385.7
$KHCO_3$	257.1
$NaHCO_3$	668.6
$MgSO_4 \cdot 7H_2O$	257.1
$FeSO_4 \cdot 7H_2O$	10.3
$MnSO_4 \cdot H_2O$	10.3
$CaCl_2 \cdot 6H_2O$	15.4

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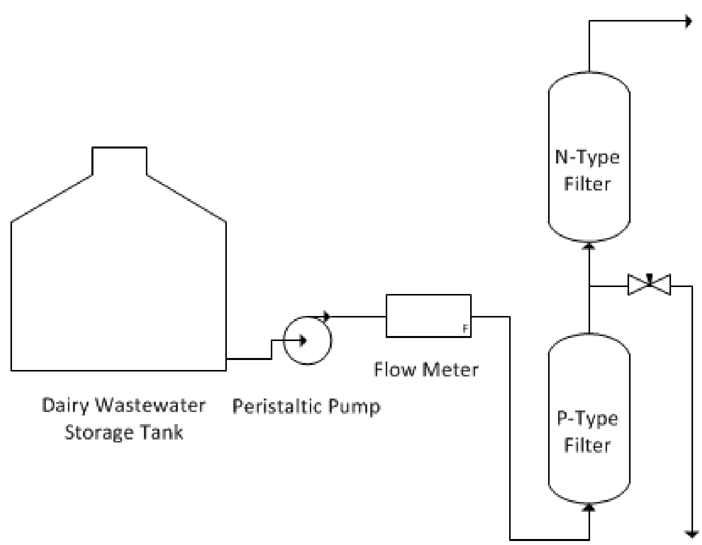


Figure 2. Setup of bench-scale column test

Effluent samples are collected periodically from the outlets of P-type and N-type columns for a list of chemical analyses shown in Table 2. After collection, sample is immediately separated into four sub-samples. The first sub-sample is for pH, conductivity, and redox potential measurements. In the selected batches when estrogen is included in the synthetic ADW, an aliquot of the first subsample is filtered with 1.2μm glass fiber and concentrated by solid-phase extraction for estrogen analysis. Any compounds remained on the sample collection bottle or filter is desorbed by rinsing the bottle and filter with methanol. The concentrated sample is analyzed using a high-performance reverse-phase liquid chromatography tandem electrospray ionization mass spectrometry (HPLC/MS/MS). Deuterated internal standards is added to the samples to correct the interferences caused by the matrix of the sample.

The second sub-sample is filtered and analyzed for alkalinity, total dissolved solids, Cl-, SO_4^{-2}, PO_4^{-3}, total Kjeldahl nitrogen, ammonia, and NO_3^-. The third sub-sample is preserved with 5% HNO_3 and analyzed for "total" elements in the solution. The final sub-sample is filtered through a 0.45-μm syringe filter and preserved with 5% HNO_3 before being analyzed for "dissolved" elements.

Table 2. List of monitoring parameters and respective analytical methods for aqueous samples

Subsample	Parameter	Detection Methods	Instruments	Locations
Subsample I	Conductivity	AWWA Sec. 2510	Thermo Orion 1234	*in-situ*
	pH		Thermo Orion 1234	*in-situ*
	Redox Potential		Thermo Orion 1234	*in-situ*
	Estrogen[c]	HPLC/MS/MS	Micromass Q-TOF II	CCIC[b]
Subsample II	Alkalinity	AWWA Sec. 2310	-	CEGE EER Lab/ OARDC STAR Lab
	Total dissolved solid	AWWA Sec. 2540	-	
	Chloride (Cl)	AWWA Sec. 4110C	Dionex 2100	
	Sulfate (SO_4^{-2})	AWWA Sec. 4110C	Dionex 2100	
	Phosphate(PO_4^{-3})	AWWA Sec. 4110C	Dionex 2100	
	Nitrate (NO_3^-)	AWWA Sec. 4110C	Dionex 2100	
	Ammonia (NH_4^+)	AWWA Sec. 4110C	Dionex 2100	
	Total Kjeldahl Method	AWWA Sec. 4500 N_{org}	-	
Subsample III/ Subsample IV	Mercury (Hg)	CVAFS	Varian CVAAs,	
	Selected Elements[a]	AWWA Sec. 3120B	Varian VISTA-AX	
	Arsenic (As)/ Thallium(Tl)	AWWA Sec. 3120B	Varian GFAAs, Varian 880Z	
	Selenium (Se)	AWWA Sec. 3120B	Varian GFAAs, Varian 880Z	

[a] Aluminum (Al), arsenic (As), barium (Ba), beryllium (Be), boron (B), cadmium (Cd), copper (Cu), chromium (Cr), iron (Fe), lead (Pb), magnesium (Mg), manganese (Mn), nickel (Ni), phosphorous (P), sodium (Na), silver (Ag), zinc (Zn).
[b] Campus Chemical Instrument Center at The Ohio State University
[c] On selected experimental batches

Chemical and Physical Integrity Evaluations

The exhausted filter materials are preserved using liquid nitrogen and freeze-dried before being analyzed for the mineral and chemical compositions, surface morphology, and forms of adsorbed phosphorus by the methods listed in Table 35. The mineral compositions and morphology of the selected N- and P- type filters materials before and after service are characterized using X-ray diffraction (XRD) and scanning electronic microscopy (SEM), respectively. A Bruker D8 Advance X-ray diffractometer or equivalent is used to identify the mineral composition. The mineral patterns in the diffractograms are matched using the DIFFRACplus EVA software with ICDD Power Diffraction File (PDF2+) database. The complete elemental composition analysis is measured with the assistance of the digestion procedure described in EPA method 3052. A reference coal fly ash, 1633b, provided by the National Institute of Standards and Technology (NIST), is included for analytical quality control. A list of the analyses performed on the materials can be seen in Table 4.

The release potential of trace elements from filter materials before and after service will also be characterized. Standard protocols, i.e., EPA Standard Method 1311, Toxicity Leaching Characteristic Procedure (TCLP), the EPA Standard Method 1312, Synthetic Precipitation Leaching Procedure (SPLP), are used.

Table 3. Physical, mineral, and chemical analyses for selected pervious filter materials

	Method	Instrument	Location
Permeability	ASTM D4525-08		CEGE Soil Lab
Hydraulic Conductivity	ASTM D7100-06		
Morphology	Scanning Electron microscopy	Hitachi S-3000 SEM	OSU Nanotech West Lab
Mineral Composition	X-ray Diffraction	Bruker D8 Advance X-ray diffractormeter	SENR Soil Lab[c]
Selected Elements[a]	ASTM D-6357	Milestone Microwave Digestor/ Varian VISTA-AX	CEGE EER Lab[b]
Mercury	ASTM D-6414	Varian CVAAs, Varian 880Z	CEGE EER Lab
Selenium	ASTM D-4606	Varian CVAAs, Varian 880Z	CEGE EER Lab
Arsenic, Thallium	ASTM D-3683	Varian GFAAs, Varian 880Z	CEGE EER Lab

[a] aluminum (Al), barium (Ba), beryllium (Be), boron (B), cadmium (Cd), chromium (Cr), lead (Pb), magnesium (Mg), manganese (Mn), nickel (Ni), phosphorous (P), sodium (Na), sulfur (S), and zinc (Zn).
[b] Environmental Engineering Research Laboratory at Department of Civil, Environmental, and Geodetic Engineering of The Ohio State University
[c] Soil Lab at School of Environment and Natural Resources of The Ohio State University

4. Current Progress and Tasks to be completed

Characterizations of Industrial By-products

The chemical compositions of fly ash, stabilized FGD material, and bauxite red mud are first characterized and the results are summarized in Table 4. As shown in the table, calcium (Ca) and sulfur (S) are the two most abundant elements in the stabilized FGD material, which is associated with the presence of hannebachite ($CaSO_3 \cdot 0.5H_2O$), portlandite ($Ca(OH)_2$), and enttringite ($Ca_6Al_2(SO_4)_3(OH)_{12} \cdot 26H_2O$) in the material. The X-ray diffractogram and mineral composition of stabilized FGD material can be seen in Figure 3. Iron (Fe), aluminum (Al), sulfur (S), and silicon (Si) are the major elements in fly ash. Based on XRD analysis, the fly ash used in this study is comprised of amorphous glass, aluminum silicates (e.g., mullite), and iron oxides (hematitem, magnetite, and maghemite). Bauxite red mud is consisted of Al, Fe, and Ca. The X-ray diffractograms of fly ash and red mud are not shown.

By properly coalescing fly ash, stabilized FGD material, and red mud under high alkaline environment, fly ash acts as an inorganic polymer binder to enchain active ingredients through a geopolymerization process. After being alkali-activated, the Si-O-Si or Al-O-Si bonds in fly ash and stabilized FGD material are disassociated and subsequently form network-like crystalline and/or amorphous alkaline aluminosilicates with structural framework similar to zeolite[13]. In a previous project, it has been demonstrated that a geotextile material derived from the geopolymerization process with a mixture of fly ash and stabilized FGD material, has effective phosphorus sorption capability by forming Ca- and Fe-precipitates[10,14,15]. However, the fly ash/stabilized FGD material mixture did not show observable effect on nitrate mitigation[10].

The addition of bauxite red mud is to enhance the nitrogen-nutrients adsorption capability of the fly ash/FGD mixture. Bauxite red mud contains minerals, e.g., iron (III) (hydr)oxides and hydrous aluminum oxides, that have high affinities for nitrate[16]. As a result, the material has been shown to be an effective nutrient sorbent[17]. Cengeloglu et al[17] used original and acid-treated bauxite red mud to remove nitrate from aqueous solution and reported 70% and over 90% of removal, respectively. They found the alkaline property of bauxite red mud hindered the adsorption performance.

In this study, bauxite red mud is used as the sole alkalinity source in the geopolymerization process, which might promote the nitrate adsorption capacity. During geopolymerization, the OH- ions from bauxite red mud is consumed (eq. 1) and redistribute the electron density around the silicon atom in fly ash, which weaken the strength of Si-O-Si bond[18] and progress the polymerization process. The reaction neutralizes the negative surface charge of red mud particles, and therefore, might promote the nitrate sorption.

$$\equiv Si - O - Si \equiv + OH^{-} \Rightarrow \equiv Si - OH + ^{-}O - Si \equiv \quad (1)$$

Preparation of P- and N-type pervious filtration

A series of P- and N-type pervious filtration materials have been prepared based on the formulas listed in Tables 5 and 6. Currently, the prepared materials are undergoing a 21-day curing process. The images of two selected prepared materials can be seen in Figure 4. The hydraulic property of the filtration materials are adjusted by the addition of woodchip. Two different sizes of woodchip, i.e., <2.3mm and 2.3-3.6mm, are used. The addition of woodchip creates larger capillary routes for water to pass through. During the geopolymerization process, active ingredients are coated on the surface of woodchip, which allows the nutrients in ADW to react with the active ingredients while passing through the void space.

Table 4. Chemical compositions of fly ash, stabilized FGD material and bauxite red mud used in this study

		Fly Ash	Stabilized FGD material	Red Mud
Phosphorus	P	531	177	1054
Potassium	K	2986	1307	310
Calcium	Ca	9836	172906	33055
Magnesium	Mg	1528	10026	227
sulfur	S	11827	85746	2867
Aluminum	Al	27050	9705	62817
Boron	B	531	313	<3
Copper	Cu	42	<0.4	<0.8
Iron	Fe	59824	18855	240960
Manganese	Mn	85	73	139
Molybdenum	Mo	22	<13	<0.5
Sodium	Na	18851	5296	32412
Zinc	Zn	109	40	22
Arsenic	As	143	36	28
Barium	Ba	177	137	61
Beryllium	Be	<0.18	<0.11	<0.18
Cadmium	Cd	2	6	5
Cobalt	Co	23	4	15
Chromium	Cr	74	25	1397
Lithium	Li	167	106	55
Nickel	Ni	48	7	6
Lead	Pb	28	8	46
Antimony	Sb	<1.5	17	<1.5
Selenium	Se	20	18	1
Silicon	Si	4771	1481	184
Strontium	Sr	229	212	117
Thallium	Tl	129	38	871
Vanadium	V	2	<1.1	<0.6
Mercury	Hg	NA	0.318	NA

NA:Not Available
Unit: mg/kg

Table 5. Formulas of Prepared P-type Filtration Materials

	P-Control	P-type I	P-type II	P-type III
Fly Ash	10.0	10.0	10.0	10.0
Stabilized FGD material	6.0	6.0	6.0	6.0
Quick Lime (CaO)	1.2	1.2	1.2	1.2
Deionized Water	10.5	10.5	10.5	10.5
Wood Chip (<2.3 mm)	0	2.5	5.0	0
Wood Chip (2.3-3.6 mm)	0	0	0	2.5

Unit: g

Table 6. Formulas of Prepared N-type Filtration Materials

	N-Control	N-type I	N-type II	N-type III
Fly Ash	10.0	10.0	10.0	10.0
Stabilized FGD material	6.0	6.0	6.0	6.0
Red Mud (dried weight)	8	8	8	8
Deionized Water	10.5	10.5	10.5	10.5
Wood Chip (<2.3 mm)	0	2.5	5.0	2.5
MnO_2	0	0	0	2.0

Unit: g

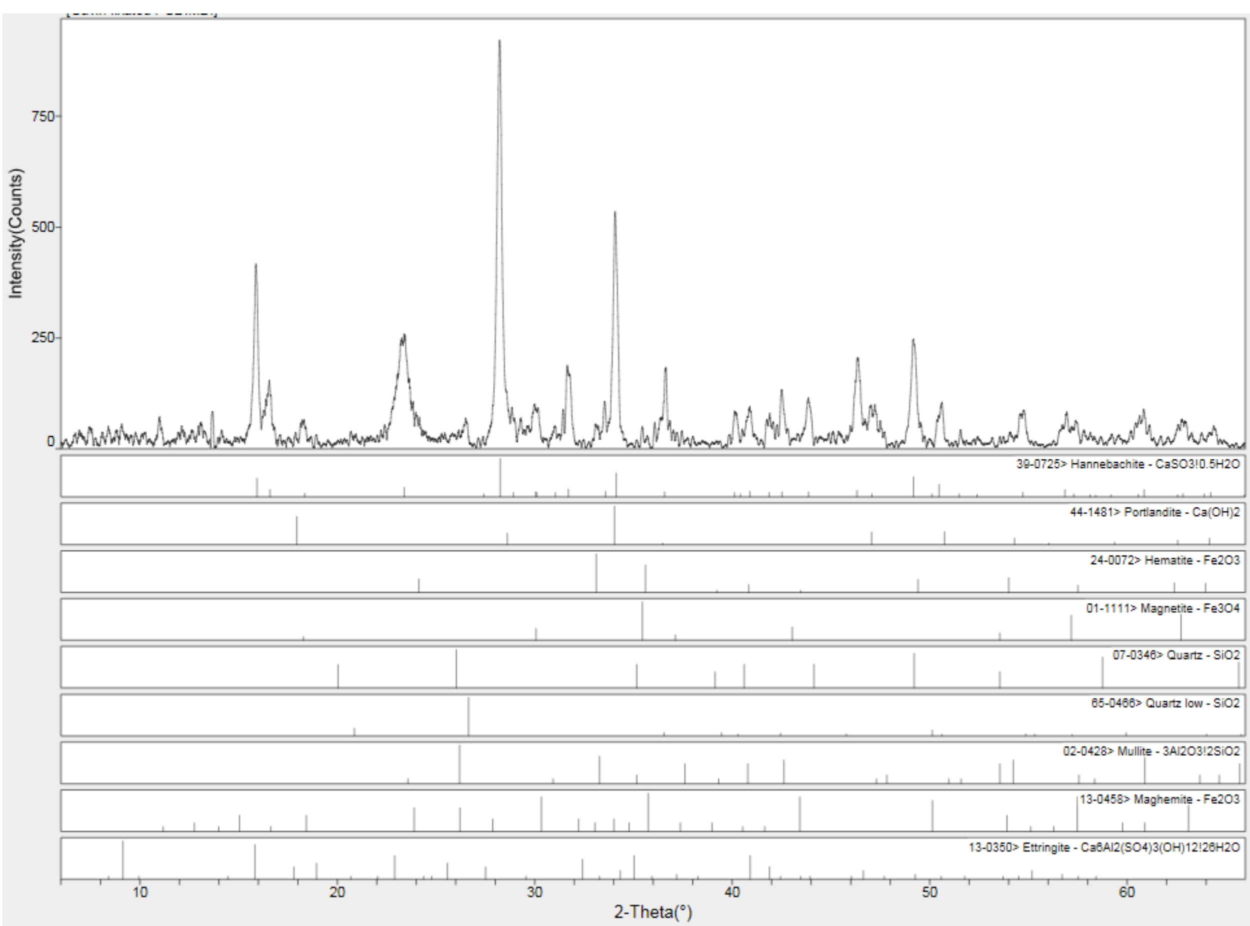


Figure 3. Mineral composition of stabilized FGD material

(a) (b)

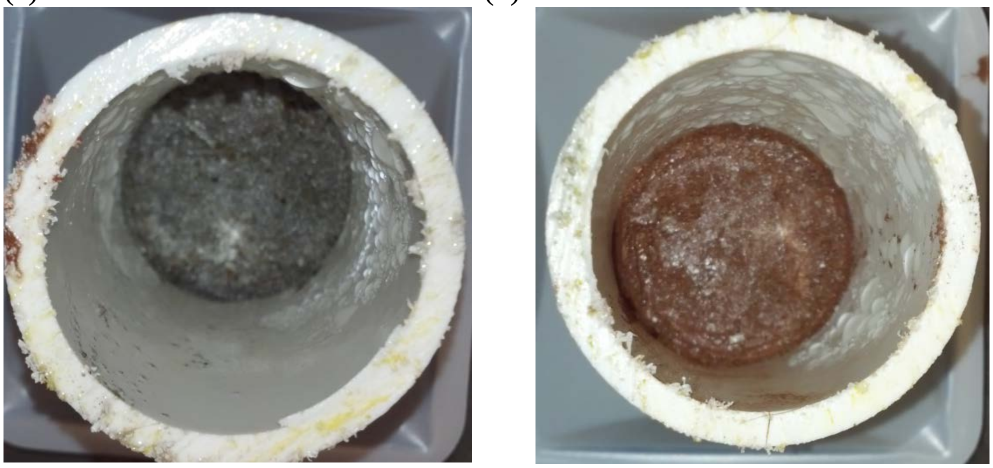


Figure 4. Prepared Pervious filtration materials. (a) P-type and (b) N-type.

These two types (i.e., P- and N-types) of pervious materials are expected to have selective sorption capacity, which can be used to sequentially separate and recover soluble phosphorous and nitrogen in agricultural drainage waters. In practice, two different pervious filter materials can be used in series. The dissolved phosphorous is expected to be selectively retained in the first pervious material (P-type) containing only fly ash and FGD material while allowing nitrate to pass through. Nitrate is captured in the second pervious material (N-Type) containing bauxite red mud, fly ash, and stabilized FGD material.

Adsorption Capacities

The nutrient adsorption capacities of P- and N-type materials were evaluated using the materials prepared from the formulas listed in Tables 5 and 6 for the P-Control and N-Control materials. For either type of the material, the adsorption experiment was carried out by adding six different amounts of the prepared solid, ranging from 0 to 1 gram, into six separate 125-mL HDPE bottles. Each bottle contains 100mL of either 250 mg/L of phosphate or 100 mg/L of nitrate solution. The bottles were then mixing by a tumbler for 24 hours at a rotating speed of 18 rpm. After mixing, the solution collected from each bottle was filtrated with 0.45mm filter and analyzed for NO_3^- or PO_4^{-3}.

The equilibrium concentrations of phosphate and nitrate in the solution after mixing as a function of material dosage are shown in Figure 5. As shown in the figure, over 97% of phosphate was removed by the P-type material with a solid-to liquid (L/S) ratio of 100. With the same L/S ratio, nearly 4% of nitrate was adsorbed by the N-type material.

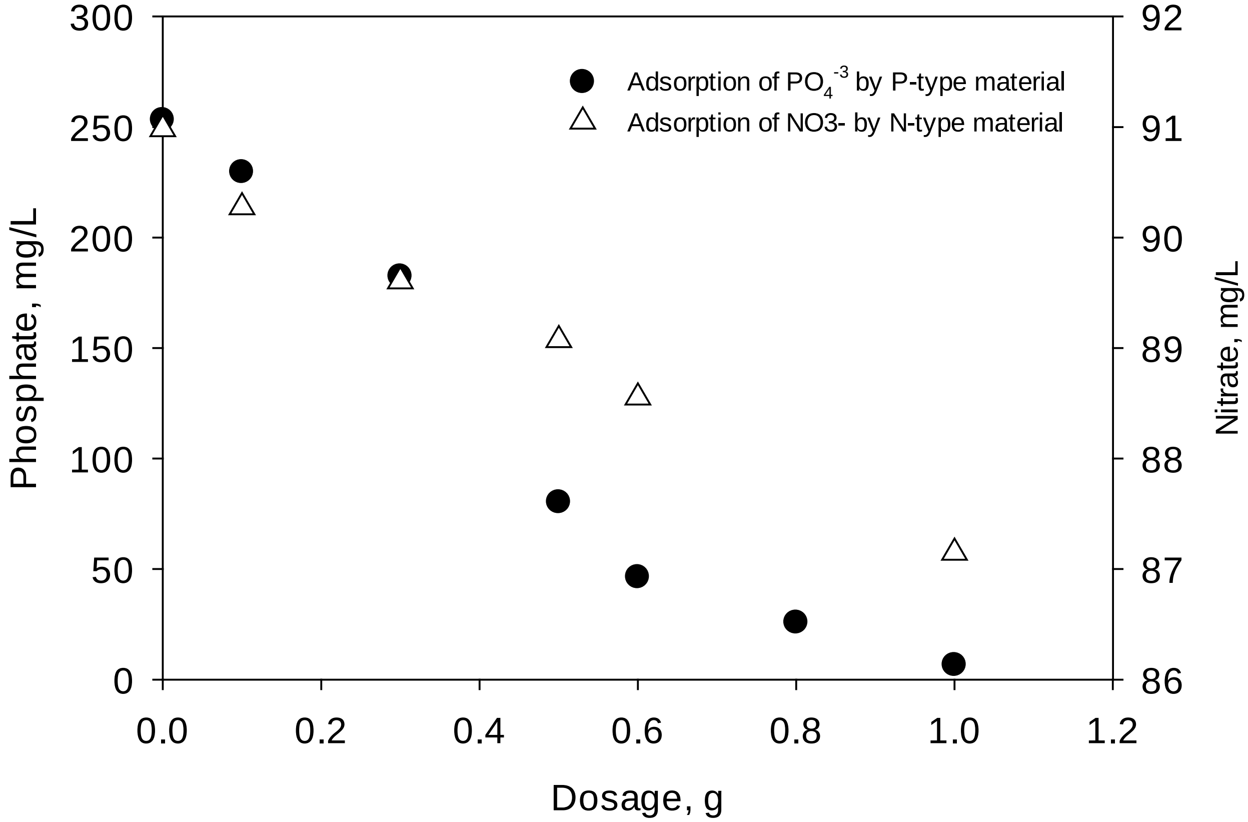


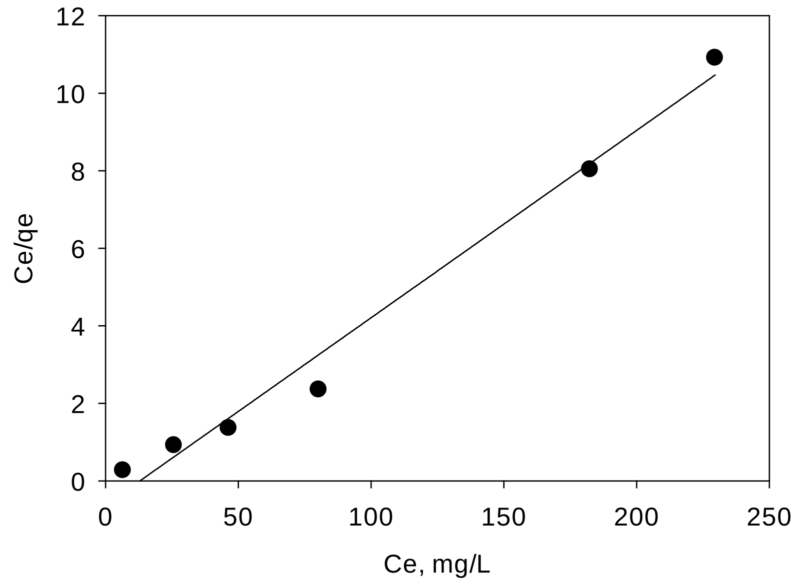
Figure 5. The equilibrium concentrations of phosphate and nitrate in the solution as a function of material dosage

The adsorption isotherms of phosphate on P-type material and nitrate on N-type material are illustrated in Figure 6. As shown in the figure, the adsorption isotherms of phosphate and nitrate can be expressed as Langmuir isotherm. The Langmuir isotherm equation is written as

$$\frac{C_e}{q_e} = \frac{1}{K \cdot Q_a^0} + \frac{C_e}{Q_a^0} \qquad \text{Eq. 1}$$

where q_e is mass of material adsorbed (at equilibrium) per mass of adsorbent; Q_a^0 represents the maximum adsorption capacity (monolayer coverage); C_e is the equilibrium concentration in solution when amount adsorbed equals q_e.; K is constant (L/mg).

(a)



(b)

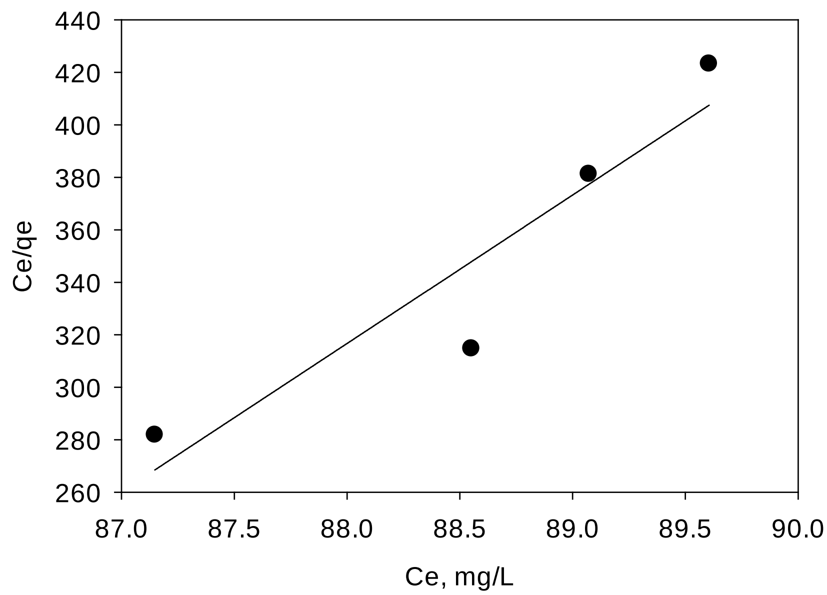


Figure 6. Langmuir isotherms for (a) phosphate and (b) nitrate

It is estimated that the maximum phosphate adsorption capacity of P-type material is 20.7 mg/g. For the N-type material, the adsorption capacity was approximately 0.18 mg/g, which is much less than the expected adsorption capacity.

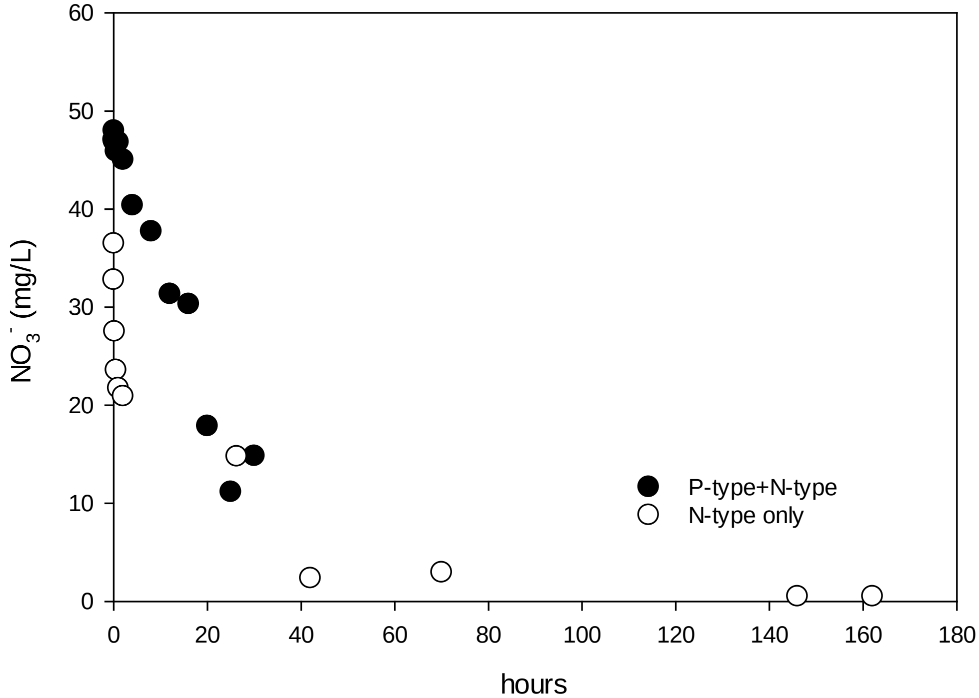
Column Test

Two series of bench-scale column tests were carried out in a close-loop mode to investigate the removal of nitrate and phosphate with extended contact time. The flow rate was kept at 1.13±0.17 mL/sec for both series. A simplified agriculturally degraded solution prepared with NaH_2PO_4 and $NaNO_3$ was used. In the first series, the solution was first introduced into P-type column and then N-type column. In the second series, only N-type column was used. A collection schedule was then setup to collect a series of eluent fractions based on pre-scheduled time interval. During each sampling interval, eluent was collected from the inlet and outlet of the first column, as well as the outlet of the second column in the first series, for nitrate and phosphate analyses.

The temporal trends of nitrate and phosphate at the inlet of the first column can be seen in Figure 7, which represent the concentrations in the storage tank. It was found that the concentration of nitrate in the first series decreased over 68.5% (from the original 47.1 mg/L to 14.8 mg/L) after 30 hours of circulation. In the second series, a similar removal efficiency (60.1%) was observed during the first 26 hours when only N-type column was used. The concentration of nitrate decreased to a level lower than the detection limit after 146 hours of circulation. In the case of phosphate, over 95% of the phosphate in the solution was removed within 30 hours of circulation in both series.

Results observed from the two series of column tests demonstrate that the pervious filter materials prepared in this study can effectively decrease the concentrations of nitrate and phosphate. However, the mechanisms involved in the decrease of nitrate need to be further investigated. Although the concentrations of nitrate showed a decreasing trend throughout the testing period, the samples collected from the inlet of the N-type column either had the same or slightly higher nitrate than the samples collected from the outlet (data not shown).

(a)



(b)

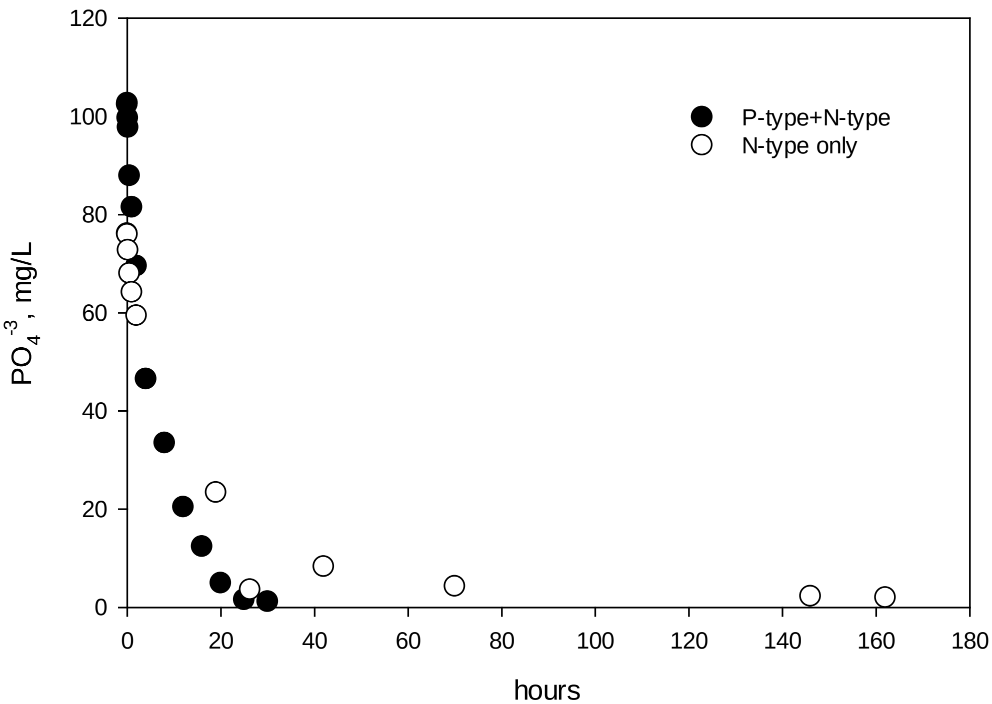


Figure 7. Temporal Trend of nitrate and phosphate in the close-loop column system

Tasks to be completed

The bench scale column test described in the "*Materials and Methods*" section will be continued. In addition, the integrities of physical and chemical properties of the pervious materials after adsorption will be evaluated.

Despite the great potential for the proposed filtration application, the major concern of reutilizing these by-products is the release of trace elements contained in the materials after being contacted with water. Cheng *et al.*[22] investigated the water quality impacts associated with using stabilized FGD material as a low permeability liner for a swine manure storage pond. Based on five-year worth of field monitoring data, the concentrations of arsenic (As), boron (B), chromium (Cr), copper (Cu), and zinc (Zn) were consistently found lower in the water passing through the liner than the water collected from the pond. Other trace elements, such as Cd, Se, and Hg were often below the analytical detection limits. Ruyter et al.[19] investigated the red mud accident occurred on October 4th 2010 in Ajka, Hungary by testing the plant toxicity and trace element availability with mixtures of red mud and non-contaminated soil. They observed the concentrations of trace elements in the leachate of red mud were either non-detectable or less than 20μg/L. In addition, Peters and Basta[18] added bauxite red mud directly to soil to reduce the bioavailable phosphorus. No excessive soil pH and increases of soil salinity, extractable Al, or heavy metals in soils were found in their study. Based on available field data, the application of coal combustion by-products and bauxite red mud has not been suggested to post adverse impacts on the environments.

However, to comprehend the overall benefits of reusing these by-products, it is vital to understand the leaching properties of the prepared pervious materials under different application scenarios.

Expected Outcomes and Significances

The outcome of this study is expected to provide:

(1) Initial feasibility evaluation of a potential beneficial utilization for by-products produced from coal combustion and aluminum production processes
(2) Insights regarding the interaction between nutrients and an agricultural emerging pollutant (i.e., estrogen) of FA zeolite-like material and the properties of biopolymers, and
(3) Results to be transferred in forms of peer-reviewed publications and conferences, and be based upon in preparing competitive proposal for external funding.

The advantage of using selective sorption materials in the filtration approach is the potential to recycle and reutilize nutrients and industrial by-products, which promotes agricultural production to be in accord with the principles of sustainability. FGD gypsum and stabilized FGD material have shown to improve the yield of crops by providing necessary elements (e.g., calcium), changing soil physical properties, and increasing water infiltration and storage when they are applied as soil amendments[20,21]. Hylander et al.[22], used different filter materials (i.e. limestone, Polonite®, and sand) to capture soluble phosphorus and evaluated the subsequent suitability for plant production. They observed some of recycled phosphorus achieved 76% of the yield increased by commercially available P-fertilizer. As demand for food increases, which results in more land to be used for agricultural purpose and a requirement for increased crop yields, the fertilizer demand have been projected to increase faster than world population[23]. With foreseeable increase in demand and depletion in reserve, use of recycled nutrients rather than a raw material is important step toward sustainable agricultural development. Currently, the majority of phosphate rock from mining goes into artificial fertilizer production[24]. It estimates that sources of high-grade phosphate ore deposits could disappear within the next 100 years at current use rates[25].

5. References

[1] Alexander, R. B., Smith, R. A., Schwarz, G.E., Boyer, E.W., Nolan, J.V., Brakebill, J.W. (2008) Differences in phosphorous and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. *Environ. Sci. Technol.*, 42, 822-830.

[2] Mueller, D.K., Helsel, D.R., Nutrients in the Nation's Waters--Too Much of a Good Thing?. U.S. Geological Survey Circular 1136, http://pubs.usgs.gov/circ/circ1136/circ1136.html#PUBS, accessed 2/4/2011.

[3] Smith, V.H., Tilman, G.D., Nekola, J.C. (1999) Eutrophication: impacts of excess nutrient inouts on freshwater, marine, and terrestrial ecosystems, *Environ. Poll.*, 100, 179-796.

[4] Smil, V. *Enriching the Earth: Fritz Haber, Carl Bosch, and the Transformation of World Food*. The MIT Press, 2001, Cambridge, U.K.

[5] Sharpley A, Foy B, Withers P (2000) Practical and innovative measures for the control of agricultural phosphorus losses to water: An overview. *J. Environ. Qual.*, 29, 1-9.

[6] Kleinman P.J.A., Sharpley, A.N., Buda, A. R., McDowell, R.W., Allen, A.L. (2011) Soil controls of phosphorus in runoff: Management barriers and opportunities. *Can. J. Soil Sci.* 91, 329-338.

[7] Sharpley A., Kleinman, P., and Weld J. (2010) Assessment of best management practices to minimize the runoff of manure-borne phosphorous in the United States, *New Zealand J. Agric. Res.*, 47, 461-477.

[8] Sonzogni, W.C., Chapra, S.G., Armstrong, D.E., Logan, L.T. (1982) Bioavailability of phosphorus inputs to lakes. *J. Environ, Qual.*, 11, 555-563.
[9] King, k.W., McDonald, J., Moore, J.F., Agrawal, S.G., Fischer, E.N., Balogh, J.C. (2010) Nutrient and pesticide removal from laboratory-simulated tile drainage discharge, *Trans. ASABE*,53, 769-777.
[10] Cheng, C.-M., Tu, W., Behrad, Z., Tarunjit B., Wolfe, W., Walker, H. (2007) Beneficial Reuse of FGD Material in the Construction of Low Permeability Liners: Impacts on Inorganic Water Quality Constituents, *J. Environ. Eng.*, 133, 523-531.
[11] Jin, N. fly ash Applicability in Pervious Concrete, Master Thesis, The Ohio State University, Columbus, OH 2010.
[12] Gall, H. E., Sassman S.A., Lee, L.S., and Jafvert, C.T., Hormone discharges from a Midwest tile-drained agroecosystem receiving animal wastes. Environ. Sci. Tech., 2011, 45, 8755-8764.
[13] Buchwald A., Dombrowski, k., and Weil, M. Influence of geopolymer binder composition on conversion reactions at thermal treatment, in Developments in porous, Biological and Geopolymer Ceramics: A collection of Papers Presented at the 31st international conference on advanced Ceramics and Composites, eds. Brito, M., Case, E., Kriven, W.M., Ceramic Engineering and Science Proceedings Volume 28. 257-271. Florida 2007,
[14] Allred, B. (2010) Laboratory batch test evaluation of five filter materials for removal of nutrients and pesticides from drainage waters. *Transactions of the ASABE*, 53, 39-54.
[15] Bryant, R.B., Buda A.R., Kkeinman, P.J.A., Church, C.D., Saporite, L.S., Folmar, G.J., Bose, S., Allen, A. (2012) Using fluegas desulfurization gypsum to remove dissolved phosphorus from agricultural drainage waters. *J. Environ. Qual.* 41, 664-671.
[16] Yao, W., Millero, F.J. (1996) Adsorption of phosphate on manganese dioxide in seawater, *Environ. Sci. Technol*, 30, 536-541.
[17] Cengeloglu, Y., Tor, A., Ersoz, M., Arslan, G. (2006) Removal of nitrate from aqueous solution by using red mud. *Sep. Purif. Technol*, 51, 374-378.
[18] Duxson P, Lukey G.C., Separovic F. van Deventer J.S.J., (2005) Effect of alkali cations on aluminium incorporation in geopolymeric gels, *Ind. Eng. Chem. Res.*, 44, 832-839.
[19] Ruters, S., Merten, J., Vassilieva, E., Dehandschutter, B., Poffijn, A., Smolders, E. (2010) The red mud accident in Ajka (Hungary): plant toxicity and trace metal bioavailability in red mud contaminated soil. *Environ. Sci. Technol.*, 45, 1616–1622.
[20] Chen L., Dick W., Nelson S., (2001) Flue Gas Desulfurization by-product additions to acid soil: Alfalfa productivity and environmental quality, *Environ. Poll.*, 114, 161-168.
[21] Punshon T., Adriano D.C., Weber, J.T. (2001) Effect of flue gas desulfurization residue on plant establishment and soil and leachate quality, *J. Environ. Qual.*, 30, 1071-1080.
[22] Hylander, L.D., Kietlinska, A., Renman, G., Siman, G. (2006) Phosphorus retention in filter materials for wastewater treatment and its subsequent suitability for plant production**.** *Biores. Technol.*, 97, 914-921.
[23] Haar, A. The Reuse of Phosphorus. Eureau Position Paper EU2-04-SL09, 2005; http://eureau.org/sites/eureau.org/files/documents/2005.02.21_recovery_of_phosphorus_from_sludge.pdf. Accessed 10/6/2012.
[24] Jasinski S.M., Phosphate rock, 2006 Minerals Yearbook. United States Geologci Survey, 2007, http://minerals.usgs.gov/minerals/pubs/commodity/phosphate_rock/myb1-2006-phosp.pdf, accessed 10/2/2012.
[25] Christen, K. (2007) Closing the phosphorus loop, *Environ. Sci. Technol.*, 41, 2078.

Scenario Analysis for the Impact of Hydraulic Fracturing on Stream Low Flows and Water Supplies: A Case Study of Muskingum Watershed in Eastern Ohio

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Scenario Analysis for the Impact of Hydraulic Fracturing on Stream Low Flows: A Case Study of Muskingum Watershed in Eastern Ohio

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Abstract

Oil and natural gas production in the United Sates has increased tremendously for the last few years. Significant amount of water is needed for the production of oil and natural gas through the application of advanced technique called Hydraulic fracturing (fracking). This has raised a serious concern about the potential impact on hydrological cycle, due to water withdrawal for fracking, especially for low flow period. Therefore, a comprehensive analysis is essential for the evaluation of stream low flow conditions due to unanticipated water withdrawal. In addition, the atmospheric greenhouse gases are believed to be increasing, leading to future climate change, which may alter the hydrologic flow regime in the future and threaten the hydrological and environmental sustainability. Therefore, this study was initiated to investigate the potential impact of fracking and climate change on stream low flows. Since limited modeling studies have been conducted to investigate the impact of hydraulic fracking for watershed scale studies, a systematic review and documentation of existing watershed models was conducted; this was important because an appropriate selection of watershed model for these studies is still a matter of investigation. A widely used watershed model, Soil and Water Assessment Tool (SWAT), was found to be appropriate for the representation of fracking process in term of spatial and temporal scale. Various future scenarios were developed based on the possible future climatic conditions, which was conducted in two steps: i) first, analysis was conducted for immediate future by generating probable set of climate data (precipitation, temperature) based on the historical records of the climate data; ii) second, climate change data from Coupled Model Intercomparision Project (CMIP5) using Max Planck Institute earth system model (MPI-ESM) were analyzed for

21st century to see the effect of climate change on stream low flows. Analysis showed that water withdrawal due to hydraulic fracking had localized impact on the water resources, especially during low flow period. 30% of the withdrawal locations showed more than 5% changes in 7 days minimum monthly flow. The flow alteration due to hydraulic fracking decreased with increase in the drainage area. Environmental low flows such as 7Q10, 4B3 and 1B3 also varied in a decreasing pattern with increased drainage area.

Similarly, the highest forced scenario, Representative Concentration Pathways (RCP) 8.5 under MPI-ESM climate model of CMIP5 was selected for the evaluation of the future climate change in the watershed. Three future periods 2035s (2021-2050), 2055s (2051-2070) and 2085s (2070-2099) was assessed against the baseline period (1995-2009). Lowest flow is projected to increase across the watershed during 2035s compared to remaining 50 years. Additionally, the 2035s climate outputs were integrated with current fracking trend to analyze the combined effect of fracking and climate change. This particular analysis was limited for first 30 years of 21st century (2035s), and analysis was conducted assuming current rate of fracking remains intact. The result was in consistent with the conclusion from the step one (mentioned above).While there was negligible impact on mean streamflows, some impact on 11 locations (out of 32) in 7 days minimum low flow, was detected. The variation was revealed only during low flow period indicating low flow period was the most critical period, especially for small order streams. This analysis under various fracking and climate change scenarios can be useful information for policy makers and planners for appropriate water resources management in future.

List of Abbreviations

AGNPS	Agricultural Non-Point Source
AnnAGNPS	Annualized Agricultural Non-Point Source
APEX	Agricultural Policy/Environmental Extender
BASINS	Better Assessment Science Integrating Point and Nonpoint Sources
BCCA	Bias Corrected Constructed Analogs
CMIP	Coupled Model Intercomparison Project
DEM	Digital Elevation Model
DWSM	Decision Support System for Agro Technology Transfer
ECHAM6	European Center-Hamburg
EMIC	Earth System Models of Intermediate Complexity
ESM	Earth System Models
GCM	Global Climate Model
GIS	Geographic Information System
GWPC	Groundwater Protection Council
HAMOCC5	Hamburg Ocean Carbon Cycle Model
HEC-HMS	Hydrologic Modeling System
HRU	Hydrologic Response Units
HSPF	Hydrologic Simulation Program-Fortran
HUC	Hydrologic Unit Codes
IPCC	Intergovernmental panel on climate change
JSBACH	Jena Scheme for Biosphere Atmosphere Coupling in Hamburg
MDG	Millennium Development Goals

MPI-ESM	Max Planck Institute earth system model
MPIOM	Max Planck Institute Ocean Model
NCAR	National Center for Atmospheric Research
NCDC	National Climatic Data Center
NCEP	National Center for Environmental Prediction
NIWR	National Institutes for Water Resources
NLCD	National Land Cover Dataset
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollution Discharge Elimination System
NSE	Nash-Sutcliffe's Efficiency
N-SPECT	Nonpoint Source Pollution and Erosion Comparison Tool
ODNR	Ohio Department of Natural Resources
OEPA	Ohio Environmental Protection Agency
PBIAS	Percentage Bias
PCMDI	Program for Climate Model Diagnosis and Intercomparison
RCP	Representative Concentration Pathways
RMSE	Root mean square error
RSR	RMSE- Standard Deviation Ratio
SDSM	Statistical Downscaling Model
STATSGO	State Soil Geographic
SWAT	Soil and Water Assessment Tool
UCRB	Upper Colorado River Watershed
USACE	United States Army Corps of Engineers

USDA	United States Department of Agriculture
USEIA	United States Energy Information Administration
USGS	United States Geological Survey
WARMF	Watershed Risk Analysis Management Framework
WCRP	World Climate Research Program

Chapter 1. Introduction

Water resources sustainability is a research topic of particular interest due to its impact in every aspect including economy, energy, ecology and welfare of living beings. Water should be properly used in order to continue human world in the indefinite future without affecting the hydrological cycle and the ecological factors (Gleick et al. 1998). Factors such as urbanization, drought, uncertain climate, flooding and many other anthropogenic activities affect the water resources sustainability. One of them is abstraction of water from different branches of water such as streams and reservoirs for different water use including irrigation, power plant, water supply, recreational purpose, and at present, the most controversial one, natural gas and oil. Likewise, there have been issues regarding the connection of energy source for the impact on regional water availability, quality and its dynamics.

Scope and Objectives

Recently, several drilling companies are advancing to Ohio for oil and gas development; therefore, drilling has been increasing tremendously on the Muskingum watershed. Significant amount of water is withdrawn from the streams and reservoirs without considering its imminent impact to water environment, ecology and human. The impact of water withdrawal may or may not be significant at the watershed or regional scale but certainly, it may have localized effect with alteration of hydrological regime in specific tributaries. In addition, global climate changes have a potential to change the stream low flows significantly. In this context, there is an urgent need to evaluate the impact of hydraulic fracking and global climate change on water resources of Muskingum watershed.

The specific research objectives of this study are:

1. To review existing watershed models, and based on review, select and develop the appropriate model, and apply for Muskingum watershed after model calibration and validation;
2. To develop and apply the model to assess the potential impact of water withdrawals under various water acquisition scenarios associated with hydraulic fracking, especially during low flow or drought period, at various spatial and temporal scales;
3. To assess the potential impact due to future climate changes of the 21st century and also evaluate the combined impact of hydraulic fracking and climate change on hydrological cycle during the first 30 years (2021-2050) of climate change period.

This report is divided into four chapters. Chapter 1 covers the background, scope, objectives, and remaining three chapters are organized in a journal paper format. Since each chapter is separate journal articles, readers may find some redundancy in content.

Chapter 2 describes the review of some watershed models with their potential capability to incorporate hydraulic fracking for watershed scale studies. Also, it describes the process involved during watershed model development in the Muskingum watershed which includes delineation, preparation of input data, model calibration and validation for flow parameter. Current fracking conditions are set up in this developed model to assess the impact in the watershed. This chapter is published in *American Journal of Environmental Sciences* as a peer review journal article.

In chapter 3, the possible impact of fracking in various climatic conditions generated based on historical climate of the region is discussed. The future climate change impact generated based on greenhouse gas emission scenarios is not the scope of this chapter and discussed in chapter 4. This article is currently review on *hydrological sciences journal*.

In chapter 4, CMIP 5 climate projection is used in order to assess the impact of future climate change on hydrology in the Muskingum Watershed during three future periods (2021-2050, 2051-2070, and 2070-2099). Additionally, first 30 years of climate change data is integrated with current rate of fracking in order to analyze how climate change would affect the future low flows from 2021-2050 assuming current rate of fracking is intact. This chapter is published in *Journal of Water Resources and Hydraulic Engineering* as a peer reviewed article.

Reference

Gleick, P. H. (1998). Water in crisis: paths to sustainable water use. *Ecological applications*, *8*(3),571-579.

Chapter 2. Hydrologic Modeling to Evaluate the Impact of Hydraulic Fracturing on Stream Low Flows: Challenges and Opportunities

Abstract

Hydraulic fracturing (fracking) has been increasing in the eastern part of Ohio for the last few years leading to the increased stress on water resources, particularly on the hydrological low flows. Yet, evaluation of the various impacts of fracking on stream low flows using appropriate tools is still a challenging issue, even though significant progress has been achieved in recent decades to advance the scientific tools and techniques for watershed modeling. While various existing watershed models are capable of addressing water resource issues, each model is unique and the appropriate selection of model depends upon several factors. Therefore, the objective of this study are: i) to review the current state of art for various available watershed models, including their potential capability, in order to conduct a study related to hydraulic fracking, and ii) to present a case study using best selected model application. Our review indicated that the Soil and Water Assessment Tool (SWAT) is one of the most competent models to assess water issues related to the fracking process at various spatial and temporal scales. The SWAT model incorporating hydraulic fracking is presented in a series of steps: i) in the first step, the preparation of input data for water use and hydraulic fracking is discussed, including detail calibration and validation of the SWAT model for this study; ii) in the second step, a case study is presented to evaluate the impact of hydraulic fracking with stream low flows by analyzing the current fracking trend in watershed; iii) finally, issues and challenges related to data availability and sources of water withdrawal is presented. The SWAT model was calibrated and validated both for daily and monthly scales for 9 various locations of the watershed, with a monthly Nash-Sutcliffe efficiency varying from 0.49 to 0.88 for calibration and from 0.55 to 0.86 for validation. Analysis indicates that fracking practices have negligible impact on annual and monthly flows,

with modest impact on seven days lows flows, especially at the localized scale, varying in the range of 5.2% to 10.6%.

Key Words: Hydraulic Fracturing, Models, SWAT, Low Flow

Introduction

Recently, there has been increasing availability and use of natural gas for the transportation sector and electrical production due to technological developments with hydraulic fracturing (fracking). Production of unconventional shale gas has increased significantly to meet the growing demand for energy and support economic development (USEIA, 2011). One of the most key aspects for the substantial growth of natural gas is the massive use of hydraulic fracturing. Annually, about 35,000 wells undergo some sorts of hydraulic fracturing in U.S (IOGCC, 2010). For State of Ohio, the Ohio Department of Natural Resources (ODNR) has projected that approximately 122 billion gallons of water will be needed if the State of Ohio drills all possible Utica wells (20,000). While the fracking technology has been considered useful in term of gas production and economic development, concern are raised about the large quantity of water needed for fracturing, and possible water resources management issues. Four to six million gallons of water are commonly needed in order to conduct fracking for one Marcellus or Utica shale well (OEPA, 2012). The water withdrawal at such a massive scale can reduce the water level in the streams, which may further reduce the surface water flows or deplete water storage in aquifers. Similarly, surface water withdrawal may also directly reduce the level in reservoirs, lakes and streams.

Regulatory and public agencies are also concerned about water withdrawals for hydraulic fracking. The impact of water withdrawal for fracking may result severe consequences; therefore, the timing, location and volume of water withdrawal for fracking are important particularly during low flow periods. Since hydraulic fracking came in practice recently, the unanticipated water withdrawal for hydraulic fracking can raise several questions about its potential impact on water resources and environment. For example, what are the possible implications on local water quality as the pollutant concentration increases due to decreased stream flows? More importantly, what are the consequences of withdrawing large amount of water from surface and groundwater resources on short and long term water availability?

In fact, there could be alterations in the flow system during various seasons as daily or monthly flows might be reduced far below from the environmental flow limits. This may cause crisis in water supply, aquatic life and water quality, leading to the complete threat in water resources sustainability. Since oil and gas industry is one of the booming sectors over the United States, and also more than 25 States of US have potential for oil and gas production, there could be a significant impact on hydrological cycle in future due to large scale oil and gas production. Therefore, there is a pressing need of a study in order to fully understand the hydrologic process at the watershed scale under the influence of fracking. For this, physically based watershed models might be appropriate tools as these models can represent the physical process within watershed and capable to make an analytical study. There are various watershed models which are capable to simulate the physical and dynamic activities within watershed in order to evaluate the effect of many watershed processes, management practices and anthropogenic influence on hydrologic process (Moriasi *et al.*, 2007). For the last few decades, water resources scientists are successful to develop and advance the existing watershed models, which are operational at various temporal and spatial scales, in order to represent the various anthropogenic influence and watershed intervention in models. Watershed models, which are fully capable to represent the watershed complexity in terms of land use, soil and digital elevation model (DEM) have been extensively explored to deal with water resources issues over the last few decades. However, there are limited reports or published articles which describe possible set of appropriate watershed models in order to simulate the watershed response under active hydraulic fracking conditions. Therefore, existing watershed models have to be carefully reviewed, and their potential capabilities/limitations to conduct study related with hydraulic fracking needs to be explored. In this context, this study is unique in two ways; i) first, it thoroughly reviews the existing watershed models with their potential capabilities and limitations, including issues and challenges in order to conduct simulation study under hydraulic fracking conditions; and ii) second, a brief case study will be presented to explain the various processes involved for hydraulic fracking study using the selected model based on the review. While there are several opportunities of utilizing various watershed models to deal with water resources issues, we will discuss several challenges for watershed modeling and future policies issues in active hydraulic fracking watershed in later part of this manuscript.

Watershed Models for Hydraulic Fracking

Since hydraulic fracking involves the water withdrawal from any location of the stream, reservoir and ground water, one of the approaches to consider this system in the model is to incorporate as negative point sources. In fact, water withdrawal for hydraulic fracking is somehow the opposite process of point source discharge. Alternatively, positive water use input can represent the water withdrawal as some models have these features. Therefore, watershed models which can incorporate the spatially located point sources in the system besides its advanced hydrologic features could be potentially considered for this type of study. The lists of widely used watershed models that can be potentially applied for the evaluation of the impact of hydraulic fracking on water resources, but not limited to followings, are:

- Hydrologic Simulation Program-Fortran (HSPF) or Loading Simulation Program C++ (LSPC);
- Soil and Watershed Assessment Tool (SWAT);
- European Hydrological System Model (MIKE SHE);
- Agricultural Policy/Environmental Extender (APEX);
- Watershed Risk Analysis Management Framework (WARMF);
- Hydrologic Engineering Center-Hydrologic Modeling System (HEC-HMS);
- Watershed Assessment Model (WAM)

The details of these model components and their potential capabilities to incorporate hydraulic fracking have been summarized in Table 1-1. These models work mostly in continuous scale with daily and sub-daily output for streamflow. In addition, these models can incorporate the addition and diversion (withdrawal) of the water for fracking from the various points of the watersheds (Table 2-1). Although all above-mentioned models are potentially capable to simulate watershed response and have their unique features, selection of appropriate model is a crucial step in order to represent hydraulic fracking for hydrologic analysis. For example, a model which is very proficient for urban area study may not be appropriate for agricultural watershed and vice versa. More importantly, model selection depends upon several factors including modeler's knowledge, understandings and technical capabilities, availability of data and several other factors. While the description of all the model processes and the model structure is beyond the scope of this article, the following section briefly presents the major features of the existing watershed models to represent fracking for hydrologic assessment. The readers can refer various journal articles for details of the model description (Borah and Bera, 2003).

Hydrologic Simulation Program-Fortran (HSPF) (Bicknell *et al.*, 1996) is a watershed model for continuous simulation, which simulates hydrology and water quality including non-point sources and point sources. It considers simulation on pervious, impervious surface, stream channels and reservoirs, respectively for the simulation of stream flow and water quality. It is also called as parameterized intensive model as some of the component are lumped into parameters. HSPF and SWAT both can be potentially used for hydraulic fracking studies as both models have been used and compared in various watershed conditions.

Recently, several literatures were published based on the comparison between SWAT and HSPF (Singh *et al.*, 2005; Van Liew *et al.*, 2003; Im *et al.*, 2003). For example, Xie *et al.* (2013) compared the performance of HSPF and SWAT for hydrologic analysis in Illinois River. The authors showed that HSPF depends on the calibration method to achieve better result, and SWAT can achieve better result despite of the limited data for calibration. Although, HSPF can simulate better sub-daily streamflow simulation, it requires numerous parameters to characterize hydrological cycle with intensive calibration process (Im *et al.*, 2003; Saleh *et al.*, 2004). Borah and Bera (2003) reviewed both SWAT and HSPF and concluded that SWAT is a very promising model in order to conduct study on agricultural watersheds, and HSPF is capable for simulation in mixed agriculture and urban watersheds. Since this study is primarily focused for stream low flow conditions due to water withdrawal for fracking, SWAT could be a better choice as SWAT is considered as a better simulator on low flow (Singh *et al.*, 2005).

Borah and Bera (2003) reviewed eleven watersheds models and found that HSPF, MIKE SHE, and SWAT have strong hydrologic component applicable to watershed-scale catchments. SWAT model was also compared with fully distributed MIKE SHE and authors concluded that both models are equally competent during calibration (El-Nasr *et al.*, 2005) while performance of MIKE SHE model was marginally better for overall stream flows prediction.

Golmohammadi *et al.* (2014) evaluated three widely used hydrological distributed watershed models: MIKE SHE, APEX and SWAT for flow simulation of small size watershed in, Canada. MIKE SHE was concluded as more accurate for simulating streamflow at watershed outlet and SWAT was regarded as another potential model as there was no significant difference in model performance.

A Report on Model selection (USACE and DES, 2008), for a study in Central Oahu Watershed, sorted out few highly used watershed models including SWAT, WARMF, and HSPF based on various specific model skills. Authors reported that WARMF model was less recognized than SWAT and HSPF. Similarly, the successful applications of HEC-HMS (Verma *et al.,* 2010) and WAM model for watershed scale studies (Bottcher *et al.,* 2012) have been described in several studies.

Even though we found models performance rating different for different application studies for various models, we selected SWAT model due to numerous reasons: i) SWAT models has advanced in comparison to other models and can disintegrate watershed into multiple subbasins and hydrologic response units (HRUs) for continuous simulation of flow at various scales (Jha, 2011); ii) model is widely accepted worldwide by scientific community and well supported by USDA; iii) model is also considered suitable for the ungagged watershed (USEPA, 2012) and watershed characterized with limited data. SWAT has been widely used for the assessment of the impact of intensive water use on the water balance and its components. In addition, SWAT is user friendly and new users can successfully apply it for the analysis of various water resource problems. It has been extensively supported through various international conferences, training workshops, online swat user group forum, broad online documentation, supporting software and open source code. While Mike SHE and HSPF are equally competent, SWAT model is chosen for this study based on its historical credentials, diverse application and open source code so that it can be modified for the intended purpose.

The successful model application for SWAT varies from drainage areas of 7.2 km^2 to 444,185 km^2 (Douglas-Mankin *et al.*, 2010). Several journal articles have been published on the application of this model to assess low flow conditions (Rahman *et al.*, 2010; Steher *et al.*, 2008) and the likely impact of many management practices on runoff (Arabi *et al.*, 2008). Since various publication records reveals enough evidences that SWAT can be potentially applied for wide and diverse watershed conditions (Gassman *et al.*, 2007; Gassman *et al.*, 2010), this is a unique opportunity to apply this model for the assessment of impact sustained due to hydraulic fracking. A systematic approach has been presented in the following section to explore the potential of SWAT model to incorporate hydraulic fracking in the watershed for the hydrologic assessment.

Overview of SWAT

SWAT is a physically-based watershed model, which is developed to predict the long term impact of watershed management in terms of hydrologic and surface water quality response of large watershed (Arnold *et al.*, 2007). SWAT simulates different physical and hydrological process across river watersheds. The model is popularly used across the various regions of the world and has many peer review publications (Gassman *et al.*, 2007; Gassman *et al.*, 2010).

Initial input to SWAT model is geographical information such as digital elevation model to spatially delineate watershed in terms of different sub-watersheds. Further, land use, soil and slope information are utilized to subdivide the sub-watersheds into smaller hydrologic response units (HRU's), which are composed of similar land use, soil and management characteristics.

The loss in flow is due to evapotranspiration and the transmission of flow through the bed. Potential evapotranspiration is determined by various methods such as Hargreaves method (Hargreaves and Samani, 1985), Penman-Monteith (Allen, 1986; Monteith, 1965), and Preistly-Taylor (Priestley and Taylor, 1972). SWAT consists of two components: i) runoff generation through the land; ii) and movement of water using appropriate routing scheme. The readers are suggested to refer SWAT user's manual for water balance equation adopted in SWAT model.

The model estimates the surface runoff from each HRU using two infiltration methods; Soil Conservation Service's curve number (CN) method (USDA, 1972) or the Green and Ampt infiltration method.

Since fracking has potential to threaten the management practices in critical conditions due to the alteration of the volume and the intensity of water withdrawal both at spatial and temporal scales, SWAT model can be utilized to incorporate water withdrawal for fracking in a similar way that it has been used for other water use and withdrawal. For example, simulation of irrigation on agricultural land is performed under five sources: reservoir, stream reach, shallow aquifer, deep aquifer and a water body out of watershed. That is, users can utilize any of these sources for providing additional water input and water withdrawal through positive and negative value, respectively. Few options for incorporating water withdrawal for hydraulic fracking are: i) to use point sources option in SWAT model with negative value, ii) imitate water withdrawal and irrigation scenario in the agricultural practices for fracking assessment. Alternatively, water use input as positive and negative value can be used as an option to represent water withdrawal and water discharge as sink and source in SWAT model to represent hydraulic fracking. In addition, the incorporation of GIS technology in SWAT provides ample potential for inputs and response through spatial locations of fracking operations. The simulation in SWAT can be executed for any particular desired dates and period.

While SWAT model was used within the Fayetteville Shale in Arkansas for analyzing the potential impacts of water extraction for hydraulic fracturing (Jackson *et al.*, 2013), we are not aware of published research paper using any hydrologic models to assess the impact of hydraulic fracturing on stream low flows. Even though EPA has initiated to conduct a study to evaluate the impacts of fracturing on water resources using SWAT model in upper Colorado River watershed, the result has not been published yet in peer review journals.

The detailed process for development of the model, which includes watershed delineation, preparation of input files, model calibration, parameterization and validation, is described in the following section.

Study Area

The simulation study was focused in the Muskingum watershed (Figure 2-1) of eastern Ohio, which is one of the most affected regions due to hydraulic fracking. Watershed covers a significant portion of Ohio State (20%) with an approximately 8,000 square miles in area. The watershed covers some or entire portion of the 26 counties in Ohio.

Originating at the union of Tuscarawas and Walhonding River near Coshocton, Muskingum River, the largest river in the watershed, eventually drains into the Ohio River at Marietta after flowing 109 miles to the South. Some of the major sub streams of the river are Tuscarawas, Walhonding, Licking River and Wills Creek. The Watershed is a HUC-4 watershed (0504), which is subdivided into number of HUC-8 level watersheds. The maximum, minimum and average flows at the outlet of the watershed are 23,900 cfs, 477 cfs, and 2,760 cfs, respectively. The average annual precipitation over the entire watershed is approximately 39 inches. The minimum elevation range in watershed from sea level is 177 and maximum up to

459 m. Watershed is characterized with several lakes and reservoirs for water supply, recreation and flood reduction purposes. Interestingly, more than 90% (approximation) of natural gas wells in Ohio lie in this watershed (Figure 2-2); most of them are concentrated in the eastern portion of the watershed.

Methodology

Model Input

The current version of the SWAT model (SWAT, 2012) was utilized for this study. The model requires the inputs including digital elevation model (DEM), land use, soil, reservoir, weather, water use, point source, for successful simulation of the stream flows. The data needed for model development has been presented in Table 2-2 with necessary source and format.

30 m resolution DEM from USGS National Elevation dataset and ARCGIS were utilized in order to delineate stream networks. The watershed was delineated with a number of subbasins (406). Since the National Land Cover Dataset (NLCD) is compatible with the SWAT model, datasets of 30 m resolution (NLCD, 2006) was utilized from NLCD database. The reason for selecting NLCD data for year 2006 is to adequately represent the land use pattern during model calibration period (2002-209). The watershed land use was mainly dominated with forest (47%) comprising both deciduous and evergreen. Other major land use categories of the watershed include agricultural land with row crops (23%) followed by hay (19%), and urban areas (10%). Nearly 1% of the watershed includes industrial area, water, range grass, southwestern arid range etc. In order to adequately represent the storage effect in hydrologic analysis, the spatial location of existing major reservoirs (Table 2-3) were identified in the watershed and manually added at a suitable location (Figure 2-1). Since the watershed is relatively larger in size, we utilized the State Soil Geographic dataset (STATSGO) (USDA, 1991), which is in-built in SWAT model. In the next step, we selected the threshold in each subbasin for landuse (5%), soils (15%) and slopes

(15%) in order to eliminate minor land uses and assign multiple HRU's for each sub basin resulting 6176 HRUs in the watershed. Since hydrological modeling requires spatially distributed long term climate data, data including precipitation, maximum temperature and minimum temperature, located at various spatial locations within the watershed, were utilized from National Climatic Data Center (NCDC) in order to capture the spatial variability of precipitation and temperature. Only 23 precipitation stations and 19 temperature gauge stations, with continuous data record for a longer period, were available within the watershed (Figure 2-1). Rest of the meteorological data including solar radiation, wind speed, and relative humidity were utilized through weather generator option available in SWAT model. Daily streamflow data available from period 1993 to 2009 were downloaded at 9 USGS locations (Figure 2-1) within the watershed in order to conduct multi-site model calibration and validation. Since watershed comprises number of multi-purpose reservoirs, storage effect and flood reduction due to these reservoirs should be incorporated in a model. For this, daily mean outflows from reservoir were obtained from US Army Corps of Engineers (USACE) through the personal communication.

Major point sources data (>0.5MGD) were downloaded from Ohio Environmental Protection Agency (OEPA) and included in SWAT model calibration. Similarly, water use data for various purposes including surface and ground water, irrigation, power plant, industry, mineral extraction, water supply and hydraulic fracturings were downloaded from Ohio Department of Natural Resources (ODNR). However, ODNR does not provide any withdrawal information for less than 100,000 gal/day; therefore, information was additionally confirmed for smaller withdrawal from OEPA in order to include all facilities especially for water supply data. The spatial locations of oil and natural gas wells and sources for freshwater, which was needed as model input were utilized from ODNR. Since, a part of water withdrawal for hydraulic fracking is recycled, we utilized the information related with recycled water, fracture data and fresh water required per well from fracfocus, the National hydraulic fracturing chemical registry. Figure 2-3 shows the fresh water withdrawal, part of the water recycled and the vertical depth of wells in various years in Muskingum watershed.

Calibration and Validation

A hydrologic model needs to be appropriately calibrated and validated before conducting any scenario analysis related with watershed management. Since model calibration is a process of determining the suitable model parameters with successive iteration, SWAT-CUP (Abbaspour, 2007) was selected to calibrate the suitable model parameters using continuous flow data from 2001 to 2009. The optimizing algorithm, SUFI-2 was utilized, which takes into account the possible parameters ranges and determine the optimum model parameters within the uncertainty range of 95% (Abbaspour *et al.,* 2007). Twenty one model parameters (not shown) were selected in order to conduct multi-site model calibration and validation within the watershed. The selected model parameters were based on the similar past studies (Abbaspour *et al.,* 1999; Abbaspour *et al.,* 2007; Faramarzi *et al.,* 2009; Schuol *et al.,* 2008; Yang *et al.,* 2008).

Since the USGS observed flow data were available since 1993, the model was simulated for 15 years from 1995 to 2009. The model was allowed three years (1993- 1995) of spin up time period before the simulation period in order to stabilize the hydrological conditions such as antecedent moisture content and base flow. The model was calibrated at 9 various locations of the watershed in a daily as well as monthly time scale using the continuous stream flow data available from 2002 to 2009. The SWAT model simulation was validated with independent datasets from period 1995 to 2001 and the performance of the model was evaluated using various statistical measures including, Percent of bias (PBAIS), coefficient of determination (R^2) (White *et al.*, 2005), and Nash-Sutcliffe coefficient of Efficiency (NSE) (Nash and Sutcliffe, 1970) .

Model Evaluation Criteria

Performance evaluation of the model is always a key issue for any hydrological modeling as there is not a single best statistical measure to check the performance of a model's outputs with observed data. There are three non-dimensional and one dimensional measure which are widely used to assess the goodness of fit. These model performance measures are coefficient of determination (R^2), Nash-Sutcliffe coefficient of Efficiency (NSE) (Nash and Sutcliffe, 1970), Root Mean Square Error (RMSE), and percent of bias (PBAIS), which are mathematically represented as follows.

$$R^2 = \frac{\left[\sum_{i=0}^{n}\left(O_{obs,i} - \bar{O}_{obs}\right)\left(O_{sim,i} - \bar{O}_{sim}\right)\right]^2}{\left[\sum_{i=0}^{n}\left(O_{obs,i} - \bar{O}_{obs}\right)^2\right]\left[\sum_{i=0}^{n}\left(O_{sim,i} - \bar{O}_{sim}\right)^2\right]}$$

R^2 varies from 0 to 1 which indicates the proportion of the total variances in the observed data.

$$NSE = 1 - \frac{\sum_{i=1}^{n}(O_{sim,i} - O_{obs,i})^2}{\sum_{i=1}^{n}(O_{obs,i} - \bar{O}_{obs,})^2}$$

NSE is a measure of how well the actual and simulated data fits and its coefficient varies from -∞ to 1. The perfect model shows the value very close to 1.

$$PBIAS = \frac{\sum_{i=1}^{n} O_{sim,i} - \sum_{i=1}^{n} O_{obs,i}}{\sum_{i=1}^{n} O_{obs,i}} = \frac{\bar{O}_{sim,i-} \bar{O}_{obs}}{\bar{O}_{obs}}$$

Here, $O_{obs,i}$ and $O_{sim,i}$ are observed and simulated streamflow for each ith observation and n is the number of observations. Similarly, $\bar{O}_{obs}$ and $\overline{ON}_{sim}$ are the mean observed and simulated streamflow. PBIAS is simply an indication of the deviation of the simulated result with the observed data.

Similarly RMSE-observations standard deviation ratio (RSR) is also another model statistics, which standardizes RMSE with standard deviation of observed data.

$$RSR = \frac{RMSE}{STDEV_{obs}}$$

The Ideal value of RSR is zero indicating the perfect match of the observed data and simulated result.

Result and Discussion

Fracking and Analysis

The calibrated and validated SWAT model was integrated with water use, point sources data and fracking condition of year 2012 in order to analyze the streamflow with given rate of fracking condition. Monthly consumptive water use was provided in model from the water use input file based on the removal of water from reach, shallow aquifer, and reservoirs within subbasin. Since the continuous lake outflow data were not available, 50 percentile of the available data from USACE was applied for a period of 1995 to 2009 in order to best represent the lake outflow. When this study was conducted, only the fracking data up to year 2012 were available; therefore, current period in this manuscript actually represent the conditions of year 2012.

Model Simulation

The performance of the model was evaluated through various criteria including visual inspection and goodness of fit. The performance of the model was satisfactory both in daily and monthly scale during model calibration and validation period. Figure 2-4 and Figure 2-5 provide the box plot of daily and monthly statistical parameters including *NSE*, R^2, *RSR* and *PBIAS* to measure the performance of the model. In majority of watershed locations, *NSE, RSR* and *PBIAS* were well above the minimum suggested ranges of Moriasi *et al.* (2007) (*NSE* > 0.5, *RSR* ≤ 0.7 and *PBIAS* ±25%). *NSE* values varied from 0.40 to 0.65 for daily streamflow calibration, and it varied from 0.4 to 0.65 for streamflow validation (Figure 2-4). Similarly, the NSE was obtained in the range of 0.49 to 0.89 for monthly streamflow calibration, and 0.55 to 0.86 for monthly streamflow validation (Figure 2-5). However, the validation of the model was limited to 8 USGS stations as the long term data were not available at the outlet. The nearest stations (USGS 3142000) near to the outlet also did not have a continuous record beyond 1998; therefore, validation at this station was accomplished using three years of data (Figure 2-5).

Since SWAT model can relatively better simulate the monthly streamflow compared to daily streamflows, the performance of the model was relatively promising in monthly scale compared to daily scale. Performance of the model was satisfactory for all stations except at one station (USGS 03136500). The model performance at this station was affected due to lack of reservoir outflow data as this station was immediately below the reservoir. As expected in any watershed modeling, the performance of the model was relatively better in the downstream portion of the watershed as these stations covers large portion of watershed. Furthermore, the performance of the model was also assessed through the visual inspection of observed and simulated streamflow time series over a long period. The performance was found to be satisfactory during calibration (Figure 2-6) and validation period (Figure 2-7). Despite of the slight underestimation in daily and monthly simulated peak, the model captured the overall spatial and temporal pattern of stream flow satisfactorily.

Impact due to Fracking

Our analysis depicted the consistent increasing drilling trend in Muskingum watershed. Model was used to quantify the effect of these withdrawals over this period. 32 subbasins were affected by fracking, which had drainage area less than 140 km^2. Analysis was categorized in yearly and monthly periods; mean for current year, dry and high flow season were calculated, separately. Results revealed that the greater alterations were found in seasonal mean (high flow) than the yearly mean flow. However, these changes were only detected in 5 subbasins out of 32 subbasins, with less than 1.5 percentage difference, indicating that impact is not significant in yearly and seasonal mean flow (high flow season) in the streams. Also, dry flow seasonal mean showed significant variances only in two subbasins (5.9% and 20.16%) with no significant changes on the remaining subbasins. However, the difference was noticed when the monthly analysis was performed. Minimum 6 percentage difference was observed while comparing current and baseline scenario.

Since it is essential to maintain environmental low flows for sustainable water availability including downstream right, aquatic habitat and others, low flows for current fracking period was evaluated considering water withdrawal over the watershed. The result showed that the water withdrawal during low flow period (August through November) was about 43% of the total water withdrawal (Figure 2-8). However, this difference was relatively more when hydraulic fracking effect is analyzed over the 7 days minimum monthly low flows. Out of 32 subbasins, 8 subbasins with less than 118 km^2 drainage area revealed more than 5% difference in 7 days minimum monthly flow while comparing baseline (without hydraulic fracking) and current scenarios (Figure 2-9). Figure 2-9 also presented both the monthly mean and seven days monthly minimum flows only in 8 subbasins; the impact in other subbasins was not significant. Interestingly, major impacts were observed in first order streams. The subbasins which showed the differences in 7 days low flows and monthly minimum flows are displayed in Figure 2-9. The spatial location of the affected subbasins is shown in Figure 2-10. In general, our analysis shows lesser impact on the annual and seasonal water balance; however, the effect might be critical over low flow such as 7 day minimum flow, especially on lower order of streams.

The case study revealed that the impact of water withdrawal is revealed during low flow period, and this effect is particularly true in small order streams. Our analysis does not show any significant impact in monthly and annual scale. Nevertheless, some localized impact in the first order stream for few days can be encountered. Similarly, baseflow variation during low flow period suggests that ground water is dominant component for the discharge into most of the rivers during this period. However, the result might be different in various subbasins in accordance with the existing water use and point source discharge of that particular subbasin.

Modeling Challenges for Hydraulic Fracking Study

While we selected SWAT models for this study, the issues and challenges associated with SWAT model application will be similar to the issues associated with abovementioned models discussed in this article if the users select other models. Therefore, we will present the challenges and difficulties in model application in a generic term, irrespective of the types of the model chosen.

Model Calibration, Validation and Uncertainties

The lacks of water withdrawal data for various purposes such as irrigation, water supply and for various other purposes are neither well recorded nor easily available. On the other hand, lack of required datasets for long term calibration and validation is another issue to evaluate the appropriate selection of watershed models in order to quantify the effect of hydraulic fracking on water resources management. The quantity of water needed for hydraulic fracking also varies from case to case basis, and there is no specific record of the water withdrawal information for hydraulic fracking as it relies in the geological condition, depth, lateral length, type of rock and other physical conditions of the sites. The water used for fracking eventually ends up in underground disposal wells and hence removed from the hydrologic cycle. The water used for fracking also primarily used from various sources including municipal water systems, streams, reservoirs, private ponds etc. The exact water withdrawal location is always uncertain and raises several questions for the reliability of the modeling results. In addition, certain amount of water will be disposed to the streams/rivers after treatment of drilled wastewater. The amount of water disposed also varies from location to location; exact data are needed to make a reliable prediction using models. Therefore, modeler needs to make an assumption due to lack of exact disposed water from each site. Also, the water disposed could be surface water or in some case ground water. More importantly, the representation of exact timing of water withdrawal and disposal within a year in a simulation model is another challenging issue because company can withdraw water any time after receiving the license from concerned federal agencies (in this case, ODNR). Similarly, modeler's also needs to rely on the assumption of exact location of well sites because companies may drill at any convenient location of the watersheds. Another limitation is that modeler is always uncertain about the possible changes in population and land use change practice in near future while developing future scenarios of water acquisition due to hydraulic fracking.

Flowback and Produced Water Effect

The disposed flow may have pronounced effect on water quality at certain sections of the stream but not for the entire streams. In future, detail database should be prepared and existing models should be modified to incorporate the possible consequences of accidental discharge, leak and spills of flowback and produced water.

Scaling Issues

The impact of fracking may vary both at temporal and spatial scales, and intensive study is needed for any generalization of the study. For instance, the impact of withdrawal may have different range of impact on daily water availability and monthly water availability. Furthermore, the spatial distribution of the fracking locations within the watershed also affects the net amount of water within the tributaries. That is, if the frakings wells are concentrated near to the particular tributaries, it may have significant impact for those particular tributaries; however, it may not have substantial effect far downstream of the stream or at the watershed outlet. Therefore, modeler's need to acquire appropriate information of fracking wells and their spatial locations. The scale of fracking that affect water resources sustainability is still a matter of further investigation and could be an interesting topics for future research.

Consequences and Future Outlook

The oil and gas production may result impact on water resources and environmental sustainability leading to the demand on policy changes in future. For example, the hydrological and biological conditions of the watershed will be affected due to fracking; therefore, water withdrawal for fracking should be incorporated in NPDES permitting and TMDL development of the affected watershed in future. The hydraulic fracking may also change the current land use practices; therefore, the locations of best management practices to be adopted within the watershed might be shifted in future.

Proper development of complete database of hydraulic fracking information is needed for the use of current watershed models to deal with the complex intervention in watershed due to hydraulic fracking. For this, stakeholder participation should be encouraged and information should be shared through the active stakeholders' consultation. More specifically, the complete database of hydraulic fracking is needed in future before analyzing impact of hydraulic fracking on water resources.

US government should devise future policies for the environmental safeguard and water resources sustainability against fracking. Decision-support systems will be useful to provide policy level solutions to the active stakeholders related with hydraulic fracking and its impact on water resources. Therefore, hydrologic models should be advanced to incorporate hydraulic fracking in future. While the site specific conditions may be different from location to location, generic effect of fracking on water resources can be extrapolated using such simulation study.

Conclusion

In this paper, the state of art of existing watershed models has been presented to conduct simulation study due to water withdrawal associated with hydraulic fracking. The capabilities of widely adopted 7 watershed models (HSPF, MIKESHE, SWAT, WARMF, APEX, WAM and HEC-HMS) to incorporate fracking was systematically reviewed and documented with proper citation. Our study does not warrant only the above-mentioned models to incorporate fracking for simulation study as there are numerous watershed models available. While most of the watershed models have an option of incorporating hydraulic fracking process, the SWAT model was selected among the various candidate watershed models. A separate case study was presented to demonstrate the potential application of SWAT model for the assessment of hydraulic fracking and its impact on the water resources, especially for low flow period.

Simulated flows in ungauged locations under the current fracking situation were used to assess the potential impact of water withdrawal for hydraulic fracking on water resources both at spatial and temporal scales. The study suggested that the impact was more significant during low flow than average flow or peak flow period. 7 days minimum flows showed some variation when compared with the 7 days minimum flow without fracking indicating that proper regulations of drilling activities are needed during the low flow period. Such flow alteration may bring the daily flow below the environmental flow limits, which may eventually threaten the water resources sustainability.

Even though simulation studies are always associated with a certain degree of uncertainty, this study concludes that SWAT could be a potential tool for the hydrologic assessment and water quality evaluations in the future under different scenarios of hydraulic fracking. This study also concludes that fracking has a modest effect on seven day low flows; therefore, this study addressed water quantity impacts due to hydraulic fracking, which continues to be an interesting research topic for the future.

Overall, this research summarizes following issues which might be helpful to improving hydrologic assessments and management strategies in the future.

- Watershed models can be utilized to evaluate several water acquisition scenarios associated with hydraulic fracking both at spatial and temporal scales with some modification, if any.
- Lack of sufficient data and information for hydraulic fracking study at various spatial locations is one of the major limitations.
- Complete database regarding the information of exact withdrawal and water disposal location is needed in future.
- Coupling of groundwater and surface water modeling process is needed in future for thorough investigation.

References

Abbaspour, K. C., M. A. Sonnleitner and R. Schulin, 1999. Uncertainty in estimation of soil hydraulic parameters by inverse modeling: Example lysimeter experiments. Soil Science Society of America Journal, 63(3): 501-509.

Abbaspour, K. C., 2007. User manual for SWAT-CUP, SWAT calibration and uncertainty analysis programs. Swiss Federal Institute of Aquatic Science and Technology, 103.

Abbaspour KC, Yang J., I Maximov, R Siber, K Bogner, J Mieleitner, J Zobrist, R. Srinivasan, 2007. Modelling hydrology and water quality in the prealpine/alpine Thur watershed using SWAT. Journal of Hydrology 333: 413–430.

Arabi, M., J.R. Frankenberger, B.A. Engel. and J.G. Arnold, 2008. Representation of agricultural conservation practices with SWAT. Hydrol. Process. 22: 3042-3055.

Bicknell, B. R., J. C. Imhoff, Jr. Kittle, J. L Donigian., A. S. Jr, R. C. Johanson, 1996. Hydrological Simulation Program-FORTRAN. User's Manual for Release 11. USEPA, Washington, DC.

Borah D. K., and M. Bera, 2003. Watershed-scale hydrologic and nonpoint-source pollution models: review of mathematical bases. Transactions of the ASAE, 46(6): 1553-1566.

Bottcher, A. B., B. J. Whiteley, A. I. James, and J. H. Hiscock, 2012. Watershed assessment model (WAM): Model use, calibration, and validation. Trans. ASABE 55(4): 1367-1383.

Douglas-Mankin, K. R., R.Srinivasan, and J. G. Arnold, 2010. Soil and Water Assessment Tool (SWAT) model: Current developments and applications. Trans. ASABE, 53(5): 1423-1431.

El-Nasr, A. A., J. G Arnold, J.Feyen, and Berlamont J. 2005. Modelling the hydrology of a catchment using a distributed and a semi-distributed model. Hydrological processes, 19(3), 573-587.

Faramarzi, M., K. C. Abbaspour, R. Schulin, and H.Yang, 2009. Modelling blue and green water resources availability in Iran. Hydrological Processes, 23(3): 486-501.

Gassman, P. W., M. R. Reyes, C. H. Green, and J. G. Arnold, 2007. The soil and water assessment tool: historical development, applications, and future research directions. Center for Agricultural and Rural Development, Iowa State University. 1211-1250.

Gassman, P. W., J. G. Arnold, R. Srinivasan, and M. Reyes, 2010. The worldwide use of the SWAT Model: Technological drivers, networking impacts, and simulation trends. In 2010 International SWAT Conference Proceedings, 21-24.

Hargreaves, G.H. and Samani Z.A. 1985. Estimating potential evapotranspiration. Tech. Note. J. Trrig. And Drain. Engr. 108(3): 225-230.

Im, S., K.Brannan, S. Mostaghimi and J. Cho, 2003. A comparison of SWAT and HSPF models for simulating hydrologic and water quality responses from an urbanizing watershed. In An ASAE Meeting Presentation, Paper (No. 032175).

Interstate Oil & Gas Compact Commission (IOGCC), 2010. Resolution on Hydraulic Fracturing 09.011, January 2009 Special Meeting. Downloaded at http://www.iogcc.state.ok.us/2009-resolutions

Jha, M. K. 2011. Evaluating hydrologic response of an agricultural watershed for watershed analysis. Water, 3(2): 604-617.

Moriasi, D. N., J. G. Arnold, M. W. Van Liew, R. L. Bingner, R. D. Harmel and T. L. Veith 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. Trans. ASABE, 50(3): 885-900.

OEPA, 2012. Ohio's Regulations: A Guide for Operators Drilling in the Marcellus and Utica Shales.

Rahman, M., T. Bolisetti and R. Balachandar, 2010. Effect of climate change on low-flow conditions in the Ruscom River watershed, Ontario. Transactions of the ASABE, 53(5): 1521-1532.

Saleh, A., and B. Du, 2004. Evaluation of SWAT and HSPF within BASINS program for the upper North Bosque River watershed in central Texas. Transactions of the ASAE, 47(4): 1039-1049.

Schuol, J., K. C. Abbaspour, R.Srinivasan and H.Yang, 2008. Estimation of freshwater availability in the West African sub-continent using the SWAT hydrologic model. Journal of Hydrology, 352(1): 30-49.

Singh, J., H. V. Knapp, J. G. Arnold, and M. Demissie, 2005. Hydrological modeling of the Iroquois river watershed using HSPF and SWAT Journal of the American Water Resources Association, 41(2): 343-360.

USEIA, 2011. International energy outlook 2011, coal, natural gas, electricity. Energy information administration, independent analysis and statistics.

USEPA, 2012. Study of the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources: Progress Report, U.S Environmental Protection Agency.

Van Liew, M. W., J. G. Arnold and J.D. Garbrecht, 2003. Hydrologic simulation on agricultural watersheds: Choosing between two models. Transactions of the ASAE, 46(6): 1539-1551.

Verma, A. K., M. K. Jha and R. K. Mahana, 2010. Evaluation of HEC-HMS and WEPP for simulating watershed runoff using remote sensing and geographical information system. Paddy and Water Environment, 8(2): 131-144.

USACE and DES, 2008. Model Selection and Recommendation Report for Central Oahu Watershed Study, Phase 2.

Xie, H. and Y. Lian, 2013. Uncertainty-based evaluation and comparison of SWAT and HSPF applications to the Illinois River Watershed. Journal of Hydrology, 481:119-131.

Yang, J., P. Reichert, K. C. Abbaspour, Xia J. and Yang H., 2008. Comparing Uncertainty analysis techniques for a SWAT application to the Chaohe Basin in China. Journal of Hydrology, 358(1): 1-23.

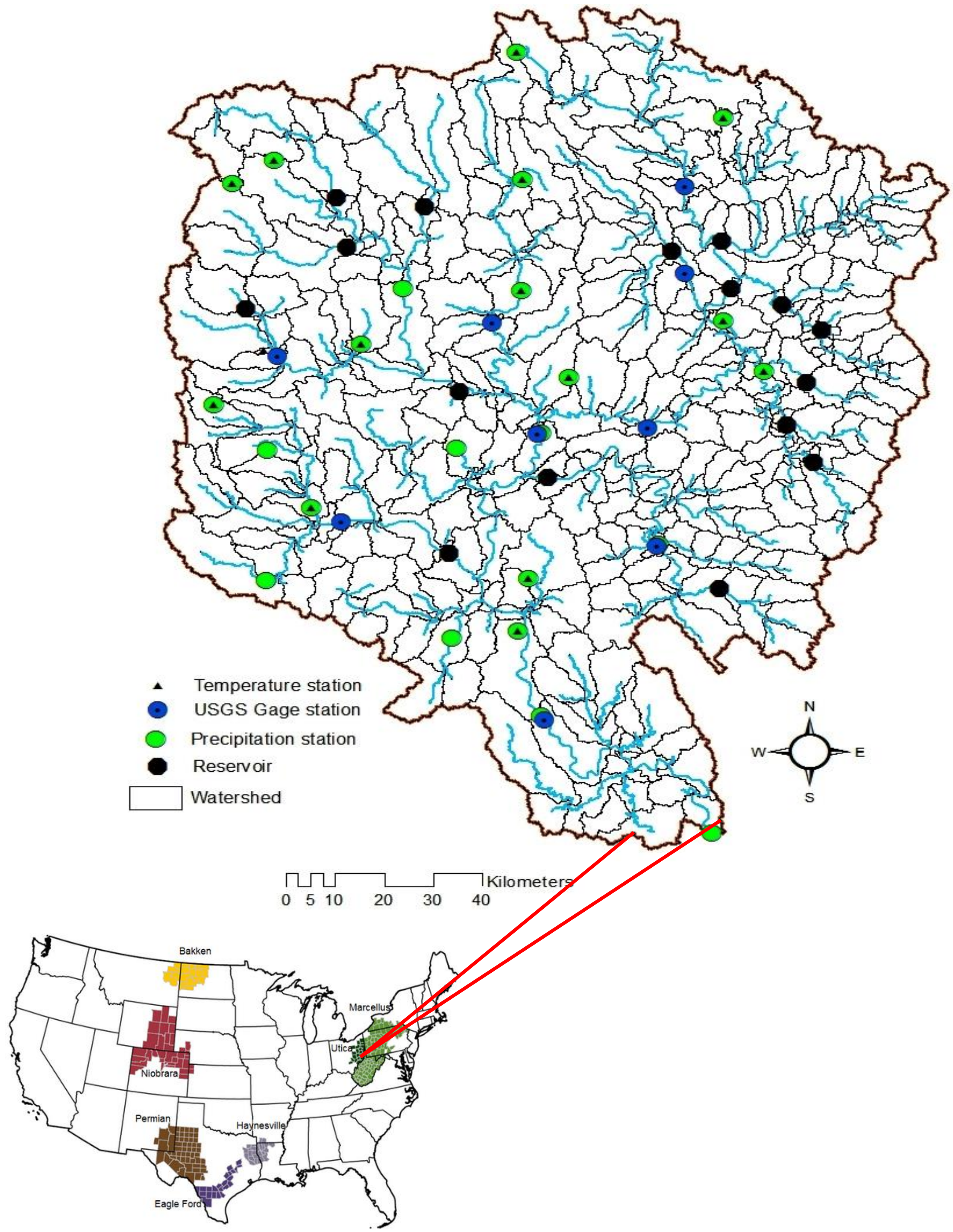


Figure 2-1: Location of climate stations including reservoir and USGS gage stations in the Muskingum watershed.

Source: US Energy Information Administration

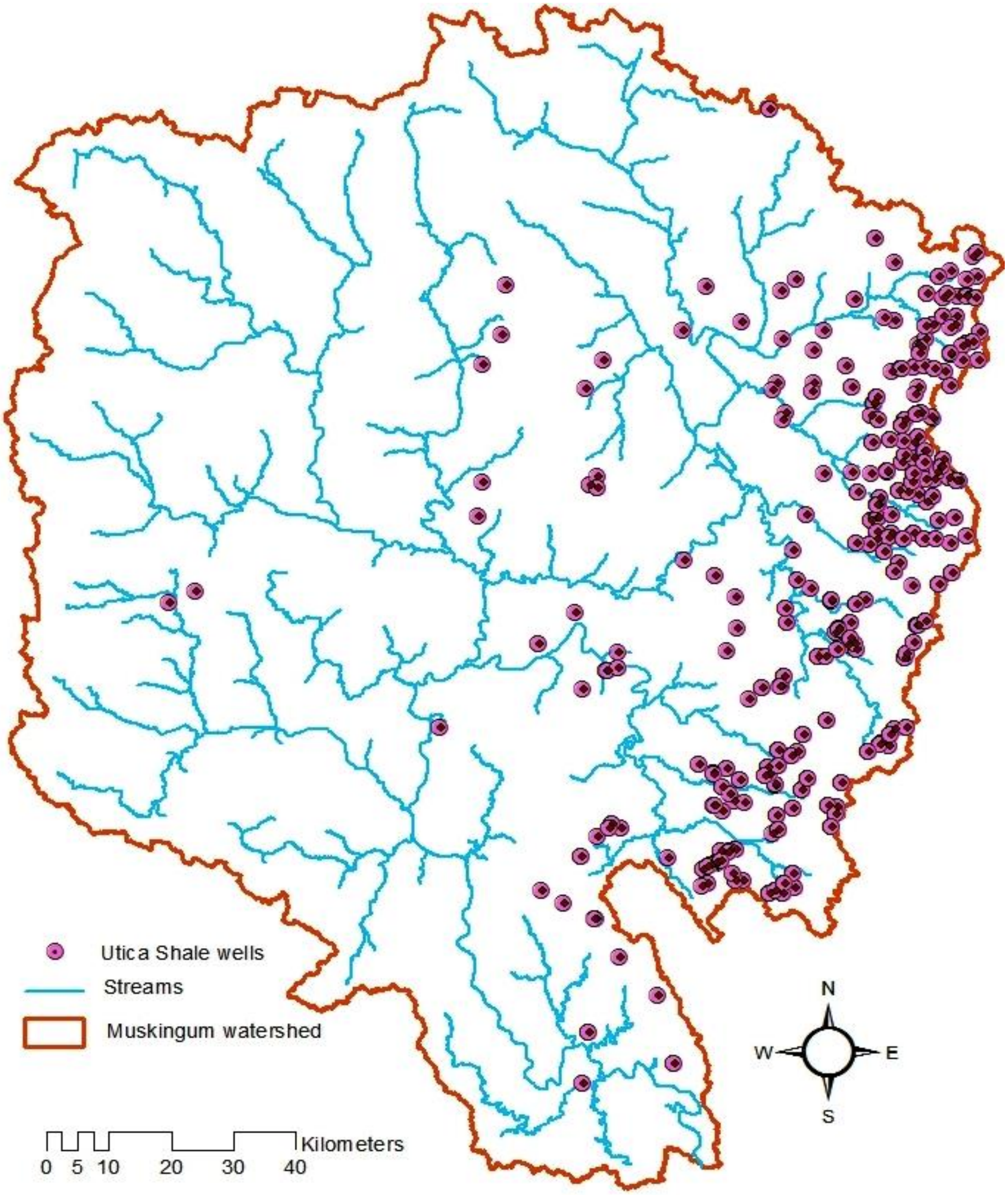


Figure 2-2: Utica shale wells in Ohio.

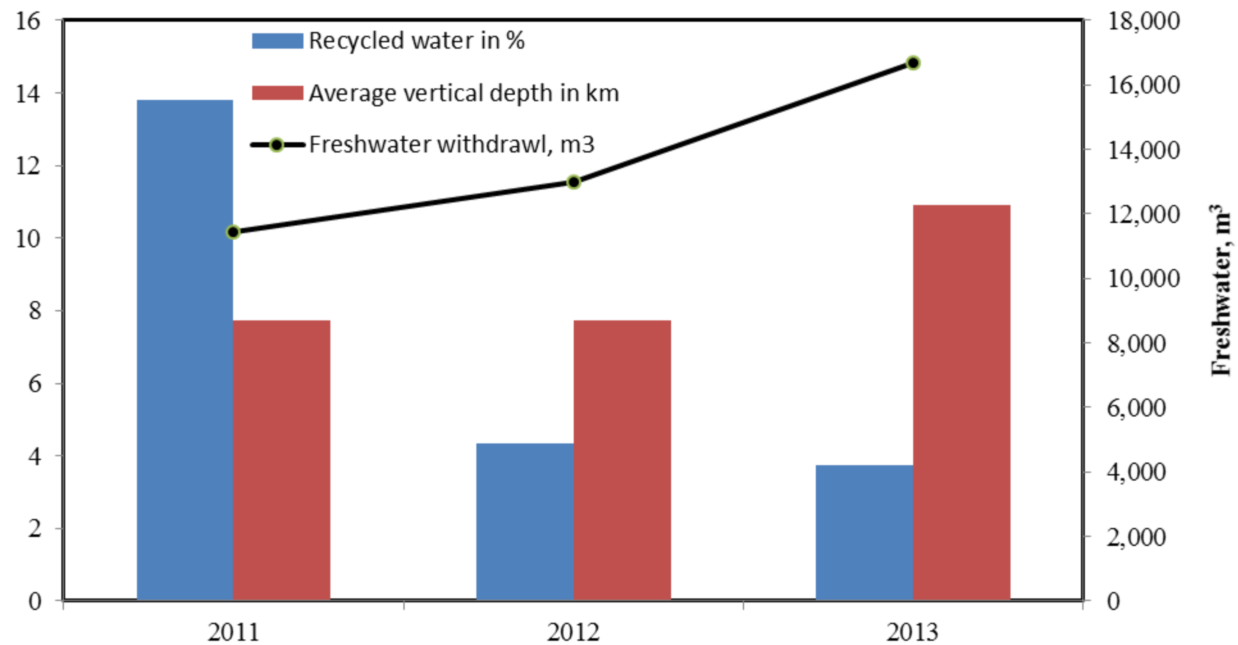
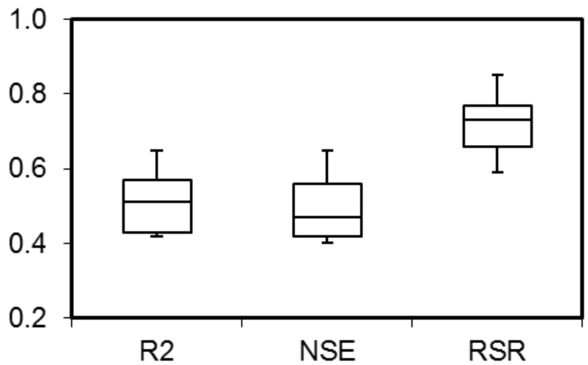
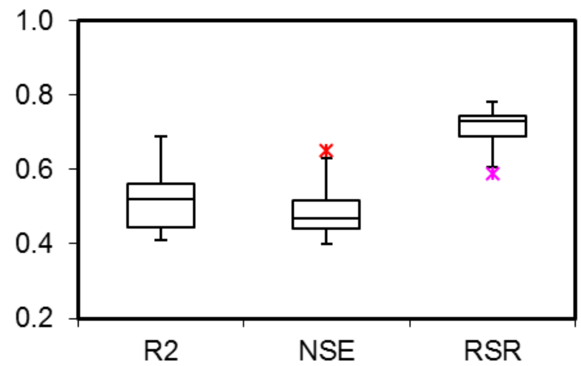


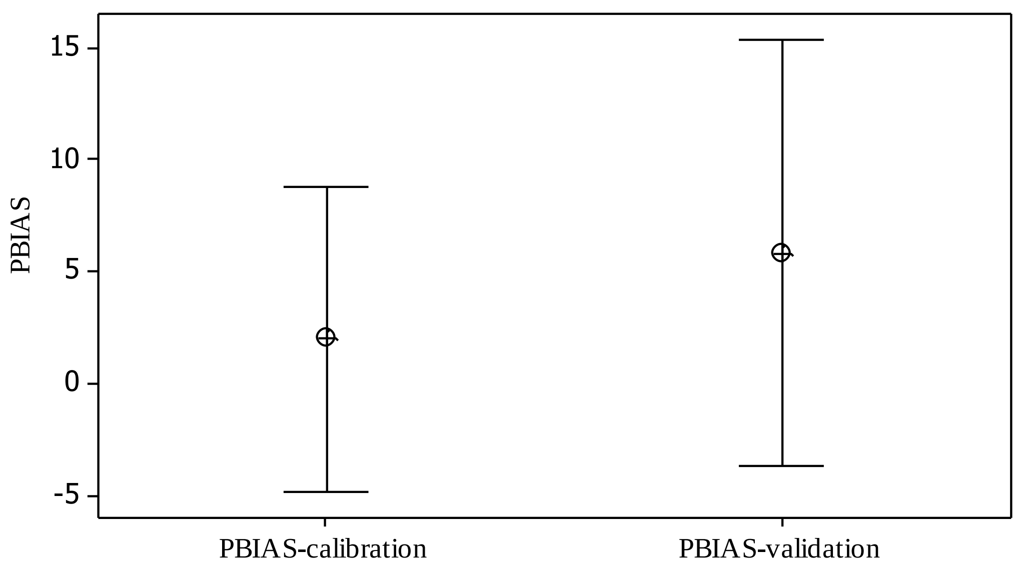
Figure 2-3: Fresh water withdrawal, average vertical depth and recycled water for hydraulic fracking in Muskingum watershed in various years



(a) Calibration



(b) Validation



Interval Plot of PBIAS (95% CI for the Mean)

Figure 2-4: Daily model statistics at 9 USGS gage stations during model calibration (a) and validation (b) period. The lower panel shows the interval plot of percentage bias (PBIAS) error for calibration and validation

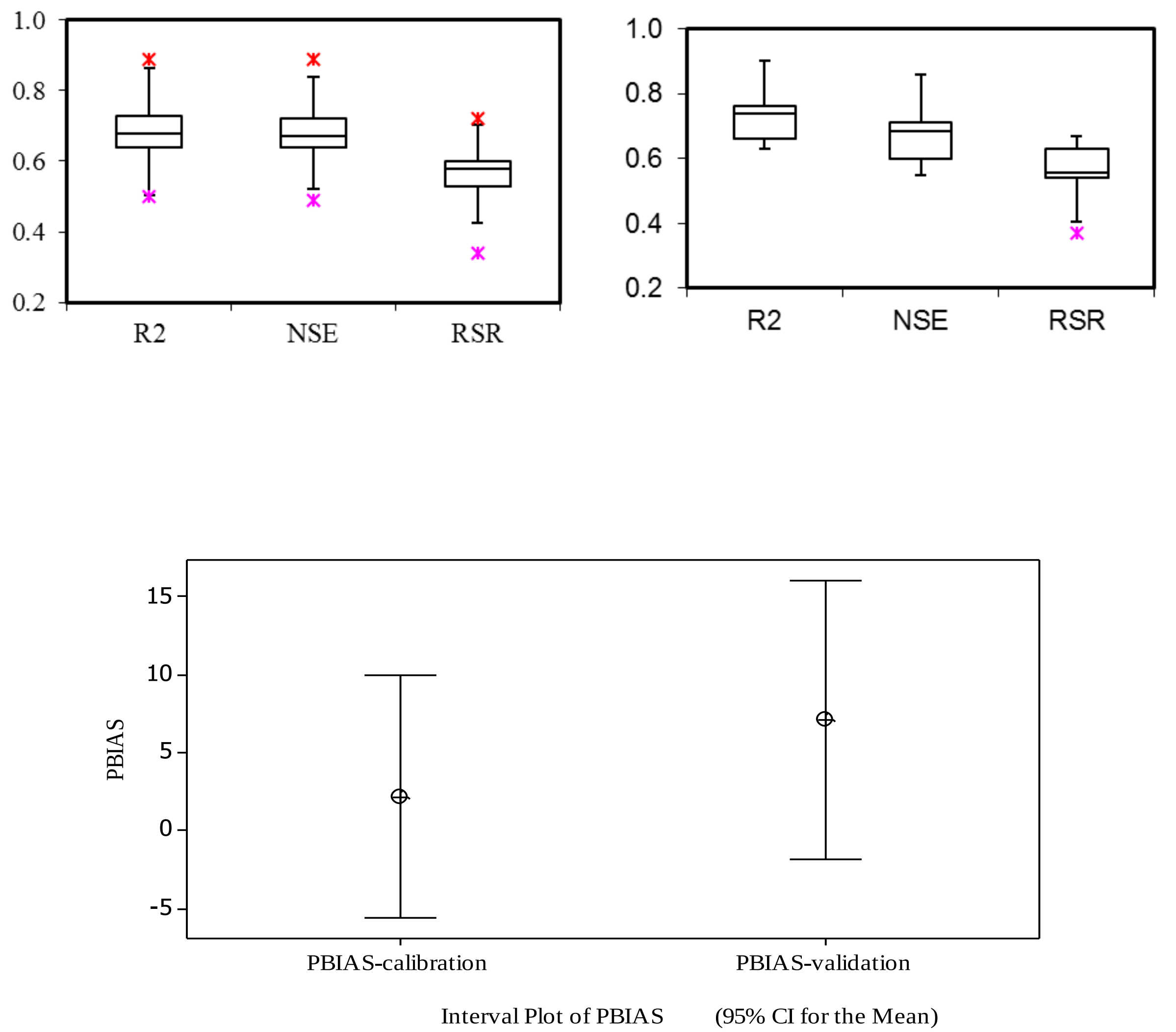


Figure 2-5: Monthly model statistics at 9 USGS gage stations during model calibration (a) and validation (b) period. The lower panel shows the interval plot of percentage bias (PBIAS) error for calibration and validation.

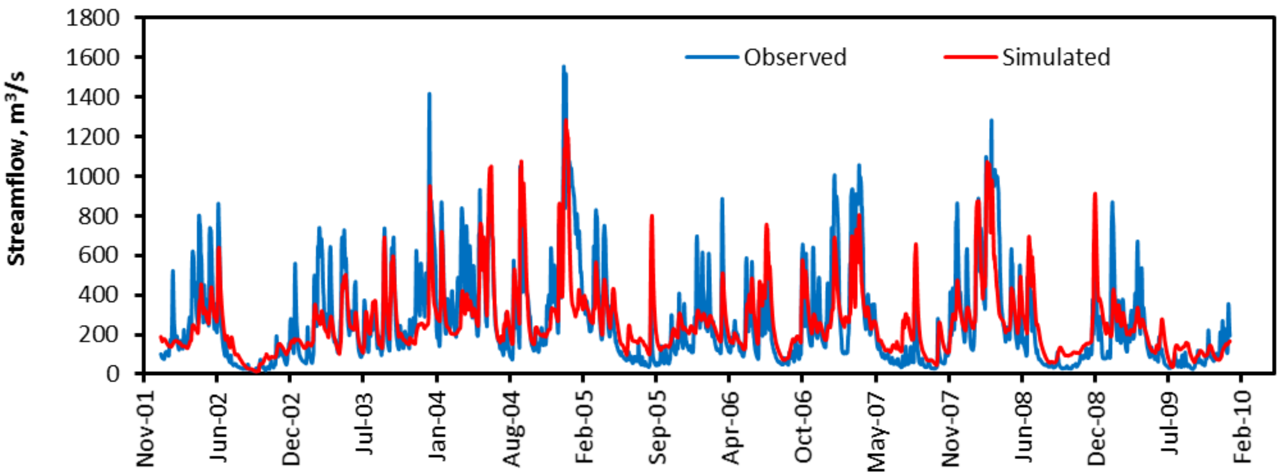


Figure 2-6: Daily streamflow calibration at watershed outlet (USGS gage 03150000).

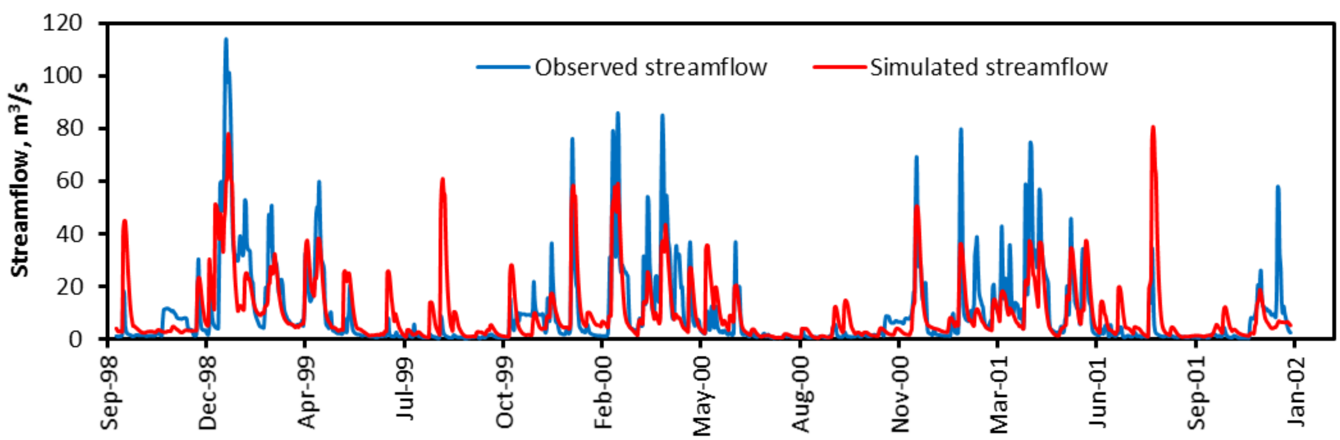


Figure 2-7: Daily streamflow validation at USGS gage 03142000 (the closest station to watershed outlet as outlet did not have long term record beyond year 2000).

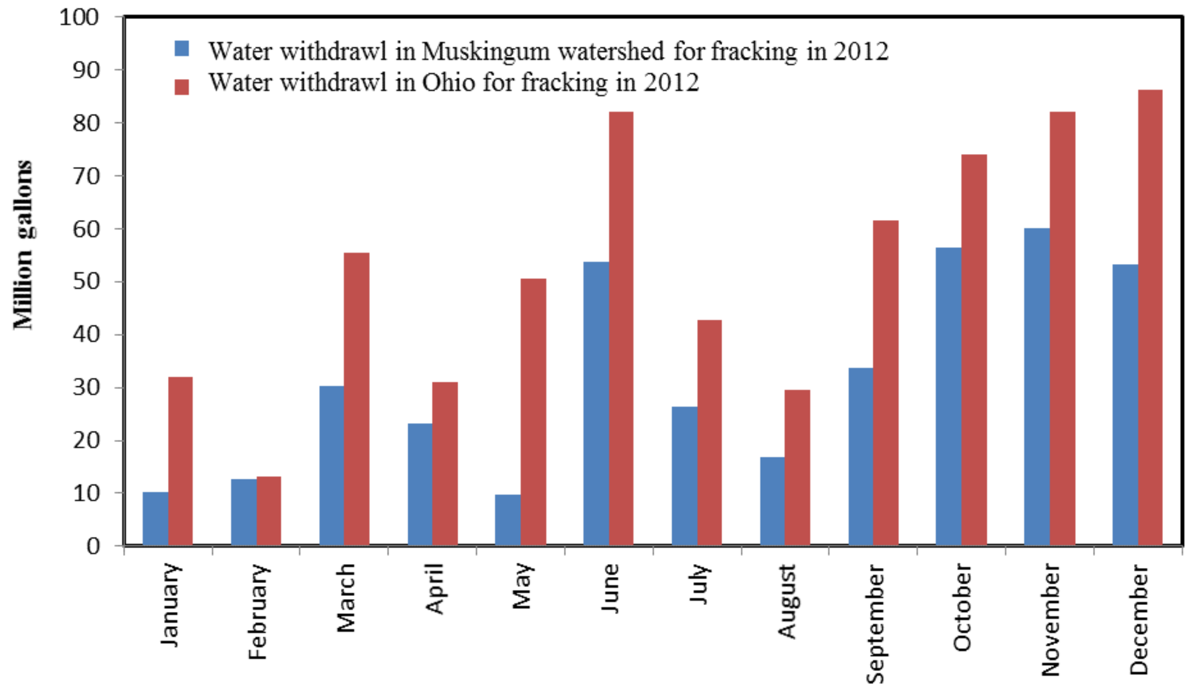


Figure 2-8: Water withdrawals for hydraulic fracturing in 2012 in Muskingum watershed and Ohio, respectively.

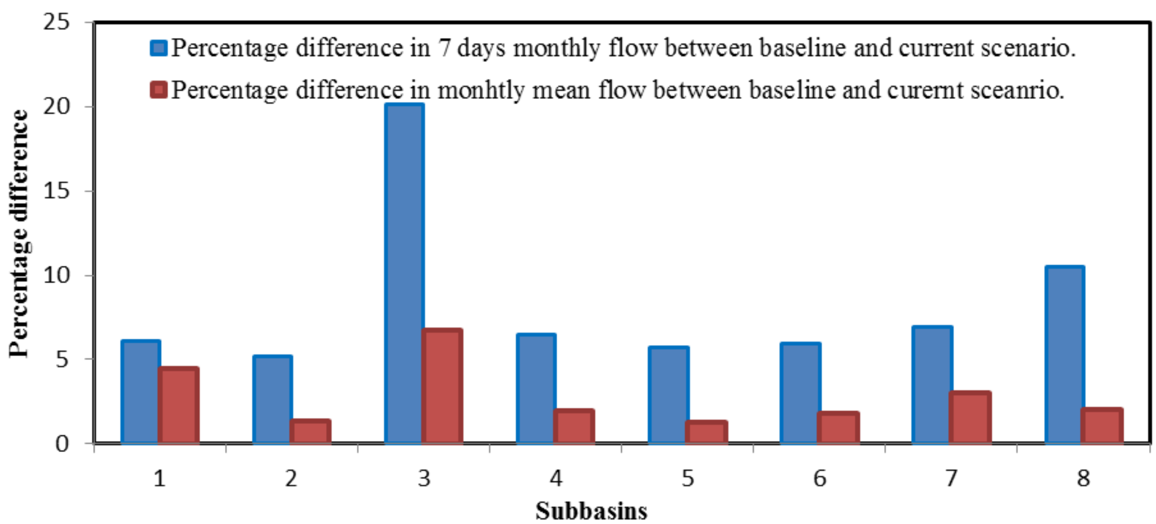


Figure 2-9: Percentage differences of 7 day minimum monthly flow and monthly mean between baseline and current fracking scenario on 8 affected subbasins for current period.

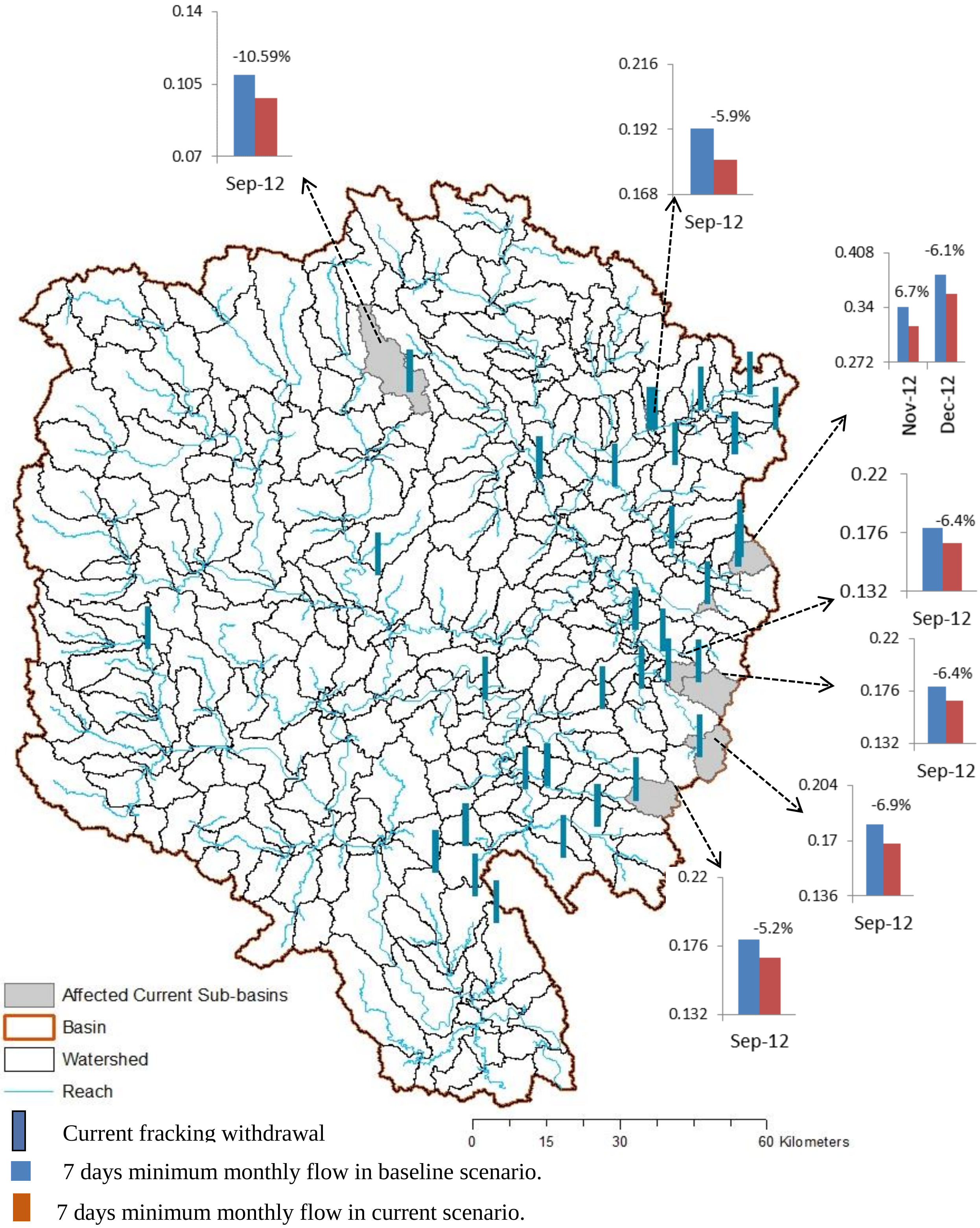


Figure 2-10: Impact of current fracking scenario on 7 day minimum monthly flow in Muskingum watershed.

Table 2-1: List of watershed models possibly used for hydraulic fracking study

Models	Model inputs for hydrologic analysis	Computational time scale	Options to incorporate point source/water withdrawal for hydraulic fracking	Internet source	Time scale
HSPF/LSPC	DEM/landuse/soil/precipitation/temperature and climate data	Daily/Sub-hourly	Yes	http://www.epa.gov/athens/wwqtsc/html/lspc.html	Continuous
SWAT	DEM/landuse/soil/precipitation/temperature and climate data	Daily/hourly	Yes	http://swat.tamu.edu/	Continuous
MIKE-SHE	DEM/landuse/soil/precipitation/temperature and climate data	Daily/hourly	Yes	http://www.mikepoweredbydhi.com/download/mike-by-dhi-2014	Continuous
APEX*	DEM/landuse/soil/precipitation/temperature and climate data	Daily	Yes	http://epicapex.tamu.edu/apex/	Continuous
WARMF	DEM/landuse/soil/precipitation/temperature and climate data	Daily/hourly	Yes	http://www.epa.gov/athens/wwqtsc/html/warmf.html	Continuous
HEC-HMS	DEM/landuse/soil/precipitation/temperature and climate data	Daily/hourly	Yes	http://www.hec.usace.army.mil/software/hec-hms/downloads.aspx	Continuous/event-based
WAM	DEM/landuse/soil/precipitation/temperature and climate data	Daily/hourly	Yes	http://www.epa.gov/athens/wwqtsc/html/wamview.html	Continuous

**Some components are available in hourly scale as well*

Table 2-2: Data and sources used for the study.

Data Type	Data	Source
GIS	DEM of 30-meter resolution	USGS
	Land cover datasets, 2006	NLCD
	Soil data	USDA, STATSGO
Climate data	Precipitation and temperatures	NCDC
Hydrology	Streamflows	USGS
	Lake and reservoir outflow	USACE
Water Use (Surface and ground water)	Water use for irrigation, public, power, mineral extraction, industries and golf course	ODNR Ohio EPA
Major point sources	Flow discharge	Ohio EPA
Information related with hydraulic fracking	hydraulic fracking information including sources of drilling water	ODNR
	Drilling water estimate per well and future drilling trend	FracFocus

Table 2-3: Major reservoirs in the Muskingum watershed.

Watershed	Reservoirs	Locations	Drainage area (km^2)
Tuscarawas River watershed	Leesville	McGuire Creek	124.32
	Atwood	Indian Fork	181.3
	Tappan	Little Stillwater	183.89
	Clendening	Stillwater Creek	181.3
	Beach City	Sugar Creek	776.97
	Piedmont	Stillwater Creek	217.56
Walhonding River watershed	Charles Mill	Black Fork	559.44
	Pleasant Hill	Clear Fork	515.41
	North Branch of Kokosing	North Branch	116.5
Will Creek watershed	Wills Creek	Mainstem	1872.6
	Senecaville	Seneca Fork	313.39
Licking River watershed	Dillion	Mainstream	1937.24

Chapter 3. Scenario Analysis for Assessing the Impact of Hydraulic Fracturing on Stream Low Flows Using SWAT Model

Abstract

Scientists and water users are concerned about the potential impact on water resources, particularly during low-flow periods, due to fresh water withdrawals for unconventional oil and gas development (hydraulic fracturing, or "fracking"). Most water management decisions are based on the hydrologic or biologic conditions, which are estimated using long term historical records of low-flow periods without accounting for water withdrawals for hydraulic fracking. This raises a question as to whether current policies of point source permitting, which rely on low-flow conditions, are appropriate given the current trends of water demand for hydraulic fracking. Moreover, additional water withdrawals from surface water during low-flow periods may pose a threat to the sustainability of water supplies and aquatic ecosystems, particularly during drought years. The objective of this paper is to assess the potential impact of hydraulic fracturing on water resources in the Muskingum watershed of Eastern Ohio, USA, especially due to the trend of increased withdrawals for hydraulic fracking during drought years. The Statistical Downscaling Model (SDSM) was used to generate thirty years of plausible future daily weather series in order to capture the possible dry periods. The generated data were incorporated in the Soil and Water Assessment Tool (SWAT) to examine the level of impact due to fracking at various scales. Analyses showed that water withdrawal due to hydraulic fracking had noticeable impact, especially during low-flow periods. Clear change in the seven day minimum flows was detected among baseline, current and future scenarios when the worst case scenario was implemented. The flow alteration in hydrologically-based (*7Q10*, i.e. 7-day 10-year low flow) or biologically-based (*4B3* and *1B3*) design flows due to hydraulic fracking increased with decrease

in the drainage area, indicating that the relative impact may not be as great for higher order streams. Nevertheless, change in the annual mean flow was limited to 10%.

Keywords: Hydraulic fracturing (fracking), SWAT, SDSM, Low-flow, drought

Introduction

Shale gas production in the United States is projected to increase by threefold, covering a significant portion of all natural gas produced by 2035 (USEIA 2011). Development of unconventional shale gas is technologically enabled by a key technique called hydraulic fracturing (fracking) (Hubbert and Willis 1972, Yew and Weng 2014). However, a large amount of water (26,500 m^3) is required for fracking, which has attracted the attention of public and regulatory agencies (Craig 2012, Nicot et al. 2012, Rahm and Riha 2012). Scientists and water users are concerned about the extent of potential impacts of this water consumption, especially from surface water (Entrekin et al. 2011), at different spatial and temporal scales. The impact could be significant if further consideration is not given to the timing, location and volume of water withdrawal for fracking, especially during low-flow periods (USEPA 2011c, Cothren et al. 2013).

The water quality standards issued by the Federal and State agencies are developed based on low-flow conditions (Sharma et al. 2012, Saunders et al. 2004). For example, the Environmental Protection Agency (EPA) and State agencies issue National Pollution Discharge Elimination System (NPDES) permit limits based on hydrologically-based design flows (e.g. *7Q10*, the 7-day 10-year low flow) and biologically-based design flows (*4B3, 1B3*). The hydrologically-based design flow is the extreme low-flow calculated exclusively based on the hydrologic records considering the lowest flow from each year, whereas the biologically-based design flow is computed considering all the low-flow events over the period of record (Eslamian 2014). These regulatory low-flow criteria are developed based on statistical analysis of long term historical stream flows records without anticipating water withdrawal for fracking. This raises a question as to whether the permit conditions developed for low-flow periods are adequate to protect water quality in the current and future conditions of fracking. Overestimation of low flows may lead to underestimation of pollutant levels below wastewater discharges; therefore, proper consideration of potential water withdrawals for fracking is essential in the permitting process. In addition, reservoirs and streams used for water supply purposes will be at critical stages during low-flow (drought) periods, which will be further reduced due to the sudden withdrawal for fracking. Therefore, water use for fracking not only reduces water quality, by reducing the assimilating capacities of the stream for pollutants, but also affects water availability for various water supply purposes.

These issues are very common in the Muskingum watershed of the Eastern Ohio, leading to the critical challenges for water resources sustainability. Several lakes (e.g. Seneca) and ground water resources are extensively used for water supply in this region. Almost one million residents of the watershed rely on private wells (Angle et al. 2001). However, groundwater use for fracking is very nominal (1%), since stream water is more convenient for use in fracking industries. Also, a significant amount of water is used for fracking and only a limited amount of water is returned to streams as flow back. In this context, detailed analysis is needed to evaluate the impact of hydraulic fracking on assimilating capacities of streams for NPDES permitting and water resources availability during low-flow periods.

Few studies have been conducted related with the impact of fracking on stream low-flows. The United States Environmental Protection Agency (USEPA) has conducted a study to evaluate the potential impact of hydraulic fracturing on drinking water resources in the Upper Colorado River Basin (USEPA 2012). Similarly, research was conducted in the Fayetteville Shale play to assess the impact on flow regime and on the environmental flow criteria of the stream (Cothren et al. 2013). This study demonstrated the impact of hydraulic fracking on environmental flow components, especially on small scales. While these studies addressed the impact of hydraulic fracking on stream flows in general, extensive analysis of stream low flows due to water withdrawal for hydraulic fracking has not been conducted yet.

In our earlier study (Sharma et al. 2015), we explored various potential watershed models to conduct a simulation study under fracking conditions and reported that the Soil and Water Assessment Tool (SWAT) could be an appropriate tool for this purpose. We conducted a multi-site model calibration and validation using SWAT in the study watersheds with satisfactory model performance, and reported several modeling challenges, opportunities and issues in a simulation study. Our previous study indicated a modest effect on watershed hydrology due to the current rate of water withdrawal for fracking. The purpose of the present study was to evaluate the potential impact on water resources of extreme conditions (climate, rainfall, and fracking withdrawals) in the future by developing various scenarios. In order to develop various water acquisition scenarios, weather and scenario generator tools of the Statistical Downscaling Model (SDSM) (Wilby et al. 2013) were adopted to generate possible climate and precipitation data and then integrated with the SWAT model to develop future scenarios. In the next step, all the scenarios were analyzed at different spatial and temporal scales to investigate the potential impact of water withdrawals on water budgets during low-flow periods.

Theoretical Background

Hydrologically-Based Design Flow

This design flow was orginally introduced by the United States Geological Survey (USGS) and has been popularly used by various States in the US for National Pollutant Discharge Elimination System permitting (NPDES) (Wiley 2006). The xQy hydrologically-based design flow is computed as the x-day consecutive average low flow with a return period of y-year. The lowest x-day flow from each year of the record period is determined. The distribution of these values is plotted and the y-year value is determined either empirically if the record is long enough or using a statistical law (Pyrce 2004). Hence xQy intends to characterize the lowest flows of each year. For example, *7Q10* refers to the minimum consecutive average 7-day low flow of each year with an expected return period of 10 years.

Biologically-Based Design Flow

This design flow was orginally introduced by the United States Environmental Protection Agency (USEPA) (Rossman and EPA 1990). This method is different from the hydrologically-based design flow as it uses the *n* lowest flow events within a given period of record of *n* years, irrespective of the individual years. It means that several lowest flow events from a given year may be incorporated for statistical analysis in biologically-based design flow computation, whereas there may be no event from other years. While the hydrologically-based design flows evaluate the risk of being below a threshold one year out of y, the biologically-based design flow intends to quantify the cumulative effects of consecutive low-flow events on aquatic life. *4B3* and *1B3* are the 4-day and 1-day average biologically-based flows, respectively, which can be expected once in three years.

Hydraulic Fracking

Hydraulic fracturing, introduced in the 1940s (Montgomery and Smith 2010), is the technique of injecting large amount of water mixed with sand and chemicals at high pressure, which fractures rock underground at great depth and releases the gas (Beaver 2014). Drilling requires approximately 2 to 4 weeks duration and the expected life of a well is typically 20 to 50 years. The water is typically needed for a few days to a week during the drilling process, depending upon the site conditions. Water is withdrawn from surface and groundwater sources, treated water from a treatment plant, and recycled water from the flow back and produced water (Gregory et al. 2011). Water usage per well varies depending upon the type of shale and its thickness, configuration and dimension of the well (e.g., length, depth, horizontal or vertical orientation, multiple leg or single leg) and fracturing operation. Water use per well is estimated to range from 25 m^3 for coalbed methane production to 49,200 m^3 for shale gas production (GWPC and ALL Consultant 2009, Nicot et al. 2012). Similarly, the fracturing process for a shale gas well requires 8,700 m^3 to 94,600 m^3 of water per well (USEPA 2011a) and an additional 151–3,790 m^3 of water is required for drilling vertically per well (GWPC and ALL Consultant 2009). The Marcellus shale data in Pennsylvania shows the water required is from 7,570 to 15,100 m^3 per well (Gregory et al. 2011, Satterfield et al. 2008). In general, 15,100 to 22,700 m^3 of water is commonly needed to frack a single Marcellus or Utica shale well (OEPA 2012).

Soil and Water Assessment Tool (SWAT)

The SWAT is a physically-based watershed model, which is developed to predict the long term impact of watershed management in terms of hydrologic and water quality response of large watersheds (Arnold et al. 1998). The SWAT simulates different physical and hydrological processes across river watersheds. The model is widely used in different regions of the world for surface water and ground water modeling, as described in many peer reviewed publications (Gassman et al. 2010). SWAT simulates ground water by partitioning soil profiles into three layers: soil layers; shallow aquifer; and deep aquifer. The shallow aquifer, which lies between the soil layers and deep aquifer, closely resembles a reservoir. Surface water and ground water modeling procedures in SWAT are described in articles by Vazquez-Amábile and Engel (2005) and Arnold et al. (1996). Similarly, SWAT can simulate reservoirs by calculating the water balance, incorporating inflow, outflow, rainfall, evaporation, any seepage and diversions. There are three options available to compute the outflow from the reservoir (Neitsch et al. 2005): i) input the observed outflow; ii) specify the outflow release rate; or iii) specify the monthly fixed volume of the reservoir.

Initial inputs to SWAT include geographical information such as digital elevation model (DEM) to spatially delineate the watershed in terms of different sub-watersheds. Further, land use, soil and slope information are utilized to subdivide the sub-watersheds into smaller hydrologic response units (HRU's), which are composed of similar land use, soil and management characteristics.

Statistical Downscaling Model (SDSM)

The SDSM is a climate change scenario generator used for risk assessment and climate studies (Wilby et al. 2002). The SDSM uses tools such as the stochastic weather generator and regression based downscaling technique as a means for weather generation (Wilby et al. 2002). The weather generator is used to generate synthetic data of weather such as precipitation, and maximum and minimum temperatures. Precipitation is simulated based on the occurrence of wet or dry periods, and on the amount of precipitation and temperature. The occurrence is modeled by a Markov chain method and the amount is sampled randomly from a suitable distribution such as a Gamma distribution. The weather generator has been used in many studies for infilling missing data and matching local climate information based on predictor variables. Five main steps were followed to generate plausible dry period precipitation through the SDSM: 1) identification of predictors and predictands; 2) SDSM model calibration; 3) parameter file generation; 4) incorporating missing data using the weather generator; and 5) generating future dry period precipitation using the scenario generator tool.

Materials and Methodology

Study Area

This research was conducted on the Muskingum watershed (Figure 3-1), which is located in the eastern part of the Ohio. It covers an area of more than 20,720 km^2, which is nearly 20% of the State. Muskingum is the largest river in the watershed, which originates at the union of the Tuscarawas and Walhonding Rivers near Coshocton, and eventually drains into the Ohio River at Marietta. The main tributaries of this river are the Tuscarawas, Walhonding, Licking Rivers and Wills Creek. The Muskingum watershed is a HUC-4 watershed (0504), which is subdivided into six HUC-8 watersheds: Licking (05040006), Walhonding (05040003), Mohican (05040002), Tuscarawas (05040001), Wills (05040005) and Muskingum (05040004). The Muskingum watershed contains nearly 19% of Ohio's wetlands and 28% of the state's lakes and reservoirs (Auch 2013).

Utica Shale in eastern Ohio has great potential for the production of natural gas and oil. Currently, most of the unconventional natural gas wells in Ohio lie in the Muskingum watershed (Figure 3-2). Recently, several drilling companies are advancing to Ohio for oil and gas development and drilling has increased tremendously on this watershed. Most of the wells are concentrated on the eastern portion of watershed, the Tuscarawas sub-watershed, which covers the area of 6,327 km^2 within the Muskingum watershed (Figure 3-3). This sub-watershed covers all or partial areas of the thirteen counties of Ohio. The northern portion of this sub-watershed is significantly covered by industrial and urban land uses whereas the southern portion is dominated by forest cover. Additionally, there are a number of reservoirs in the eastern part of the watershed that are used for municipal water supplies.

SWAT Model Input

The DEM of 30 m resolution was downloaded from USGS National Elevation Dataset in order to delineate stream networks using ArcGIS, resulting in 406 sub-basins. Similarly, land use data of 30 m resolution were downloaded from the National Land Cover Database 2006 (NLCD 2006) to best represent the land use pattern within the calibration period (from 2002 to 2009) of this model. The watershed was comprised of forest (47%), agriculture land with row crops (23%), hay (19%), and urban areas (10%). The remaining 1% of land use includes industrial area, water, range grass and forested wetlands. The existing 12 reservoirs were spatially located manually at proper locations of the watershed with reference to the stream outlet. The soil data taken from the State Soil Geographic dataset (STATSGO) (USDA 1991) were included as a default in the SWAT with a map at 1:250,000 scale. The appropriate numbers of HRUs (6,176) were obtained by assigning multiple HRUs for each subbasin in order to account in detail for the complexity of the landscape. These large numbers of HRUs were achieved despite eliminating minor land uses, soils and slopes by selecting thresholds of 5%, 15% and 15%, respectively. The large numbers of HRUs were highly beneficial for accurate prediction of streamflows (Arnold et al. 1996).

Seventeen years of climate data, including precipitation and the maximum and minimum temperature were downloaded from the National Climatic Data Center (NCDC) website. Altogether, 23 precipitation stations and 19 temperature gauge stations were located within the watershed (Figure 3-1). The remaining meteorological time series inputs such as solar radiation, wind speed, and relative humidity were available from the SWAT's built-in weather generator. The daily streamflow data needed for model calibration and validation were available from the USGS website for 9 USGS gauging stations from 1993 to 2009 (Figure 3-1). The reservoir daily mean outflows data were available from the US Army Corps of Engineers (USACE) for the same duration.

Data on point source discharges greater than 1,890 m^3/d were collected from the OEPA and used as model inputs before calibrating the model. Similarly, other water use information was obtained from the ODNR in order to adequately represent the watershed conditions before model calibration and validation. The consumption of water for hydraulic fracking was more significant than the consumption of water for agriculture during 2012. The surface water withdrawal in Muskingum watershed in 2012 was 334 m^3/d for hydraulic fracking, 308 m^3/d for golf course irrigation, 220 m^3/d for agriculture and 716 m^3/d for industry. Ground water withdrawal for hydraulic fracking in the same year was small (1%) compared to surface water withdrawal.

Water withdrawal information was verified from ODNR and OEPA. Similarly, the locations of oil and natural gas wells and sources for freshwater withdrawals greater than 378 m^3/d, were collected from ODNR. The completed well data, freshwater required per well, and recycled water quantities were obtained from FracFocus, which is the national hydraulic fracturing chemical registry. A summary of input data required, sources and format is presented in Table 1.

Model Calibration and Validation

In order to reduce the uncertainty in model prediction, a hydrologic model needs to be properly calibrated and validated (Engel et al. 2007). For this, SWAT-CUP (Abbaspour et al. 2007) was selected to calibrate the model parameters using streamflow time series at various locations using SUFI-2 algorithm. The efficiency of SUFI-2 for large-scale modeling studies is suggested by Yang et al. (2008). Since model calibration is an iterative process of establishing the best agreement between simulated and observed data, 21 model parameters (not shown), as discussed in several earlier studies (Abbaspour et al. 1999, Abbaspour et al. 2007, Faramarzi et al. 2009, Schuol et al. 2008a, Yang et al. 2008), were selected for model calibration and validation at various locations of the watershed. Since sufficient model parameters were chosen for model calibration, the model predictions were assumed to be reliable.

The model simulations were run for 15 years using observed data at 9 USGS stream gauge stations (Figure 3-1) from 1995 to 2009 including the calibration and validation period. A warm up period of two years was used in order to minimize the effect of initial unknown parameters and stabilize the hydrologic component of the model. The model was then calibrated in a daily time scale on 9 different USGS stream gauges from 2002 to 2009. In the next step, validation of the model was performed from 1995 to 2001 using statistical criteria measuring the goodness of fit such as the coefficient of determination (R^2) (Krause et al. 2005), Nash-Sutcliffe coefficient of Efficiency (*NSE*) (Nash and Sutcliffe 1970), Percent Bias (*PBIAS*). Also, we compared the simulated and observed flows for less than 75 percentile low flows. For better assessment of the model, users can perform the full split sample test, inverting the roles of the calibration and validation periods. A detailed description of the model evaluation criteria is not included in this paper. Readers can refer to Sharma et al. (2015) for additional information on model evaluation procedures.

Scenario Analysis

The calibrated and validated SWAT model was integrated with water use and point sources data in order to develop theoretically realistic scenarios with different rates of fracking development. The baseline, current and future scenarios were developed to assess the impact on water resources under various levels of fracking. The baseline scenario referred to the watershed conditions for 2012, with water consumption including public water supply, domestic, industrial and other water uses for irrigation, livestock, mining, power plants and point sources, but no hydraulic fracturing water use. The current scenario utilized watershed conditions in the year 2012, including the fracking withdrawal rate of 2012 in addition to other consumptive uses. As the fracking boom was just beginning in 2012, we only had three years of drilling data available for this study. The future scenario projected the current rate of development of natural gas in the Muskingum watershed up to year 2030, based on the recent drilling trends. However, drilling water demands were the only parameters changed and other consumptive use was considered constant in all the scenarios. The future scenario was developed into two separate scenarios with respect to temporal scale. The "future scenario on current climate period" was developed by simulating streamflows using the projected hydraulic fracking conditions of year 2030 but using current climate conditions (2012). Similarly, the "future scenario on future climate period" referred to the projected hydraulic fracking condition of the year 2030 simulated over a climate period of 30 years, which was generated based on historical climate records. Monthly consumptive water use was provided from the water use input file based on the removal of water from stream reaches, shallow aquifers, and reservoirs within the subbasin.

Since hydraulic fracking in Ohio is predominantly occurring in the eastern part of the state, detailed analysis for the future scenario was conducted for the Tuscarawas sub-watershed, as it covers the major eastern portion of the Muskingum watershed. Future fracking wells in the watershed (Figure 3-4) were estimated based on the past three years of drilling trends in Ohio. Even though ODNR data reports 865 wells were drilled in Ohio, drilling information on only 517 wells were available from FracFocus. Therefore, projections up to 2030 were based on actual wells drilled from FracFocus. The future scenario was adopted with fracking wells increasing by

224 wells per month in Ohio by 2030. Even though the future trend of hydraulic fracking is uncertain and depends on many political and socio-economic constraints, we assumed that the extreme increase in fracking assumed in the future scenario would provide an idea of the maximum environmental impact on stream low-flows for policy makers. This study was motivated by the growing concern of communities and scientists about water resources availability in this region. The extreme worst case scenario analyzed for the future period was particularly important as an indicator of potential conflicts among water uses.

A similar trend of hydraulic fracking that was experienced in Ohio was assumed in the Tuscarawas subwatershed. Subbasins with a minimum net area of 2 km^2, after eliminating residential and water bodies, were used for analysis. This resulted in 149 out of 168 potential subbasins to study in this subwatershed. The water withdrawal for each well was estimated based on water use trends in the Muskingum watershed for 2012 and 2013, available from FracFocus. The water use for each well in the Muskingum watershed was approximately 12,870 m^3 in 2012 and 16,656 m^3 in 2013 (Table 2). Based on this increasing trend, the freshwater withdrawal for each well was assumed to be 17,034 m^3 in 2030. The existing trends indicate a decreasing use of recycled water from 2012 (4.3%) to 2013 (3.7%); hence, recycled water for 2030 was considered simply as 4% of initial withdrawals. Water was assumed to be taken from multiple nearby sources such as the nearest streams and reservoir in the watershed, as suggested by Arthur et al. (2010). For the temporal distribution of the water use for fracking, especially 5 to 7 days are allowed for fracturing the well or using the freshwater (Sullivan et al. 2013). Generally, density of 4,047 m^2 are considered for the development of shale wells (Myers 2009, Robbins 2013), and this information was used to determine the maximum possible limit of the wells in the watershed. Therefore according to the distribution of projected numbers of 2030 wells on 149 subbasins with consideration of 17,034 m^3 of freshwater and 4% recycled water for each well for consecutive 7 days which was equally distributed for 149 subbasins of Tuscarawas watershed. Later, this projected trend was integrated with 30 years of plausible climate data in the SWAT model to simulate future scenarios. The future possible climate data was generated using a statistical downscaling model (SDSM) based on historical climate records of the region.

Developing Future Climate

In this research, the SDSM tool was utilized to generate the future climate data by establishing the quantitative relationship between local surface variables (predictands) and large scale variables (predictors). The National Center for Atmospheric Research (NCAR) and the National Centers for Environmental Prediction (NCEP) have developed more than 50 years of global analysis of atmospheric components (Kalnay et al. 1996). Therefore, this reanalysis data was selected to recover the missing measured data in the data assimilation system and make consistent climate variables throughout the reanalysis period. Large scale predictors including mean temperatures, vorticity at surface and 850 hPa, zonal velocity and many more were selected from predictors obtained from NCEP/NCAR reanalysis data for the period of 1961-1990, based on regression techniques (Table 3). The observed data were used from 1961 to 1975 to develop the regression model (calibration) and then regression weights produced a parameter file to validate for the period from 1976 to 1990. This process was repeated for all the precipitation stations. Once the missing data (1961-1990) were infilled using a weather generator tool, the scenario generator from the SDSM was applied to generate precipitation. In order to generate a possible set of dry period precipitations, we used the historical set of observed precipitation and temperature data in the respective stations (23 precipitation and 11 temperature stations). The SDSM model was used to generate a possible set of future precipitation for a 30 year period. Since our intention was to run the model for a worst case, low flow scenario, we generated the 25th percentile precipitation datasets within the watershed at all stations.

Results

Model Simulation

The model calibration and validation were performed on a daily and monthly time scale. The model performance was satisfactory during calibration and validation period with reasonable accuracy, which was assessed through a visual inspection and statistical criteria. The daily and monthly statistical criteria measuring the performance of the model, such as *NSE*, R^2 and *PBIAS* are listed in Table 4 and Table 5, respectively. Analysis showed that *NSE* and *PBIAS* were satisfactory and mostly above the recommended ranges (Moriasi et al. 2007) (*NSE* > 0.5 and *PBIAS* ±25%) except at a few stations, especially for monthly simulation. The *NSE* values varied from 0.40 to 0.65, and 0.4 to 0.65 for daily streamflow calibration and validation, respectively (Table 4). Similarly, the *NSE* varied from 0.49 to 0.89 for monthly streamflow calibration, and 0.55 to 0.86 for monthly streamflow validation (Table 5). The simulated streamflows slightly underestimated the daily and monthly peaks, which is consistent with previous findings (Bieger et al. 2014, Santhi et al. 2014). The mean observed and simulated low flows lower than the 75th percentile were compared with *PBIAS* (17%) and R^2 (0.78) (not shown). Overall, the model captured the temporal pattern of streamflow with reasonable accuracy.

Scenario Evaluation

Our earlier study (Sharma et al., 2015) analyzed the impact of water withdrawal for the current conditions of hydraulic fracking, and found modest impact on seven-day low flows. The present study analyzed the increasing drilling trend in the Muskingum watershed in order to determine the potential impact on environmental low flows and sustainable water availability for multiple uses. Low flows were evaluated for various fracking scenarios (Table 6); results are described in the subsequent section.

Future and Baseline Scenario for Current Climate Period

Sixteen subbasins with increasing drainage areas in the Tuscarawas watershed were selected to assess the impact of fracking on low flows at various locations. The analysis was accomplished by comparing the baseline scenario against the "future scenario on current climate period", i.e. only changing the withdrawals due to fracking activities. Figure 3-5 shows the relative change in the 7-day minimum flows between the baseline and future scenarios. The average flow measured during the seven consecutive days was computed, and minimum value from such record was compared for baseline and future scenario corresponding to various drainage areas. The link between the degree of impact of fracking and subbasin size is presented in Figure 3-6. This shows that the effect of withdrawal decreases with an increase in the drainage area, but remains above about 20%, which still represents a significant impact. Some outliers observed in the graph are mainly due to the interactions of other water use components and point sources on the same subbasins. A difference of approximately 9% in annual average flow was predicted between the two scenarios (Figure 3-7).

Future, Baseline and Current Scenarios for Future Climate Period

Environmental flow criteria were analyzed, using a plausible set of generated climate data for sixteen similar subbasins of the Tuscarawas watershed. These flow criteria were evaluated by using DFLOW 3.1 (EPA 2006) as a Windows-based tool, which was developed by the EPA. Figure 3-8 presents the comparison in *7Q10* between the baseline (without fracking) and future scenarios for the 30 year simulation period. Significant differences were detected under the future scenario compared with the baseline. The excess withdrawal due to future fracking reduced *7Q10* to zero for drainage areas less than 800 km^2.

Figure 3-9 shows the decreasing trend in the percentage difference in *7Q10* with the increase in drainage area when the baseline scenario was compared with the future scenario. A relatively smaller difference was detected when analysis was conducted in a large drainage area. However, the percentage difference in 7Q10 remains significant (about 15%) even for the largest drainage areas considered. Additionally, comparisons between the baseline and future scenario for 1B3 and 4B3 were performed on six different sub basins, and also showed a similar decreasing trend. The difference in the *4B3* and *1B3* between the two scenarios is shown in Figure 3-10. Figure 3-11 shows the decreasing trend in percentage differences in both criteria with an increase in drainage area.

Analysis was also conducted to see the effect of the future scenario on the flow duration curve, as it is one of the important statistical tools used to quantify hydrologic regimes (Kim et al. 2009). The ninety-five percentile flow exceedance was considered as the threshold for the extreme low flows, which indicates a stressful drought period as streams drop to the lowest level. Similarly, the seventy-five percentile flow exceedance was considered as low flow, which is the dominant low-flow condition sustained by the ground discharge into the streams. A subbasin with a drainage area of 920 km^2 was selected to analyze the flow duration curve between the current and future scenarios for a period of 30 years (Figure 3-12). The results showed that the high flow was not affected in this drainage area. However, the low flows were slightly affected as the alteration was noticed below 85% flow exceedance. The time series of the baseline, current and future scenario depicts the visible change in the base flow (not shown) indicating the impact of fracking withdrawals on flows in low-flow periods.

Conclusion

In this paper, the impacts of hydraulic fracking on water resources, especially during low-flow periods, was explored in the Muskingum watershed of Eastern Ohio, USA considering the worst case scenario. The watershed model, SWAT, was selected for the simulation of stream flows. The simulated flows at various locations were used to generate three scenarios that represent the range of potential impacts of water withdrawals for hydraulic fracking. The baseline scenario was based on realistic conditions of all water use data, but excluded withdrawals for fracking. The current scenario also used actual data for all water uses, including the present rate of water withdrawal for hydraulic fracking. The future scenario was modeled using 30 years of generated climate data based on the historical temperature and precipitation to the SWAT model. The SDSM was used to generate future temperature and precipitation based on the occurrence of the 25th percentile dryer period precipitation in a 30 year period (1961-1990). Since we were not sure whether or not the highest temperature and precipitation would occur simultaneously, we simply used the generated temperature. The condition might be more critical if the lowest precipitation and highest temperature occurred simultaneously.

Seven-day low flows showed some variability when compared baseline scenario with future projected scenario, indicating that flow alteration during low-flow periods might be an important consideration for regulation of water supply and quality. Simulations showed that *7Q10* could be reduced to zero for drainage areas less than 800 km^2 due to projected future fracking activity. The percentage decrease in *7Q10* for larger drainage areas (>6000 km^2) could be 15% or more. The difference due to fracking withdrawals was also noticeable on flow duration curves and base flow time series, further reinforcing the potential impact of fracking during low-flow periods. Even mean annual flows were forecast to decrease by 9% or more under the future scenario.

It is important to note that the analysis was conducted using a worst case scenario, assuming that fracking would continue to increase at the current trend. Based on the results of this study, we can conclude that the impact of hydraulic fracking may be very serious for the headwater streams. Impacts on higher order streams will be less severe, but potentially significant enough to affect aquatic life, water quality, and regulatory decisions..

The hydrologic (*7Q10*) and biological (*4B3* and *1B3*) design streamflows in subbasins of the Muskingum watershed were altered substantially, and in some cases (i.e., for headwater streams) dramatically, due to water withdrawal for hydraulic fracking in extreme scenarios. Therefore, it is essential that planners and decision makers account for water withdrawal for fracking while setting environmental flow criteria in NPDES permitting, particularly for headwater streams. This may prove challenging in a state like Ohio, where different agencies issue permits for oil and gas drilling activities (ODNR) and wastewater discharges (OEPA).

Uncertainties exist in the complex watershed model associated with the input data, model development, future projected well drilling and water use, future advancement in drilling technology, climate, etc. However, this study has shown that modeling tools are available to estimate the impact of fracking withdrawals on streamflows and provide regulators with a better understanding of potential management options

References

Abbaspour, K. C. 2007. User manual for SWAT-CUP, SWAT calibration and uncertainty analysis programs. *Swiss Federal Institute of Aquatic Science and Technology*, Eawag, Duebendorf, Switzerland, 1596-1602.

Abbaspour, K. C., Sonnleitner, M. A., and Schulin, R. 1999. Uncertainty in estimation of soil hydraulic parameters by inverse modeling: Example lysimeter experiments. *Soil Science Society of America Journal*, 63(3), 501-509.

Abbaspour, K. C., et al. 2007. Modelling hydrology and water quality in the pre-alpine/alpine Thur watershed using SWAT. *Journal of hydrology*, 333(2), 413-430.

Angle, M. A., Walker, D. D., and Jonak, J. 2001. Ground water pollution potential of Muskingum County, Ohio. Ohio Department of Natural Resources, Division of Water (No. 54). GWPP Report, 30-45

Arabi, M., et al. 2008. Representation of agricultural conservation practices with SWAT. *Hydrological Processes*, 22(16), 3042-3055.

Arnold, et al. 1996. SWAT. Manual, USDA. Agricultural Research Service and Blackland Research Center, Texas.

Arnold, J. G., et al. 1998. Large area hydrologic modeling and assessment part I: Model development1. *Journal of the American Water Resources Association.* 34(1) 73-89.

Arthur, J. D., Uretsky, M., and Wilson, P. 2010. Water Resources and Use for Hydraulic Fracturing in the Marcellus Shale Region. Prepared in conjunction with Co-Researchers including the New York State Energy Research and Development Authority, Susquehanna River Basin Commission, and the Delaware River Basin Commission under U.S. Department of Energy Research Project DE-FE0000797. International Petroleum and Biofuels Environmental Conference, San Antonio, Texas, August 30 – September 2. Available from: http://www. all-llc.com/publicdownloads/WaterResourcePaperALLconsulting.Pdf.

Auch. T. 2013. The Muskingum Watershed and Utica Shale Water Demands. (Avaliable from http://www.fractracker.org/2013/12/water-demands).

Beaver, W. 2014. Environmental Concerns in the Marcellus Shale. *Business and Society Review*, 119(1), 125-146.

Bieger, K., Hörmann, G., and Fohrer, N. 2014. Simulation of Streamflow and Sediment with the Soil and Water Assessment Tool in a Data Scarce Catchment in the Three Gorges Region, China. *Journal of Environmental Quality*, 43(1), 37-45.

Brunner, J., et al. 2000. Freshwater Systems. Washington, DC: World Resources Institute.

Cothren, J., et al., 2013. *Integration of Water Resource Models with Fayetteville Shale Decision Support and Information System*. Final Technical Report.University of Arkansas and Blackland Texas A&M Agrilife.

Craig, R. K. 2012. Hydraulic Fracturing (Fracking), Federalism, and the Water-Energy Nexus. Idaho L. Rev., 49, 241.

Engel, B., et al. 2007. A Hydrologic/Water Quality Model Application protocol. *Journal of the American Water Resources Association,43(5)*,1223-1236.

Entrekin, S., et al. 2011. Rapid expansion of natural gas development poses a threat to surface waters. *Frontiers in Ecology and the Environment*, 9(9), 503-511.

Eslamian, S. (Ed). 2014. *Handbook of Engineering Hydrology: Fundamentals and Applications*. CRC Press, 2014.

Faramarzi, M., et al. 2009. Modelling blue and green water resources availability in Iran. *Hydrological Processes*, *23*(3), 486-501.

Gassman, P. W., et al. 2010. The worldwide use of the SWAT Model: Technological drivers, networking impacts, and simulation trends. *International SWAT Conference Proceedings.* Seoul, South Korea, May 15 2012, pp. 129-137. (Available from http://swat.tamu.edu/media/33774/swat2010-proceedings.pdf).

Gregory, K. B., Vidic, R. D., and Dzombak, D. A. 2011. Water management challenges associated with the production of shale gas by hydraulic fracturing. *Elements*, 7(3), 181-186.

Gupta, H. V., Sorooshian, S., and Yapo, P. O. 1999. Status of automatic calibration for hydrologic models: Comparison with multilevel expert calibration. *Journal of Hydrologic Engineering*, 4(2), 135-143.

GWPC and ALL Consultant 2009. Modern Shale Gas Development in the United States: A Primer. US Department of Energy, Office of Fossil Energy, 2009.

Heber Green, W., and Ampt, G. A. 1911. Studies on Soil Physics. *The Journal of Agricultural Science*, 4(01), 1-24.

Howarth, R. W. 2014. A bridge to nowhere: methane emissions and the greenhouse gas footprint of natural gas. *Energy Science & Engineering*.[online]. (Available from: http://www.eeb.cornell.edu/howarth/publications/Howarth_2014_ESE_methane_emissions.pdf).

Hubbert, M. K., and Willis, D. G. 1972. Mechanics of hydraulic fracturing. Wahington, DC and Atherton, California. 239- 256.

Kim, U., and Kaluarachchi, J. J. 2009. Climate Change Impacts on Water Resources in the Upper Blue Nile River Basin, Ethiopia. *Journal of the American Water Resources Association*, *45*(6), 1361-1378.

Krause, P., Boyle, D. P., and Bäse, F. 2005. Comparison of different efficiency criteria for hydrological model assessment. *Advances in Geosciences*, 5(5), 89-97.

Montgomery, C. T., and Smith, M. B. 2010. Hydraulic Fracturing:History of an Enduring Technology. *Journal of Petroleum Technology*, 62(12), 26-40.

Moriasi, D. N., et al. 2007. Model evaluation guidelines for systematic quantification of accuracy in watershed simulations. *Transactions of ASABE,* 50(3), 885-900.

Myers, T. 2009. Review and Analysis of DRAFT Supplemental Generic Environmental

Impact Statement On The Oil, Gas and Solution Mining Regulatory Program Well Permit Issuance for Horizontal Drilling and High-Volume Hydraulic Fracturing to Develop the Marcellus Shale and Other Low-Permeability Gas Reservoirs.New York, New York and Reno, Nevada. 4-18.

Nash, J., and Sutcliffe, J. V. 1970. River flow forecasting through conceptual models part I—A discussion of principles. *Journal of hydrology*, 10(3), 282-290.

Neitsch, S. L., et al. 2005. SWAT theoretical documentation version 2005. Blackland Research Center, Temple, TX.

Nicot, J.P. 2013. Sources of data for quantifying hydraulic fracturing water use in

Texas. Washington D.C. ACS Publications.

Nicot, J. P., and Scanlon, B. R. 2012. Water use for shale-gas production in Texas, US. Environmental science & technology, 46(6), 3580-3586.

Pielke, R. A., and Wilby, R. L. 2012. Regional climate downscaling: What's the point?. Eos, *Transactions American Geophysical Union*, 93(5), 52-53.

Pyrce, R. 2004. Hydrological low flow indices and their uses. *Watershed Science Centre,(WSC) Report* (04-2004).

Rahman, M., Bolisetti, T., and Balachandar, R. 2010. Effect of climate change on low-

flow conditions in the Ruscom River watershed, Ontario. *Transactions of the ASABE*, 53(5), 1521-1532.

Rahm, B. G., and Riha, S. J. 2012. Toward strategic management of shale gas development: Regional, collective impacts on water resources. Environmental Science & Policy, 17, 12-23.

Rallison, R. E., and Miller, N. 1982. Past, present, and future SCS runoff procedure.

In Rainfall-runoff relationship/proceedings, International Symposium on Rainfall-Runoff Modeling held May 18-21, 1981 at Mississippi State University, Mississippi State, Mississippi, USA/edited by VP Singh. Littleton, Colo.: Water Resources Publications.

Robbins, K. 2013. Awakening the Slumbering Giant: How Horizontal Drilling

Technology Brought the Endangered Species Act to Bear on Hydraulic Fracturing. *Case* Western Reserve Law Review, 63(4), 13-16.

Rossman, L. A., and C. EPA. 1990. DFLOW user's manual Cincinnati. Ohio, *US Environmental Protection Agency*, *Risk Reduction Engineering Laboratory*, 946-950.

Santhi, C., et al. 2001. Validation of the SWAT Model on a large RWER basin with point and non point sources, *Journal of the American Water Resources Association*, 1169-1188

Santhi, C., et al. 2014. An integrated modeling approach for estimating the water quality benefits of conservation practices at the river watershed scale. *Journal of Environmental Quality*, 43(1), 177-198.

Satterfield, J., et al. 2008, September. Managing Water Resources Challenges in Select Natural Gas Shale Plays. Chesapeake Energy Corp. Presented at the *Ground Water Protection Council Annual Meeting*. *Proceedings of the GWPC Confrenece*.

Schuol, J., et al. 2008. Estimation of freshwater availability in the West African sub-continent using the SWAT hydrologic model. *Journal of Hydrology*, *352*(1), 30-49.

Saunders, J. F., et al. 2004. The influence of climate variation on the estimation of low flows used to protect water quality: A nationwide assessment. *Journal of the American Water Resources Association.*, 1339-1349.

Sharma, S., et al. 2012. Incorporating climate variability for point-source discharge permitting in a complex river system. *Transactions of the ASABE*, 55(6), 2213-2228.

Sharma S., et al. 2015. Hydrologic Modelling to Evaluate the Impact of Hydraulic Fracturing on Stream Low Flows Using SWAT Model: Challenges and Opportunities for Simulation Study. *American Journal of Environmental Science*, 11(4), 199-215.

Srinivasan, R., et al. 1998. Large area hydrologic modeling and assessment – Part II: Model application. *Journal of the American Water Resources Association*, 34 (1) (1998), 91–101.

Stehr, A.,et al. 2008. Hydrological modelling with SWAT under conditions of limited data availability: evaluation of results from a Chilean case study. *Hydrological Sciences Journal*, 53(3), 588-601.

Sullivan, K.,et al. 2013. Modeling the Impact of Hydraulic Fracturing on Dringking Water Resources based on water acquition scenarios: *U.S. Environmental Protection Agency*

Office of Research and Development National Exposure Research Laboratory (NERL) *Ecosystems Research Division* (ERD) Athens, Georgia 30605 USA, 1(1).16-30.

Tatli, H., Nüzhet Dalfes, H., and Sibel Menteş, Ş. 2004. A statistical downscaling method for monthly total precipitation over Turkey. *International Journal of Climatology*, 24(2), 161-180.

USDA, S. 1991. State soil geographic data base (STATSGO) data user's guide. Miscellaneous publication, 1492.

US EIA. 2011. Summary Maps: Natural Gas in the Lower 48 States, Washington DC. (Available from: http://www.eia.gov/oil_gas/rpd/shale_gas.pdf [accessed 26 July 2011].

US EPA. 2012. Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources: Progress Report. EPA/601/R-12/011.

US EPA 2006. DFLOW Version 3.1 User’s Manual. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

US EPA 2011. Plan to Study the Potential Impacts of Hydraulic Fracturing on Drinking Water Resources. EPA/600/R-11/122. US Environmental Protection Agency, Office of Research and Development. (Available from: http://water.epa.gov/type/groundwater/uic/class2/hydraulic fracturing/upload/hf_study_plan_110211_final_508.pdf.)

Vazquez-Amábile, G. G., and Engel, B. A. 2005. Use of SWAT to compute groundwater table depth and streamflow in the Muscatatuck River watershed. *Transactions of the ASAE*, 48(3), 991-1003.

Whetton, P.,et al. 2012. Use of Representative Climate Futures in impact and adaptation assessment. *Climatic Change*, 115(3-4), 433-442.

Wilby, R. L., Dawson, C. W., and Barrow, E. M. 2002. SDSM—a decision support tool for the assessment of regional climate change impacts. *Environmental Modelling & Software*, 17(2), 145-157.

Wilby, R. L., and Dawson, C. W. 2013. Statistical downscaling model–decision centric (SDSM-DC) version 5.1 supplementary note. *Loughborough University, Loughborough*.

Wiley, J. B. 2006. Low-flow Analysis and Selected Flow Statistics Representative of 1930-2002 for Streamflow-gaging Stations in Or Near West Virginia. US Geological Survey.

Yang, J., et al. 2008. Comparing uncertainty analysis techniques for a SWAT application to the Chaohe Watershed in China. *Journal of Hydrology*, 358(1), 1-23.

Yew, C. H., and Weng, X. 2014. Mechanics of hydraulic fracturing. Gulf Professional Publishing Co., Houston, TX, USA,182 PP, Second Edition.

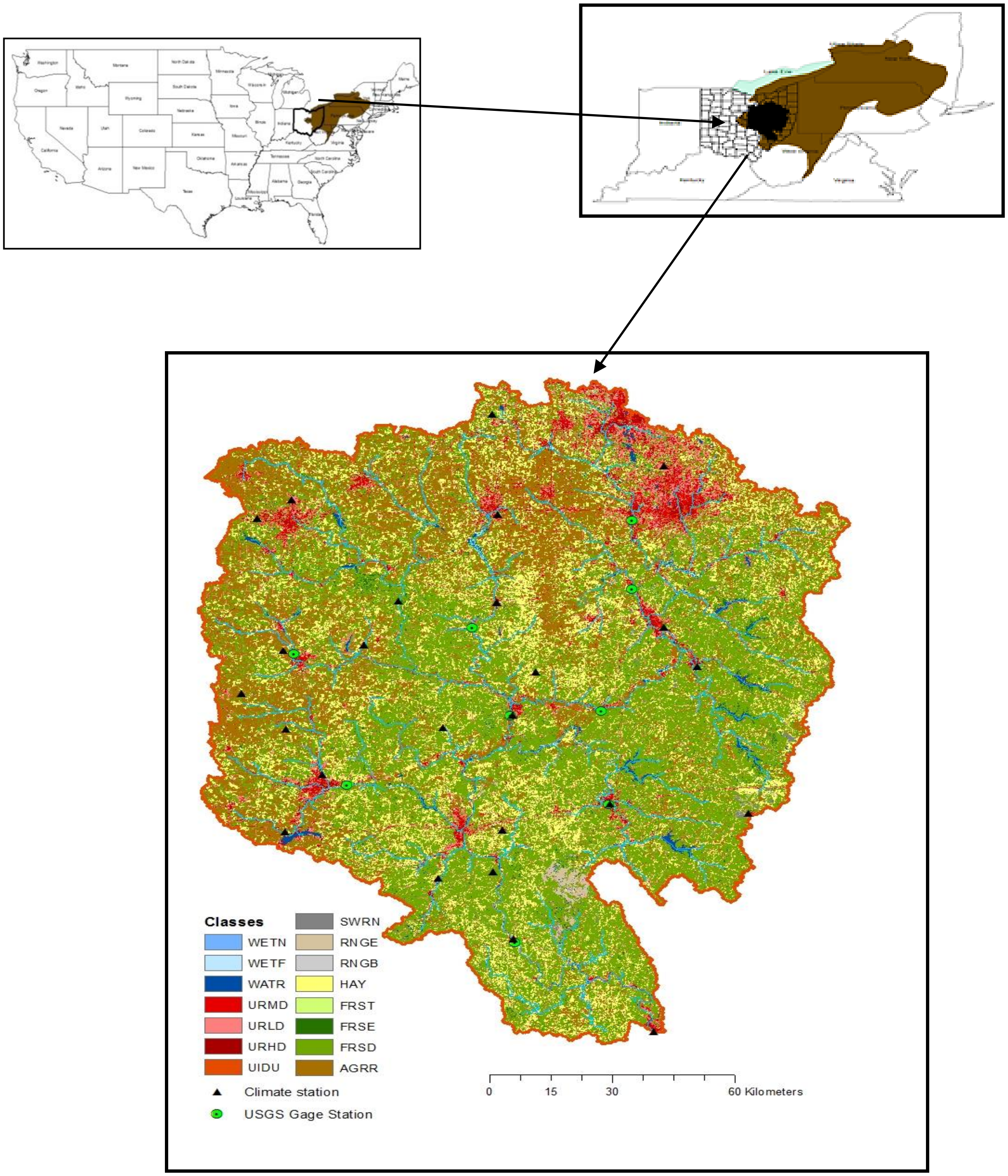


Figure 3-1: Location map with climate and USGS stations in the Muskingum watershed. Legend represents the herbaceous wetland (WETN), wetland forest (WETF), open water (WATR), urban medium density (URMD), urban low density (URLD), urban high density (URHD).

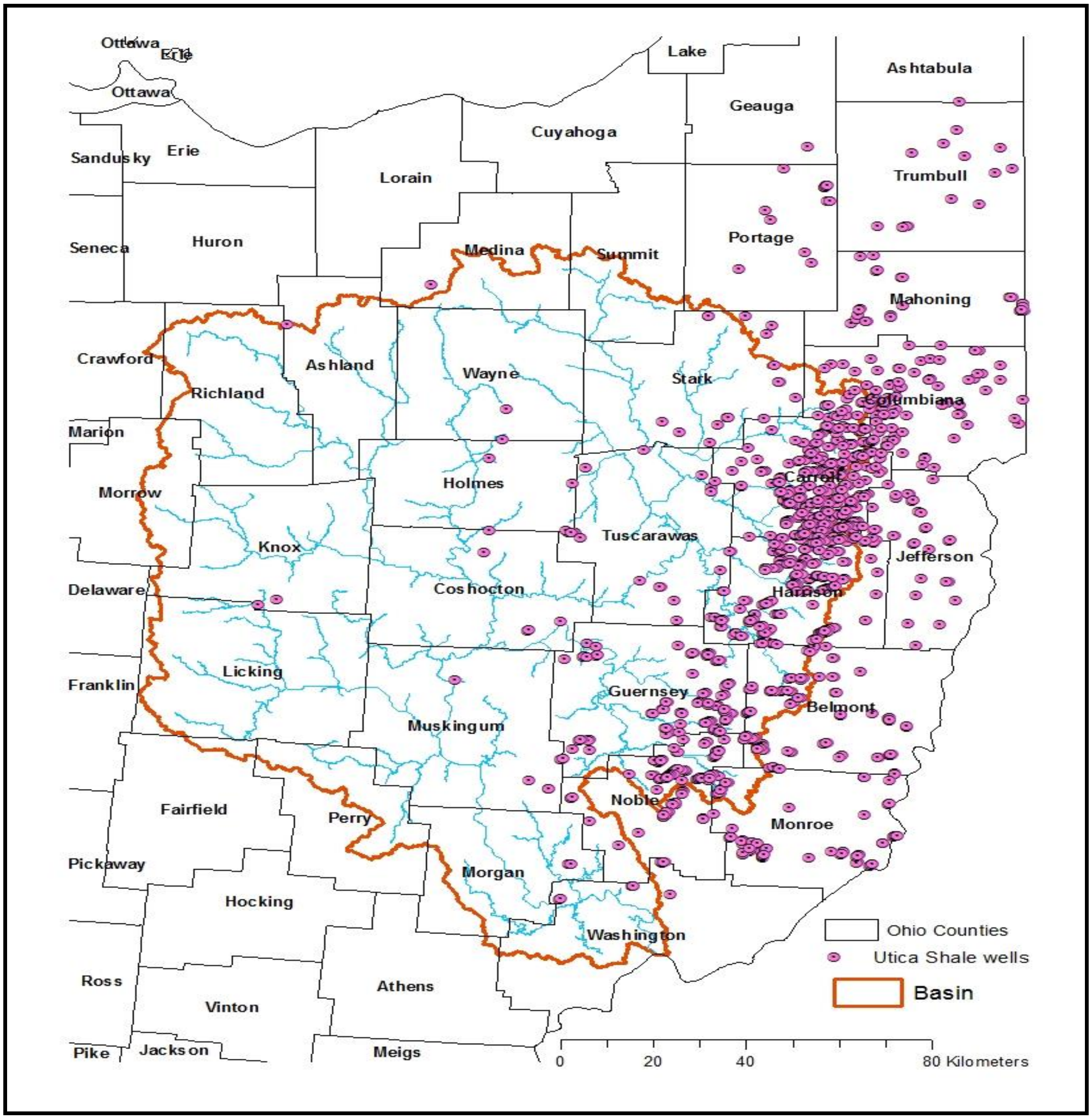


Figure 3-2: Utica shale wells in the State of Ohio from January 2011 to September 2013, indicating more wells in the Eastern Ohio.

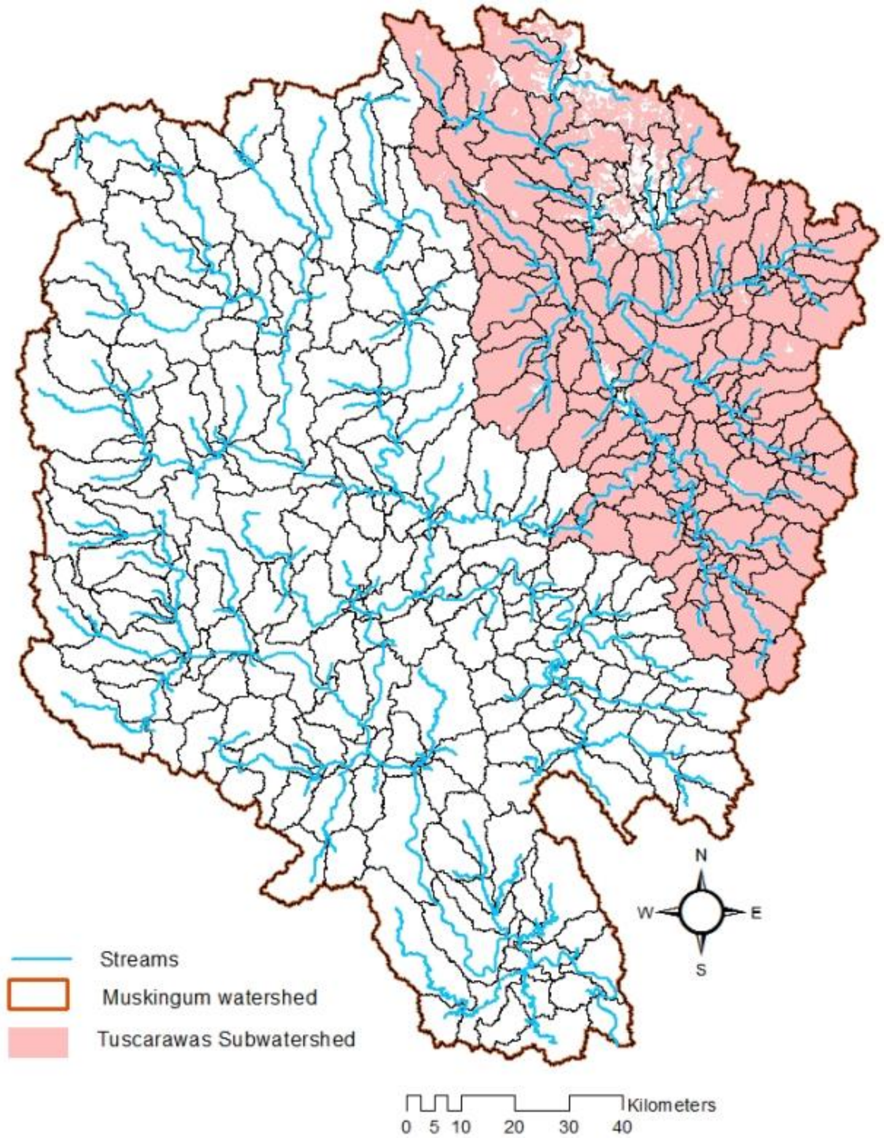


Figure 3-3: Tuscarawas sub watershed (red color) in upper part of the Muskingum watershed, where the drilling for hydraulic fracking has been increasing significantly.

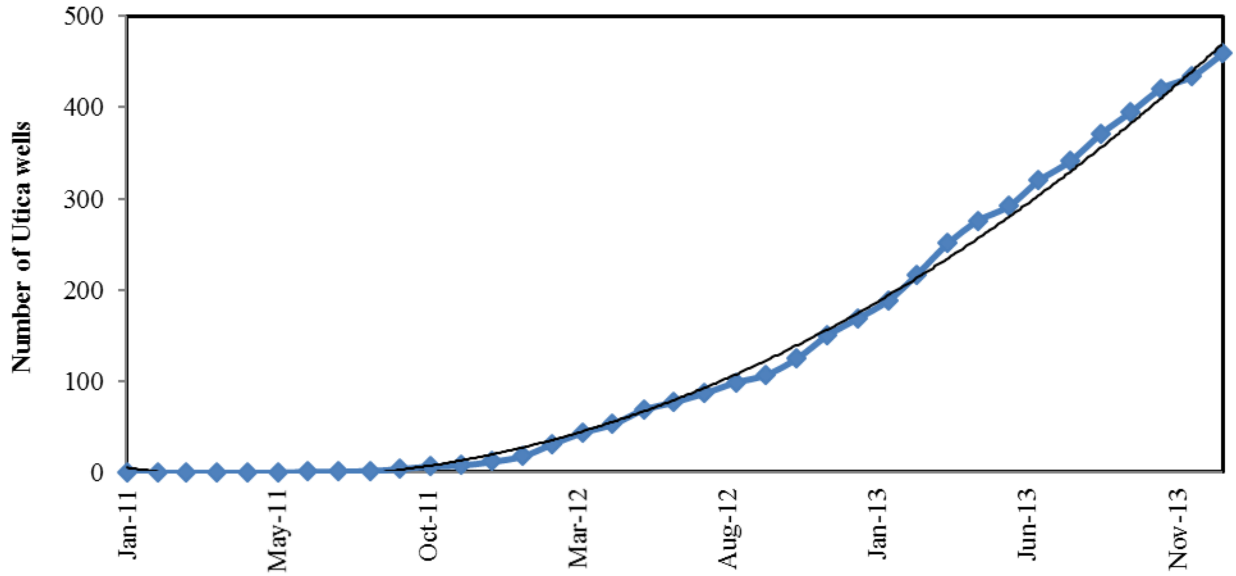


Figure 3-4: The trend of Utica wells in Ohio from January 2011 – May 2014.

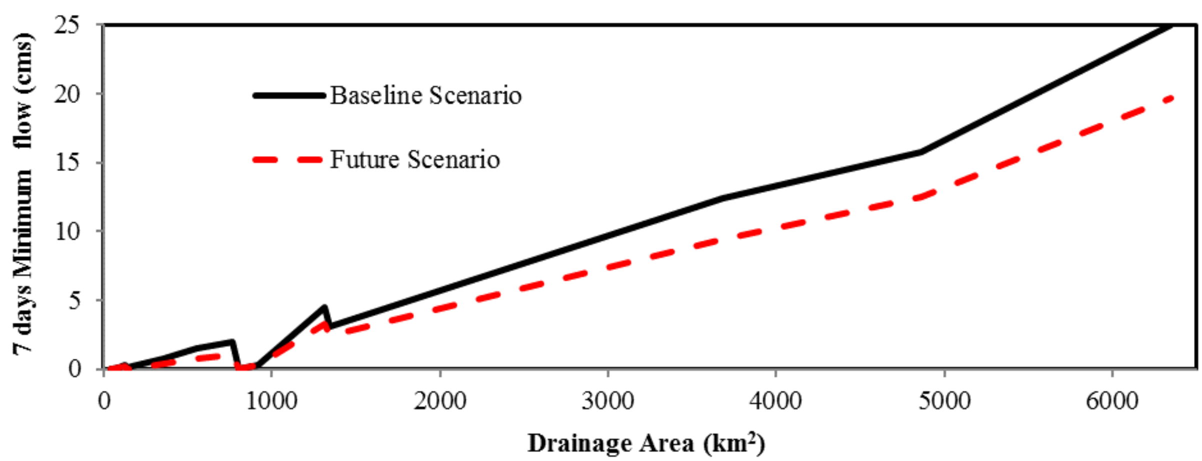


Figure 3-5: The seven-day monthly minimum flow during low-flow period for baseline and future scenario analyzed for current climate period in Muskingum Watershed.

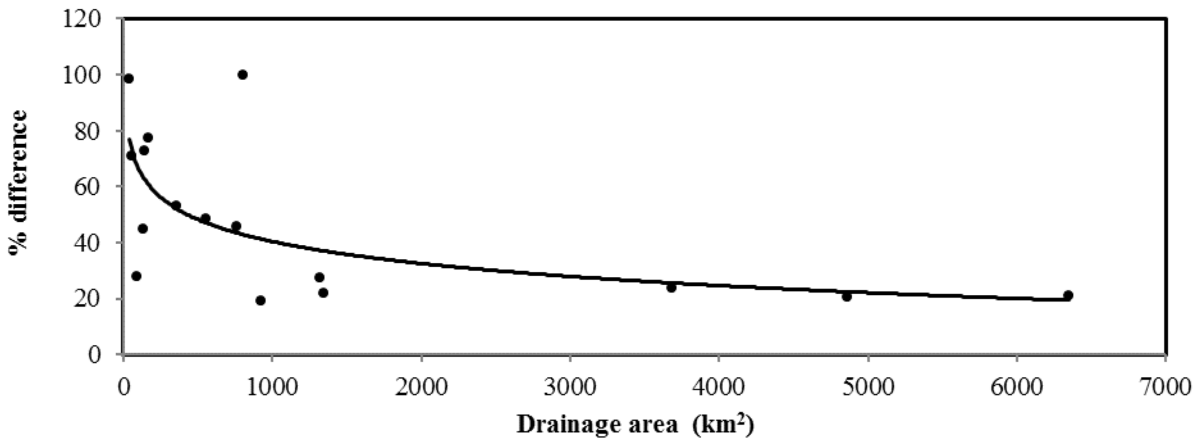


Figure 3-6: The percentage difference in the 7 day minimum flows between the baseline and future hydraulic fracking scenarios simulated using current climate data in Muskingum watershed.

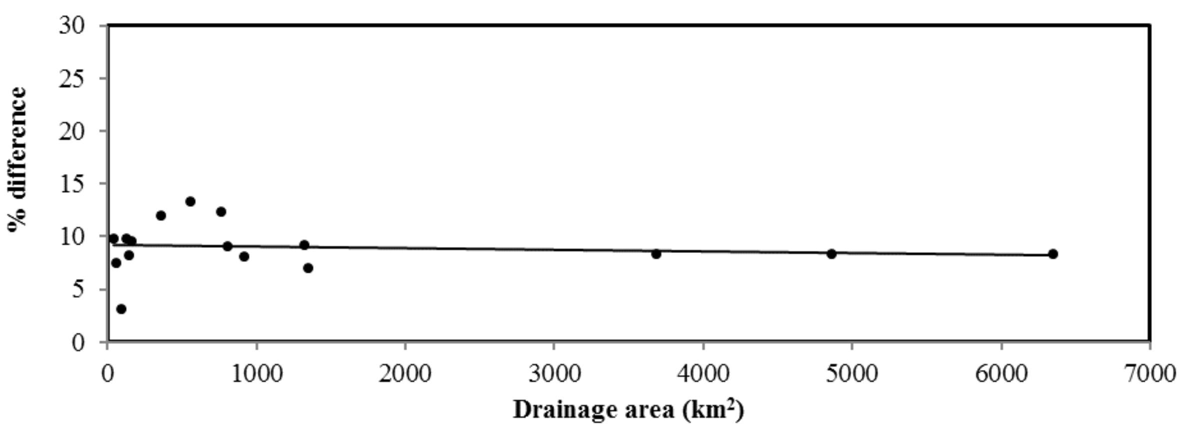


Figure 3-7: The percentage difference in annual average flows between the baseline and future hydraulic fracking scenarios simulated using current climate data in Muskingum watershed.

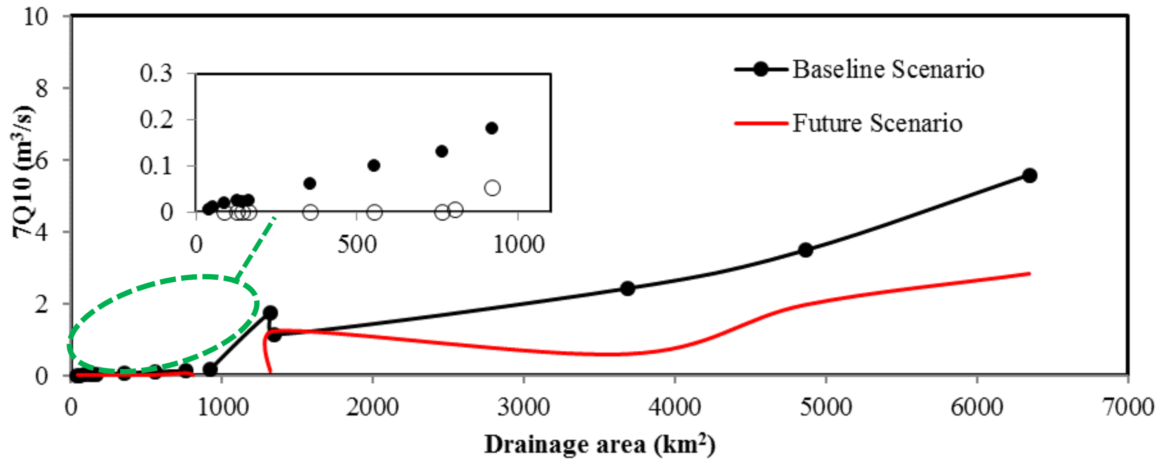


Figure 3-8: The 7Q10 flows for the baseline and future projected hydraulic fracking scenarios using the climate data of future period for 30 years generated by SDSM.

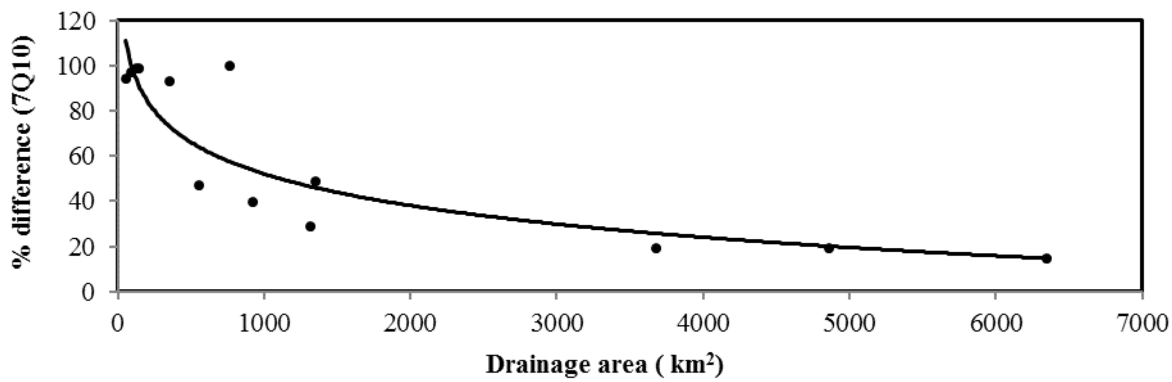


Figure 3-9: The percentage difference in 7Q10 between the baseline vs. projected future hydraulic fracking scenario using 30 years of projected climate data from SDSM.

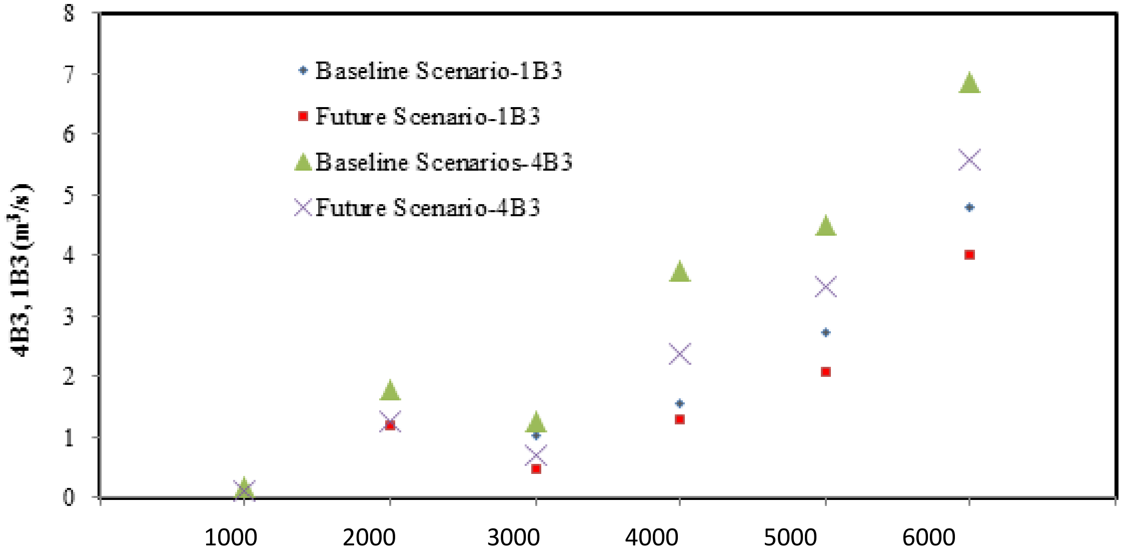


Figure 3-10: The 4B3 flows for the baseline vs. future fracking scenario using the climate data generated by SDSM for 30 years.

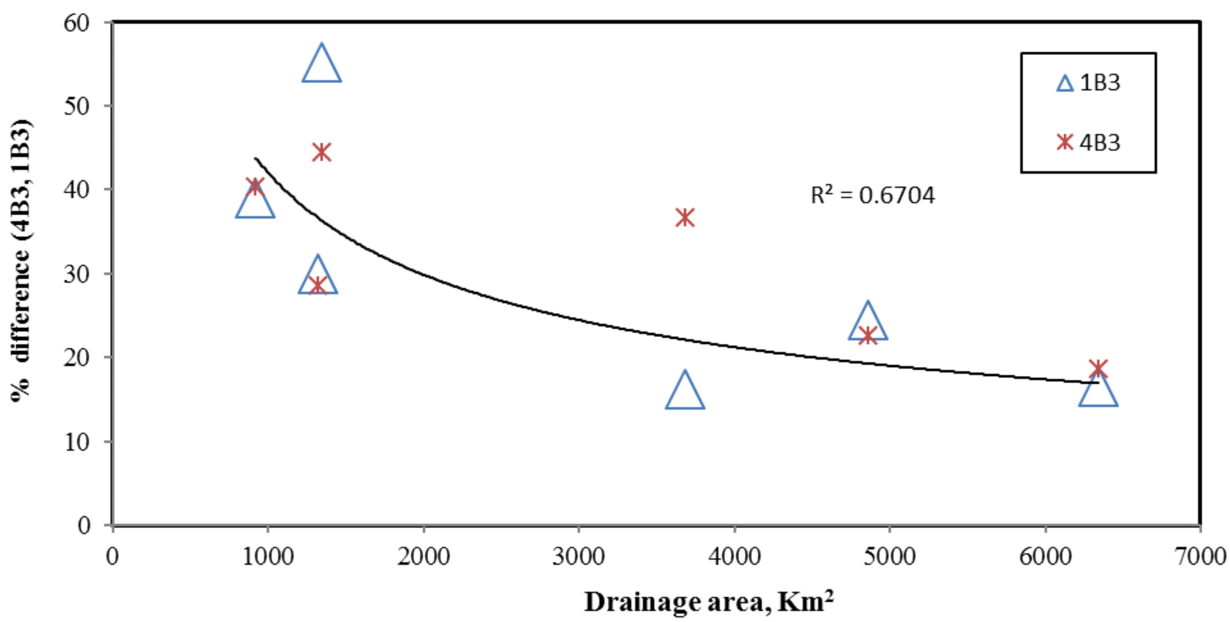


Figure 3-11: The percentage difference in 4B3 and 1B3 between the baseline and future hydraulic fracking scenarios using climate data generated by SDSM for 30 years.

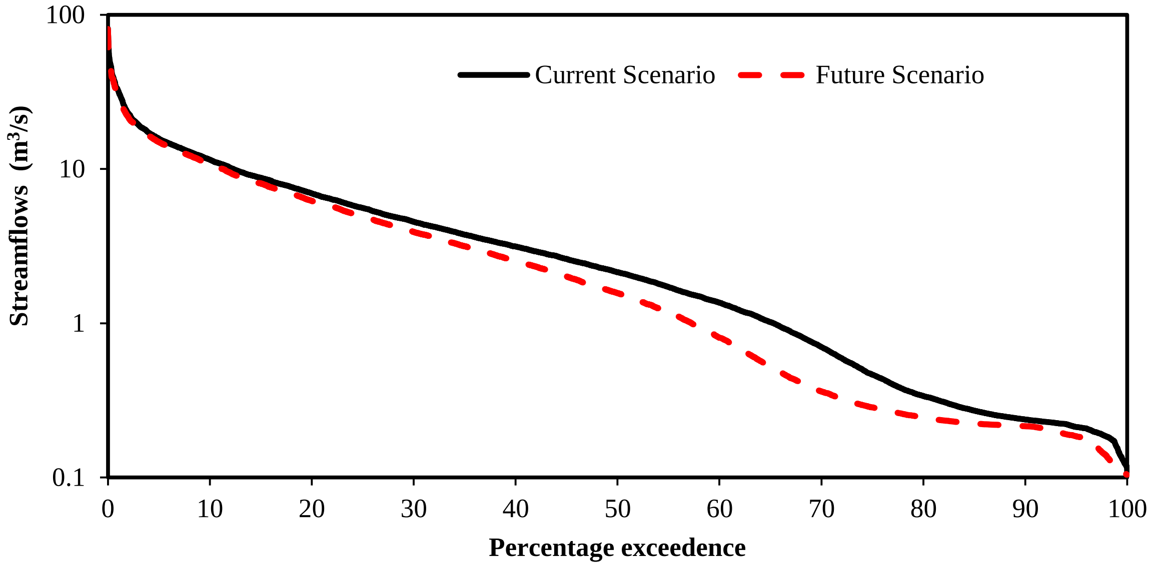


Figure 3-12: The flow duration curve for the current and future hydraulic fracking scenarios using the climate data generated by SDSM for 30 years in Muskingum watershed.

Table 3-1: The data used for SWAT model development in the Muskingum watershed

Data Type	**Data**	**Source**
GIS	30-meter DEM	USGS National Geospatial Program (NGP) http://viewer.nationalmap.gov/viewer/
	Land use and Land cover 2006	USGS National Geospatial Program (NGP), National Land Cover Dataset (NLCD), http://viewer.nationalmap.gov/viewer/
	Soil Data	Geographic STATSGO soil map (scale of 1:250,000)
Climate	Rainfall and Temperatures	NOAA's National Climatic Data Center (NCDC) http://www.ncdc.noaa.gov/cdo-web/
Hydrology	Streamflows	USGS http://waterdata.usgs.gov/usa/nwis/sw
	Reservoir outflow	U.S. Army Corps of Engineers (USACE)
Stream Networks/Water Bodies	Streams and flow direction, reservoirs	USGS National Geospatial Program (NGP) National hydrograph dataset (NHD)
Water Use (Surface and Ground Water)	Irrigation, Public, Power, Mineral Extraction, Industries and Golf Course	Ohio Department of Natural Resource (ODNR) Ohio Environmental Protection Agency (OEPA)
Point Sources	Flow discharge	OEPA
Oil and Natural Gas	Wells for Hydraulic Fracking	ODNR
	Sources of Drilling Water	ODNR
	Drilling Water Estimate per well and Future Drilling Trend	FracFocus (http://fracfocus.org) (National hydraulic fracturing chemical registry)

Table 3-2: Water withdrawal for hydraulic fracking in the Muskingum watershed, 2011-2013.

Year	Average Vertical Depth (m)	Freshwater (m^3)	Recycled Water (%)
2011	7,717	11,449	13.79
2012	7,734	13,011	4.35
2013	10,897	16,680	3.74

Table 3-3: List of predictors used in the SDSM model to downscale the NCEP reanalysis data

Station Parameters	Predictor Variable
Precipitation	Zonal velocity component at surface level
	Zonal velocity component at 850 hpa
	Geostrophic airflow velocity at 850 hpa
	Vorticity at surface level
	Vorticity at 850 hpa
	Sea level pressure
	Specific humidity at 500 hpa
	Specific humidity at 850 hpa
Temperature	Mean Temperature
	Near surface specific humidity

Table 3-4: The statistical criteria measuring the daily performance of the SWAT model during the calibration (2002-2009) and validation (1995-2001) period

USGS Gage Station	Calibration			Validation		
	R^2	*NSE*	*PBIAS*	R^2	*NSE*	*PBIAS*
3117000	0.42	0.42	0%	0.45	0.45	-2%
3124500	0.43	0.40	-16%	0.53	0.47	-10%
3139000	0.51	0.49	-4%	0.63	0.60	-3%
3136500	0.47	0.46	9%	0.41	0.40	18%
3129000	0.57	0.56	3%	0.54	0.47	21%

3140500	0.63	0.63	2%	0.69	0.65	12%
3146500	0.42	0.40	11%	0.43	0.42	12%
3142000	0.55	0.47	13%	0.51	0.49	-2%
3150000	0.65	0.65	0%	No Data		

Table 3-5: The statistical criteria measuring the monthly performance of the watershed model during the calibration (2002-2009) and validation (1995-2001) period.

USGS Gage Station	**Calibration**			**Validation**		
	R^2	*NSE*	*PBIAS*	*R^2*	*NSE*	*PBIAS*
3117000	0.89	0.89	0%	0.9	0.86	7%
3124500	0.59	0.53	-16%	0.63	0.61	-9%
3139000	0.64	0.64	-4%	0.72	0.71	-3%
3136500	0.50	0.49	9%	0.63	0.56	18%
3129000	0.68	0.66	3%	0.67	0.55	21%
3140500	0.68	0.67	1%	0.76	0.69	13%
3146500	0.79	0.72	11%	0.76	0.71	12%
3142000	0.71	0.69	13%	0.77	0.68	-1%
3150000	0.73	0.72	0%	No Data		

Table 3-6: Characteristics of the various scenarios that were used for analysis.

Scenarios	**Climate conditions**	**Hydraulic Fracking**
Baseline (without fracking) scenario	Current	No
Current (with fracking) scenario	Current	Yes
Future scenario on current climate period	Current	Projected fracking
Future scenario on future climate period	Future	Projected fracking

Chapter 4. Impact of Global Climate Change on Stream Lowflows in a Hydraulic Fracking Affected Watershed

Abstract

The impact of fresh water withdrawals for hydraulic fracturing has concerned water resource scientists and communities interested in sustainable water resource management. Specifically, low flow conditions in watersheds may be further reduced due to global climate change, as it has the potential to decrease streamflow. Since an earlier study found that the current rate of fracking had some impact on water availability, this study was conducted in order to ascertain whether or not the current fracking trend will have an impact in stream low flow in the future. This study was conducted on the Muskingum watershed in Eastern Ohio, which has been subjected to the rapid expansion of hydraulic fracking. The watershed model, Soil and Water Assessment Tool (SWAT), was used for watershed simulation using the climate output of Coupled Model Intercomparison Project Phase 5 (CMIP5). Precipitation and temperatures outputs from Max Planck Institute Earth System Model (MPI-ESM) were used to evaluate the variation in streamflow during the 21st century using three Representative Concentration Pathways (RCP) scenarios: RCP 2.6; RCP 4.5; and RCP 8.5. Three future periods, namely, 2035s (2021-2050), 2055s (2051-2070) and 2085s (2070-2099) were set against the baseline condition (1995-2009). Lowest flow was projected to increase across the watershed during 2035s period compared to the remaining 50 years period, under the highest forced climate scenario (RCP8.5). Similarly, mean flow also could be expected to decrease during 2035s in the eastern, north-western and south western portion of the watershed. Additionally, the streamflow was simulated using current fracking scenarios and 2035s climate output in order to assess the impact of water withdrawal for a continuous trend of current fracking rates. A modest effect on stream low flow was detected, when extreme scenario (RCP 8.5) was considered, especially in the

headwater streams. While results indicate that 14 of 32 subbasins were affected, with maximum difference up to 55% in lowest 7 days minimum low flow (considered lowest value from each year), negligible impact was detected on mean monthly and annual streamflow. Analysis with RCP 2.6 and RCP 4.5 indicated that stream low flow would not be affected especially in higher order streams. Even though a localized effect of hydraulic fracking to reduce the environmental flow was detected; this research indicated that future climate change may not have additional adverse impacts if hydraulic fracking trends are stable.

Keywords: CMIP5, climate model, hydrologic analysis, MPI-ESM, SWAT, Low flow, and Drought.

Introduction

Water resources managers have a particular concern about the water resources management during low flow in order to optimally utilize the freshwater resources for various purposes including water supply, recreation, wildlife conservation and reservoir flow regulation. Hydrological, droughts are the most crucial and categorized as the most stressful events in the hydrological cycles. Therefore, stream low flow due to hydrologic drought has become a particular interest of research topics among scientists due to its characteristics of reducing the groundwater, lowering of the reservoir or lake level and decreasing streamflow discharge for consecutive years (Smakhtin, 2000).

Various natural factors contribute to low flow variability leading to the social and economic impacts (Burn et al. 2008). Additionally, fluctuation of low flow is affected by anthropogenic impacts, which may cause supplementary severe conditions in the dry period (Smakhtin, 2000). For example, a large amount of water abstraction for industrial uses, irrigation, power generation and domestic water use reduces the downstream water volume

(Benejam et al. 2010). Similarly, agricultural practices may also cause significant increase in the frequency of low flow discharge (Wilber et al. 1996; Kottegoda and Natale 1994; Eheart and Tornil 1999), leading to the frequent low flow and complications in optimal allocation of water resources.

In addition to conventional anthropogenic influences, water withdrawals for hydraulic fracking have been an emerging critical issue, especially for low flow periods when severe drought occurs (Burton et al. 2014). A significant amount of water has been used from the streams and reservoirs for hydraulic fracking without consideration for ecological and environmental impacts. While the fracking water volume is small in comparison with the total water availability in any area, the water withdrawal for drilling and fracturing operations over a small tributary might be ecologically stressful and may threaten the sustainable water resource management. For example, it may create additional deficits to municipal water supplies and adversely impact aquatic life in the stream during low flow period. Spatially, this imbalance can be further worsened for specific small tributaries, and the streamflow variation may be more prominent at the sub basin scale (Cothren et al. 2013). There are also various impact from fracking operations such as groundwater contamination, drying out the groundwater wells and streams, however these studies were outside the scope of this paper.

Declining flow rates may be further stressed due to increasing rise in global temperature (Vorosmarty et al. 2000; Alcamo et al., 2003) and future precipitation trend associated with global climate change leading to the alteration in the hydrological cycle and threatening the sustainable water resources management. Therefore, there is a pressing need to explore the impact of global climate change on stream low flows, especially for a watershed which is subjected to the long term water demands for hydraulic fracking. Therefore, this study was conducted in order to determine the impact of projected global climate change during stream low flow conditions in the watershed under continued hydraulic fracking in the future.

Materials and Methodology

Climate Model

World Climate Research Program (WCRP) has developed the multi-model climate dataset through Coupled Model Intercomparison Project (CMIP) and made freely available through the Program for Climate Model Diagnosis and Intercomparison (PCMDI) (Brekke et al. 2013, Taylor et al. 2011). Various phases of CMIP have increased slowly over the years. The third phase of CMIP; Coupled Model Intercomparison Project Phase 3 (CMIP3) (Meehl et al. 2005; 2007a) has provided important information to the Intergovernmental Panel on Climate Change (IPCC) fourth assessment repot (IPCC, 2007). Similarly, IPCC's fifth assessment report relies heavily in Coupled Model Intercomparison Project Phase 5 (CMIP5). CMIP5 dataset incorporates four newly developed sets of climate forcing scenarios called representative concentration pathways (RCPs). RCP 8.5, RCP 6.0, RCP 4.5 and RCP 2.6 (Moss et al. 2010, Vuuren et al. 2011) are scenarios with concentration, emission and land-use trajectories (Janssen 2013). Among these scenarios, RCP 8.5 is the highest emission scenario, including greater greenhouse gas concentrations and warming effects than other three scenarios. Similarly, RCP 6.0 is considered as a midrange emission scenario and RCP 4.5 as a low range emission scenario. RCP 2.6 is considered as a strong mitigations scenario, which includes the increase of greenhouse gases and temperature changes to the first part of the 21st century and decreasing trends for both features on the second half of century (Maurer et al. 2014; Taylor et al. 2012). CMIP5 incorporates Earth System Models (ESMs), Atmosphere-Ocean General Circulation Models (AOGCMs) and Earth System Models of Intermediate Complexity (EMICs), which helps to study the impact of carbon responses on climate change (Taylor et al. 2012).

Since CMIP5 has incorporated new General Circulation Model (GCM) projections with relatively more physical process than previously published dataset CMIP3 (Knutti and Sedlacek 2013), the recently available CMIP5 data has been utilized for this study. A widely used, Soil and Water Assessment Tool (SWAT) (Arnold et al. 1998) was developed in order to simulate the coupled impact of hydraulic fracking and future climate change on stream low flow. While there are several publications regarding the application of CMIP3 dataset to assess the variability on hydrological regimes (Arnell et al. 2013, Vliet et al. 2013), relatively limited articles have been published (Ficklin et al., 2013) using CMIP5 data. To the authors' knowledge, no studies have been conducted yet, for the evaluation of stream low flow due to the integrated effect of climate change and fracking, particularly in a watershed, which has a tremendous potential for hydraulic fracking.

The Max Planck Institute Earth System Model (MPI-ESM) is a newly revised model in the CMIP5 with the essential addition of carbon cycle, radiative transfer scheme, aerosol forcing and integration of dynamic vegetation at the land surface (Giorgetta et al. 2013, Hagemann et al. 2013). The model has three configurations: MPI-ESM low resolution (MPI-ESM-LR), mixed resolution (MPI-ESM-MR) and paleo resolution (MPI-ESM-P). Among these configurations, MPI-ESM-LR has been widely used for the experimentations and simulations of CMIP5 (Giorgetta et al. 2013).

Study Area

The study was conducted in the Muskingum watershed, which is located in the eastern part of the Ohio, USA (Figure 4-1). The Muskingum watershed is one of the major watersheds of Ohio, which covers almost 20% (8000 square mile) area of entire Ohio. This is a Hydrologic Unit Code (HUC)-4 watershed characterized with several wetlands, lakes and reservoirs. The average flow at the outlet of the watershed is 78 m^3/s and average annual precipitation over the entire watershed is almost 990 mm. The development of oil and natural gas is evolving substantially in this region. Currently, 90% of the total numbers of wells located in Ohio are concentrated in this watershed as several drilling companies are exploring in this region for oil and gas development. The water consumption for hydraulic fracking was noteworthy than the agricultural water consumption during 2012. The surface water consumption for the studied area, Muskingum watershed is depicted in Table 4-1 during 2012. Even though the fracking water withdrawal was far less than other primary use such as public, power, mineral extraction, industry and other uses, this might be very useful to study the fracking impact during low flow period in extreme drought conditions. In comparison to ground water withdrawal for hydraulic fracking, it was merely 1% as compared to surface water withdrawal during year 2012. In order to evaluate the impact of climate change and water withdrawal for fracking in stream low flow, SWAT model was developed. The impact of fracking without considering climate change was also conducted in this watershed in a previous study (Sharma et al., 2015). The input data needed for SWAT model development are briefly described in the following section.

SWAT Model Input

The digital elevation model (DEM) needed for watershed delineation and soil data (STATSGO) for watershed modeling were utilized from the United States Geological Survey (USGS) and United States Department of Agriculture (USDA), respectively. Land use data were downloaded from the National Land Cover Database 2006 (NLCD 2006). The complexity of landscape was obtained by the division of land use and subdivision of sub-watersheds into total 6176 HRUs. Similarly, precipitation and temperature datasets were utilized from 23 precipitation and 19 temperature gage stations in order to adequately address the spatial variability of the rainfall and temperature. The daily streamflow data needed for the multi-site model calibration and validation were obtained from nine USGS locations within the watershed from period 1993 to 2009. Since the watershed was characterized with several reservoirs and point sources, daily mean reservoir outflow were obtained from the United States Army Corps of Engineers (USACE) and major point sources with greater than 0.026 cms were used from the Ohio Environmental Protection Agency (OEPA). The water use data from the watershed for various purposes such as ground water, irrigation, water supply, power plant, industry and hydraulic fracking were obtained from the Ohio Department of Natural Resources (ODNR). All types of information related with oil and natural gas were utilized from the ODNR. However, the information related with the water use and water recycling associated with fracking well was utilized from the fracfocus. The spatial location of climate stations including USGS gauging stations, reservoirs and fracking wells are presented in Figure 4-1. Readers can refer our earlier publication (Sharma et al. 2015) for additional input information for the SWAT model development and modelling issues of hydraulic fracking in SWAT.

Model Calibration and Validation

The multi-site SWAT model calibration and validation were performed using automated calibration, validation and uncertainty analysis, developed in Swat Calibration and Uncertainty Program (SWAT-CUP) (Abbaspour et al., 2007). Sequential Uncertainty Fitting version 2 (SUFI-2) algorithm was selected in SWAT-CUP that finds out the most favorable model parameters within the uncertainty ranges of 95% after incorporating the possible parameters ranges (Abbaspour et al., 2007, Sharma et al. 2015). Twenty-one model parameters (not shown) were initially selected and optimal set of model parameters were chosen based on the model performance. The SWAT model was calibrated at nine USGS stations using streamflow data from 1993 to 2009. Two years of streamflow data were used for model spin up period in order to stabilize the initial hydrological conditions. The model was calibrated at nine various locations of the watershed both in daily and monthly scale using USGS streamflow data from 2002 to 2009. The model was also validated using the independent datasets from 1995 to 2001 at nine subsequent USGS locations. The goodness of fit was evaluated with a popularly used objective functions including Nash-Sutcliffe efficiency (NSE), R-square (R^2), root mean square error (RMSE), percentage bias (Pbias) etc. Readers can refer Sharma et al. (2015) for the detail description of these statistical criteria.

Climate Change Analysis

In order to evaluate future impacts on freshwater resources under climate change scenario, the latest daily time scale of climate data (precipitation, minimum and maximum temperature) were downloaded from publicly available archives for CMIP5 climate data, using bias corrected-constructed analogs (BCCA) (Maurer et al. 2010) downscaling technique. The spatial resolution was selected at 1/8 degree across the watershed. For this study, two CMIP5 simulations were analyzed: one for the evaluation of various climate model performances, and the second for future projected climate change. Since several climate models exist with different climate forcing functions, it was essential to evaluate the performance of each climate model and find an appropriate model in a given watershed. For this, historical climate data was downloaded from 1961 to 1990 at two climate stations (0335747 and 0014891) for two forced scenarios (RCP 4.5 and 8.5). The study indicated that the Max Planck Institute earth system model (MPI-ESM) was superior in its performance, based on the correlation coefficient.

In order to best conduct a future climate change study, RCP 2.6, RCP 4.5 and RCP 8.5 forced scenarios were selected from 2020 to 2099 for 23 climate stations, and downscaled to the same climate stations which were used for SWAT model calibration and validation for the hydrological simulations.

Result and Discussion

SWAT Model Calibration and Validation

The model performance is satisfactory both for daily and monthly simulation. The statistical criteria measuring the performance of the model including *NSE*, R^2, and *Pbias* are listed in Table 4- 2. The time series of the observed flow vs. simulated flow using the calibrated and validated SWAT model is reported in Figure 4-2 and Figure 4-3, respectively. The calibrated model was utilized for the prediction of future streamflow using bias corrected downscaled climate data at various stations.

Selection of a Climate Model

The performance of 19 climate models were examined by comparing model projected data for a historical period with observed data using squared correlation coefficient. For this, CMIP5 datasets using BCCA downscaling methods were downloaded for RCP scenarios 4.5 at precipitation stations 00335747 and 00014891 and RCP scenarios 8.5 at station 00014891. The performance of the model varied significantly, and the model performances in terms of squared correlation coefficients for monthly mean precipitations are presented in the Figure 4-4. Out of the 19 models, the performance of MPI-ESM-LR was superior, which was determined based on the squared correlation coefficient (Figure 4-4). Both the configurations: MPI-ESM-LR and MPI-ESM-MR performed well for RCP 8.5 and RCP 4.5 at station 0014891 (not shown). However, the performance of MPI-ESM-LR model with RCP 4.5 was relatively better at station 00335747 (Figure 4-4). As the MPI-ESM-LR configuration fitted well with the observed output in all the correlation tests and used with wide range for the CMIP5 simulations, MPI-ESM-LR was selected for this specific study.

Subsequently, MPI-ESM-LR dataset for RCP 8.5 was selected for the assessment of climate change on the hydrological cycle at three time periods: 2035s, 2055s and 2085s.

Reference Periods and Scenarios

Future Climate change was studied for future periods from 2021 to 2099. It is categorized as: 2035s, 2055s and 2085s which are basically 2021-2050, 2051-2070 and 2070-2099 respectively. These time periods were also adopted by the climate assessment report from NOAA (Kunkel et al. 2013). Past fourteen years from 1995 to 2009 were regarded as baseline period in order to compare with above mentioned three future periods. In the next step, climate dataset for three periods were incorporated in a SWAT model to simulate the streamflow for future climate change.

In order to analyze the fracking impact on stream, the current fracking operation of 2012 was applied in the calibrated and validated SWAT model for the evaluation of climate change effect over a period of 2035s. An analysis was limited for fracking and climate change with 2035s assuming that the current trend of hydraulic fracking would remain intact. Typically, baseline and current scenarios were developed to assess the impact on water resources under current level of fracking. The baseline scenario referred to the watershed conditions of 2012 for period 2035s without incorporating water use for hydraulic fracturing. The current scenario utilized watershed and current fracking conditions of 2012 for future period (2035s). Fracking in the other two periods were not evaluated in this study due to tremendous uncertainties of unconventional drilling trends in the future as the continuous operation of hydraulic fracking depends upon the several political and socio-economic factors. Monthly fracking water use was provided in the model from the water use input file. The simulated flow for current fracking trends during this period was compared with the flow without fracking conditions.

Climate Change in the Basin

The bias corrected downscaled precipitation and temperature were utilized for all stations (23 for precipitation and 19 temperatures) to best represent the spatial variability of precipitation and temperature and create better predictions of streamflow for future. The motive behind this study was to evaluate extreme scenarios first, to determine if climate change could have adverse impacts in the future if the current trend of hydraulic fracking continues. This is essential because various GCMs and scenarios may or may not be required if the impact is not significant, even for the maximum scenarios. Therefore, the results were presented under the highest emission scenarios (RCP 8.5).

The hydrologic cycle is mainly influenced by patterns of temperature and rainfall; therefore, the simulated flow pattern over the 21st century was consistent with the variability of rainfall and temperature patterns that could be expected in this century (not shown). This assessment was performed for three future periods against the baseline periods. The percentage exceedance flow taken at the outlet of the watershed indicated that the chances of the occurrence of low flow would be higher in 2055s than 2035s and 2085s for RCP 8.5; however, high flow could be expected in 2055 while using RCP 4.5 (Figure 4-5) and lowest flow could be realized in 2035 while using RCP 2.6. The percentage change in the annual mean, seasonal mean and monthly mean flow at the outlet of the watershed is presented for RCP 8.5 in Figure 4-6. The monthly mean percentage change showed that September might be a stressful month in all three periods as the study showed 12.2%, 12.8% and 21.6% reduction in the streamflow, respectively indicating that water withdrawal in the September month needed to be considered seriously for the water resources management (Figure 4-6a). Results from the early period of 21st century (2035s) shows that the reduction of flow by -5.4% in January, -14.2% in June, -1% in July and -12% in September could be expected. On seasonal scale, mean seasonal flow showed an increase for all periods except summer in the 2035s period (Figure 4-6b) for RCP 8.5, which was consistent with the precipitations trend of the period. An increasing trend was also revealed in the annual mean streamflow of three consecutive future periods (Figure 4-6c), which was consistent with the increasing trend of mean annual precipitations (not shown). The increment was found out to be approximately 38 cms in the 2035s, 46 cms in the 2055s, and 49 cms in the 2085s compared to the baseline annual mean flow. The increase in average annual streamflow, were approximately 15%, 18.2% and 19.3% in the 2035s, 2055s and 2085s, respectively.

The increasing pattern was detected for RCP 4.5 up to 2055s, and the percentage increase in 2085s was slightly less than 2055s (Figure 4-7a). This trend was also observed for seasonal (Figure 4-7b) and annual (Figure 4-7c) flow. A consistently increasing flow pattern was found from 2035 to 2085s for RCP 2.6 (Figure 4-8). The lowest flow was detected in 2035s compared to the remaining other two periods in all the monthly, seasonal and annual scale.

The increment of flow for low flow periods, especially during the later part of the century, showed a positive signal for water resources management. The percentage increase in seasonal and annual scale flow for RCP 4.5 (Figure 4-7) and RCP 2.6 (Figure 4-8) was consistent with the monthly precipitation pattern (not shown).

Low flow conditions (for few months) are possible only under the RCP 8.5 scenario as indicated in the earlier analysis. So, this highest emission scenario was used to evaluate the impact of fracking conditions on stream low flow. Then, in order to evaluate the impact of climate change on the hydrological cycle for the entire watershed, streamflow outlets from all subbasins were systematically compared with baseline period and presented in Figure 4-9. The monthly flow could be expected to increase in all three periods for all scenarios (Figure 4-9) except the possible decrease in low flow for RCP 8.5 (Figure 4-9).

Similarly, thematic maps were created to explain the variation of streamflow in the future compared to baseline (in terms of the percentage change in flow). However, these maps were based on the annual mean and minimum streamflow to spatially represent the percentage change in annual flow volumes across the watershed using the maximum emission scenario (RCP 8.5). While the watershed may experience low flow in the early 21st century (2035s) for specific months, the annual percentage mean change in streamflow showed that the watershed would experience wet conditions in the 2021 to 2050 period (Figure 4-10). Yet, projections for the eastern portion of the Tuscarawas subwatershed, encompassing eastern and western portion of Muskingum watershed, remained drier than other watershed portions in this period (Figure 4-10). During 2055s period, drier portions would be expected on the eastern portion of the Tuscarawas subwatershed region in the same pattern of period 2035s, but the percentage of the wet zone would increase compared to the first 30 years (Figure 4-11).

During 2085s, the wet zone would be expected to increase through a larger extent of the watershed (Figure 4-12), whereas, the drier region would be expected only in the eastern portion of the Tuscarawas subbasins. This analysis concluded that the drier regions could remain more prevalent in the first 30 years than other 50 year periods, and the watershed would get progressively wetter in future time periods.

The annual minimum flow percentages across the watershed are fairly dry in the first 30 years compared to remaining 50 years (Figure 4-13) as some portion of the watershed experienced high flow in this period. Importantly, Figure 4-13 was based on the annual minimum flow, which decreased even though the increasing pattern of annual streamflow was detected. Conversely, the larger wetter regions were experienced for the second 20 year period (2055s) (Figure 4-14). Similarly, progressively larger portion of wetter area with increased percentage difference in minimum flow was detected in the last 30 years period (Figure 4-14). It is interesting to note that 2055s showed the major dry portion in the 1st and 2nd order streams (Figure 4-14), whereas 2085s showed the dry portion in the major stream regimes (Figure 4-15).

Hydraulic Fracking with Climate Change

Similarly, the impact of water withdrawals for hydraulic fracking with the future climate change (2035s) was evaluated over 32 subbasins. The subbasins where current water withdrawals for fracking exist were analyzed, assuming that the fracking trend would likely remain fairly similar in the future. Here, the impact of fracking was not analyzed during the rest of the two periods for two primary reasons: i) it was not sure how the fracking rate would continue in future as this might be governed by socio-economic and political conditions in future; ii) increased streamflow was realized in other periods.

Results revealed no impact on yearly mean flow as compared to the current and baseline scenarios (Figure 4-16). Some impacts were detected on seven days monthly minimum flow (Figure 4-17) in 14 out of 32 subbasins; however, the difference was just greater than 2%. In fact, this study included all the upstream subbasins as further progressively analyzing the downstream node of the streams. Hence, the area of consideration increased in the downstream node but the percentage change in streamflow showed a decreasing trend in those respective nodes. Percentage changes in seven days monthly minimum flow for baseline and current scenario with the increase in drainage area are displayed in Figure 4-18. Maximum changes up to 55% in 7 days low flow (minimum annual) were observed in the watershed if fracking withdrawals are continued on first order streams. The result varied from 3% to 55% on all affected subbasins, indicating the minimum change in a large drainage area. In general, current fracking conditions showed a change of 34% of the total sources with more than 5% change in seven days minimum low flow. Interestingly, all these changes were limited to the first order streams with no impact for higher order streams at all. The same analysis was repeated for RCP 4.5 and RCP 2.6 (not shown). Analysis indicated that the negligible decrease in flow (2% to 3%) was encountered in three out of 32 subbasins while considering RCP 4.5. Also, results concluded a negligible decrease (3% to 4%) in only two subbasins while considering RCP 2.6. No impact was detected in the rest of the subbasins while using RCP 4.5 and RCP 2.6.

Although this analysis suggests possible variability in the hydrological cycle due to future projections of global climate change, impacts from hydraulic fracking and other water resource limitations may be buffered for the later part of the century except for certain months (e.g. September of 2035s). Period 2035s could be critical for sustainable water resource management in some months, especially for the first order streams compared to the other two periods. However, increase in flow could be expected even in 2035s compared to the baseline period. Regardless, it is recommended that planners devise a policy framework that incorporates the appropriate adaptations and mitigation measures to preserve water resources in light of future climate change scenarios, especially during summer seasons of the early 21st century. While climate change studies, including this research, have inherent uncertainty related to future emission scenarios of the greenhouse gasses, land cover changes and energy fluxes, this research constitutes a comprehensive framework for the systematic variation of streamflow in response to future climatic conditions, particularly in a watershed affected by hydraulic fracking.

Conclusion

The potential impact of climate change on streamflow in the Muskingum watershed was evaluated using the MPI-ESM-LR model with RCP 8.5, RCP 4.5 and RCP 2.6 scenario for the 21st century. The research objective was to determine whether the projected global climate change would enhance low flow conditions in the watershed under continued hydraulic fracking in the future. For this, the SWAT model was used to simulate future streamflow using bias corrected downscaled data. The correlation coefficient used to evaluate the performance of various climate models suggested that the performance of MPI-ESM-LR model was one of the best models. This study found a consistent increase in temperature and precipitation for all three time periods as compared to the baseline period, especially for RCP 8.5.

The variation in the streamflow was consistent with the precipitation and temperature patterns of the region. Results concluded that flow would increase in the coming decade as indicated by mean annual percentage increase with 38.3% in 2035s, 46.9% in 2055s and 49.6% in 2085s. However, the analysis on a monthly scale depicted that the coming decade would have a critical reduction of flow during September (low flow period). Similarly, the assessment on a regional scale across the watershed suggested that 2035s would be a relatively dry period among the three modeled periods, characterized by a reduction in streamflow in some months.

Similarly, the assessment of the streamflow using current rate of fracking revealed that the low flow period would be the crucial period over the year as 7 days minimum monthly flow indicated some variation when compared with the lowest flow, either with or without fracking; this effect was negligible in larger order streams but clearly visible in smaller order streams.

References

Abbaspour, K. C. (2007). "User manual for SWAT-CUP, SWAT calibration and uncertainty analysis programs". *Swiss Federal Institute of Aquatic Science and Technology*, Eawag, Duebendorf, Switzerland, 1596-1602.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., and Siebert, S. (2003). "Development and testing of the WaterGAP 2 global model of water use and availability." *Hydrological Sciences Journal*, 48(3), 317–337.

Arnell, N. W., and Gosling, S. N. (2013). "The impacts of climate change on river flow regimes at the global scale." *Journal of Hydrology*, 486, 351–364.

Arnold, J. G., Srinivasan, R., Muttiah, R. S., and Williams, J. R. (1998). "Large Area Hydrologic Modeling And Assessment Part I: Model Development." *Journal of the American Water Resources Association J Am Water Resources Assoc*, 34(1), 73–89.

Benejam, L., Angermeier, P. L., Munné, A., and García-Berthou, E. (2010). "Assessing effects of water abstraction on fish assemblages in Mediterranean streams." *Freshwater Biology*, 55(3), 628–642.

Brekke, L., Thrasher, B., Maurer, E. P., & Pruitt, T. (2013). "Downscaled CMIP3 and CMIP5 climate projections: Release of downscaled CMIP5 climate projections, comparison with preceding information, and summary of user needs." *Technical Service Center, Bureau of Reclamation, US Department of the Interior, Denver*, CO, 1.

Burn, D. H., Buttle, J. M., Caissie, D., Macculloch, G., Spence, C., and Stahl, K. (2008). "The Processes, Patterns and Impacts of Low Flows Across Canada." *Canadian Water Resources Journal*, 33(2), 107–124.

Burton, G. A., Basu, N., Ellis, B. R., Kapo, K. E., Entrekin, S., and Nadelhoffer, K. (2014). "Hydraulic 'Fracking': Are surface water impacts an ecological concern?" *Environmental Toxicology and Chemistry Environ Toxicol Chem*, 33(8), 1679–1689.

Cothren, J., Thoma, G., Diluzio, M., and Limp, F. (2013). "Integration of Water Resource Models with Fayetteville Shale Decision Support and Information System."

Eheart, J. W., and Tornil, D. W. (1999). "Low-flow frequency exacerbation by irrigation withdrawals in the agricultural midwest under various climate change scenarios." *Water Resources Research Water Resour. Res.*, 35(7), 2237–2246.

Ficklin, D. L., Letsinger, S. L., Stewart, I. T., & Maurer, E. P. (2013, December). How do hydrologic projections for the western United States change with the new CMIP5 models?. In *AGU Fall Meeting Abstracts* (Vol. 1, p. 04).

Giorgetta, M. A., Jungclaus, J., Reick, C. H., Legutke, S., Bader, J., Böttinger, M., Brovkin, V., Crueger, T., Esch, M., Fieg, K., Glushak, K., Gayler, V., Haak, H., Hollweg, H.-D., Ilyina, T., Kinne, S., Kornblueh, L., Matei, D., Mauritsen, T., Mikolajewicz, U., Mueller, W., Notz, D., Pithan, F., Raddatz, T., Rast, S., Redler, R., Roeckner, E., Schmidt, H., Schnur, R., Segschneider, J., Six, K. D., Stockhause, M., Timmreck, C., Wegner, J., Widmann, H., Wieners, K.-H., Claussen, M., Marotzke, J., and Stevens, B. (2013). "Climate and carbon cycle changes from 1850 to 2100 in MPI-ESM simulations for the Coupled Model Intercomparison Project phase 5." *J. Adv. Model. Earth Syst. Journal of Advances in Modeling Earth Systems*, 5(3), 572–597.

Hagemann, S., Loew, A., and Andersson, A. (2013). "Combined evaluation of MPI-ESM land surface water and energy fluxes." *J. Adv. Model. Earth Syst. Journal of Advances in Modeling Earth Systems*.

IPCC (2007) In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds). Climate change 2007: *the scientific basis*. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, p 996

Knutti, R., and Sedláček, J. (2012). "Robustness and uncertainties in the new CMIP5 climate model projections." *Nature Climate Change Nature Climate change*, 3(4), 369–373.

Kottegoda, N., and Natale, L. (1994). "Two-component log-normal distribution of irrigation-affected low flows." *Journal of Hydrology*, 158(1-2), 187–199.

Kunkel, K. K., Stevens, L. E., Stevens, S. E., Sun L., Janssen, E., Wuebbles, D., Hillberg, S. D., Timlin, M. S., Stoecker, L., Westcott N. E., & Dobson, J. G. (2013). "NOAA Technical Report NESDIS 142-3." *National Environmental Satellite, Data and Information Service, US Department of Commerce, NOAA.*

Maurer, E. P., Hidalgo, H. G., Das, T., Dettinger, M. D., and Cayan, D. R. (2010). "The utility of daily large-scale climate data in the assessment of climate change impacts on daily streamflow in California." *Hydrol. Earth Syst. Sci. Hydrology and Earth System Sciences*, 14(6), 1125–1138.

Meehl, G. A., Covey, C., Taylor, K. E., Delworth, T., Stouffer, R. J., Latif, M., Mcavaney, B., and Mitchell, J. F. B. (2007). "THE WCRP CMIP3 Multimodel Dataset: A New Era in Climate Change Research." *Bull. Amer. Meteor. Soc. Bulletin of the American Meteorological Society*, 88(9), 1383–1394.

Meehl GA, Stocker TF, Collins WD, Friedlingstein P, Gaye AT, Gregory JM, Kitoh A, Knutti R, Murphy JM, Noda A, Raper SCB, Watterson IG, Weaver AJ, Zhao ZC (2007). "Global climate projections. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: the scientific basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change, chap 10." *Cambridge University Press, Cambridge*, pp 747–846

Janssen, E.E. (2013). "Trends and Projections of extreme precipitation over the contiguous united states." Graduate thesis. http://hdl.handle.net/2142/46718

Moss, R. H., Edmonds, J. A., Hibbard, K. A., Manning, M. R., Rose, S. K., Vuuren, D. P. V., Carter, T. R., Emori, S., Kainuma, M., Kram, T., Meehl, G. A., Mitchell, J. F. B., Nakicenovic, N., Riahi, K., Smith, S. J., Stouffer, R. J., Thomson, A. M., Weyant, J. P., and Wilbanks, T. J. (2010). "The next generation of scenarios for climate change research and assessment." *Nature*, 463(7282), 747–756.

Shrestha, A. (2015). "Modeling Impact of Hydraulic Fracturing and Climate Change on Stream Low Flow: A Case Study of Muskingum Watershed in Eastern Ohio." (MS Thesis, Youngstown State University).

Sharma, S., Shrestha, A., McLean, C. E., & Martin, S. C. (2015). "Hydrologic Modeling to Evaluate the Impact of Hydraulic Fracturing on Stream Low Flows: Challenges and Opportunities for a Simulation Study." *American Journal of Environmental Sciences*, *11*(4), 199.

Smakhtin, V. (2001). "Low flow hydrology: a review." *Journal of Hydrology*, 240(3-4), 147–186.

Taylor, K. E., Stouffer, R. J., and Meehl, G. A. (2012). "An Overview of CMIP5 and the Experiment Design." *Bull. Amer. Meteor. Soc. Bulletin of the American Meteorological Society*, 93(4), 485–498.

Vliet, M. T. V., Franssen, W. H., Yearsley, J. R., Ludwig, F., Haddeland, I., Lettenmaier, D. P., and Kabat, P. (2013). "Global river discharge and water temperature under climate change." *Global Environmental Change*, 23(2), 450–464.

Vorosmarty, C. J. (2000). "Global Water Resources: Vulnerability from Climate Change and Population Growth." *Science*, 289(5477), 284–288.

Vuuren, D. P. V., Edmonds, J., Kainuma, M., Riahi, K., Thomson, A., Hibbard, K., Hurtt, G. C., Kram, T., Krey, V., Lamarque, J.-F., Masui, T., Meinshausen, M., Nakicenovic, N., Smith, S. J., and Rose, S. K. (2011). "The representative concentration pathways: an overview." *Climatic Change*, 109(1-2), 5–31.

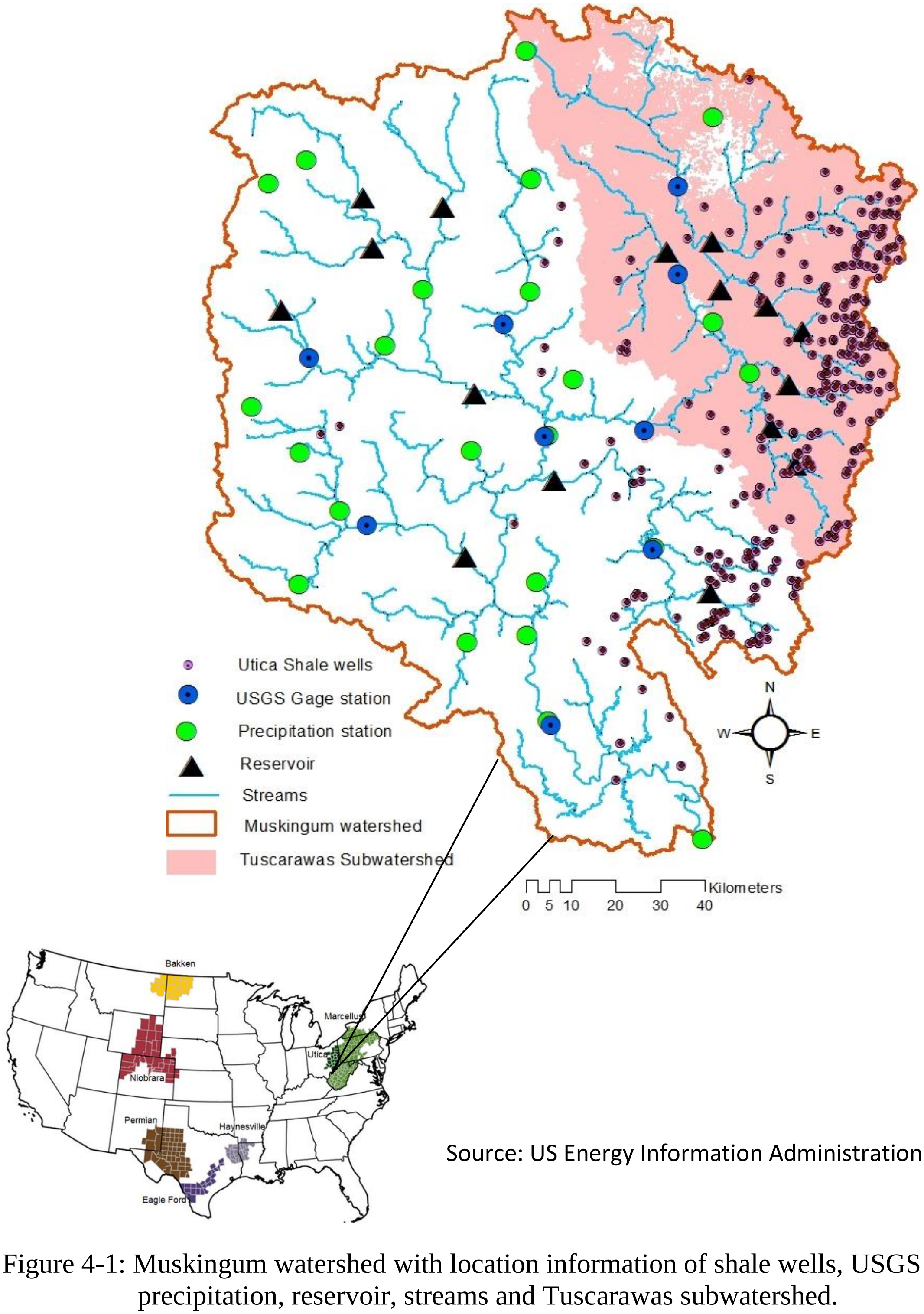
Wilber, D. H., Tighe, R. E., and O'Neil, L. J. (1996). "Associations between changes in agriculture and hydrology in the Cache River basin, Arkansas, USA." *Wetlands*, 16(3),

Table 4-1: Surface Water withdrawal for various primary uses based on facilities in the Muskingum Watershed during year 2012 (Source: Ohio Department of Natural Resources (ODNR)).

Primary Use	**Surface Water (cubic meters/day)**	**Number of Facilities**
Agriculture	220	13
Golf Course	308	38
Hydraulic fracking	334	36
Miscellaneous	677	7
Industry	716	6
Mineral Extraction	2448	16
Public	10516	23
Power	196225	6
Total	211444	145

Table 4-2: The Statistical criteria measuring the performance of the model

USGS Gage Station	**Scale**	**Calibration**				**Validation**			
		R^2	NSE	RSR	PBIAS	R^2	NSE	RSR	PBIAS
3117000	Monthly	0.89	0.89	0.34	-0.4	0.9	0.86	0.37	6.76
	Daily	0.42	0.42	0.76	-0.03	0.45	0.45	0.74	-1.6
3124500	Monthly	0.59	0.53	0.68	-15.6	0.63	0.61	0.62	-9.45
	Daily	0.43	0.4	0.85	-15.88	0.53	0.47	0.73	-9.93
3139000	Monthly	0.64	0.64	0.6	-4.15	0.72	0.71	0.54	-2.65
	Daily	0.51	0.49	0.71	-4.09	0.63	0.6	0.63	-2.7
3036500	Monthly	0.5	0.49	0.72	9.25	0.63	0.56	0.66	17.72
	Daily	0.47	0.46	0.73	9.09	0.41	0.4	0.78	17.75
3129000	Monthly	0.68	0.66	0.59	2.87	0.67	0.55	0.67	20.93
	Daily	0.57	0.56	0.66	2.87	0.54	0.47	0.73	20.82
3140500	Monthly	0.68	0.67	0.58	1.48	0.76	0.69	0.55	12.61
	Daily	0.63	0.63	0.61	1.52	0.69	0.65	0.59	12.53
3146500	Monthly	0.79	0.72	0.53	11.15	0.76	0.71	0.54	12.27
	Daily	0.42	0.4	0.77	11.26	0.43	0.42	0.76	12.24
3142000	Monthly	0.71	0.69	0.56	12.85	0.77	0.68	0.56	-1.16
	Daily	0.55	0.47	0.73	12.91	0.51	0.49	0.71	-2.39
3150000	Monthly	0.73	0.72	0.53	0.31	No Data			
	Daily	0.65	0.65	0.59	0.36				



Source: US Energy Information Administration

Figure 4-1: Muskingum watershed with location information of shale wells, USGS flow gage, precipitation, reservoir, streams and Tuscarawas subwatershed.

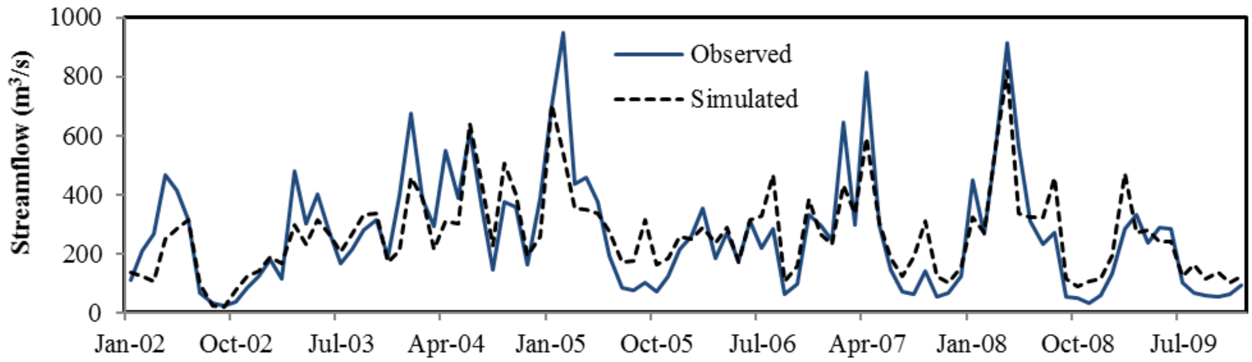


Figure 4-2: Streamflow calibrations at watershed outlet (USGS gage 03150000) from January 2002 to December 2009.

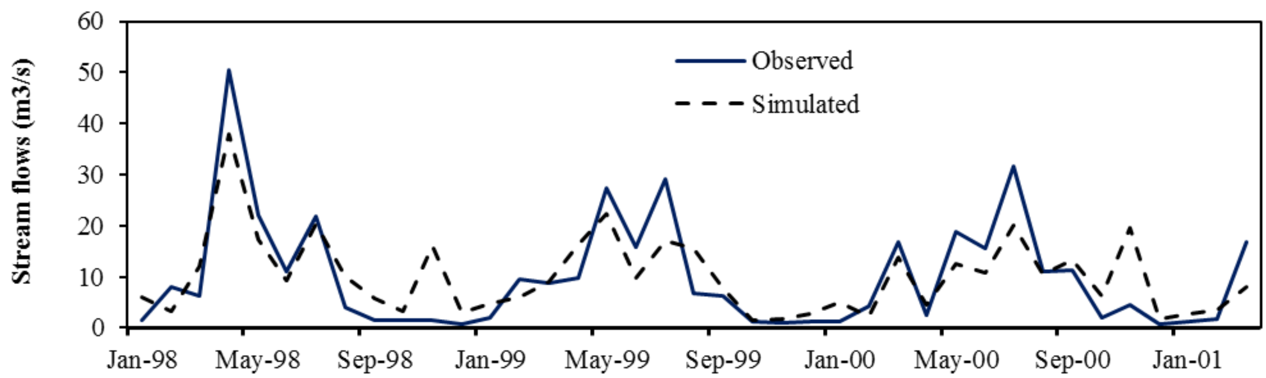


Figure 4-3: Streamflow validation at USGS gage 03142000 from January 1998 to March 2001 (long term data were not available in USGS gage 03150000).

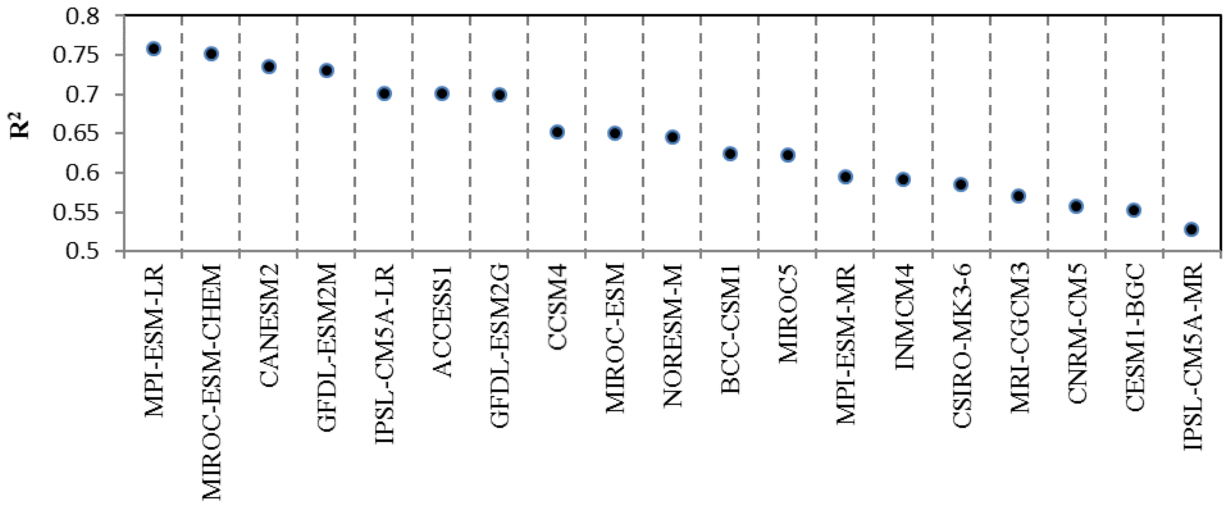
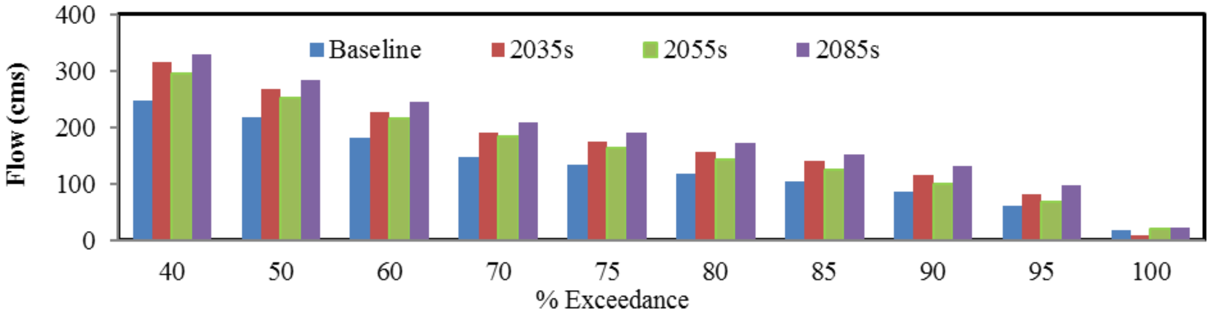
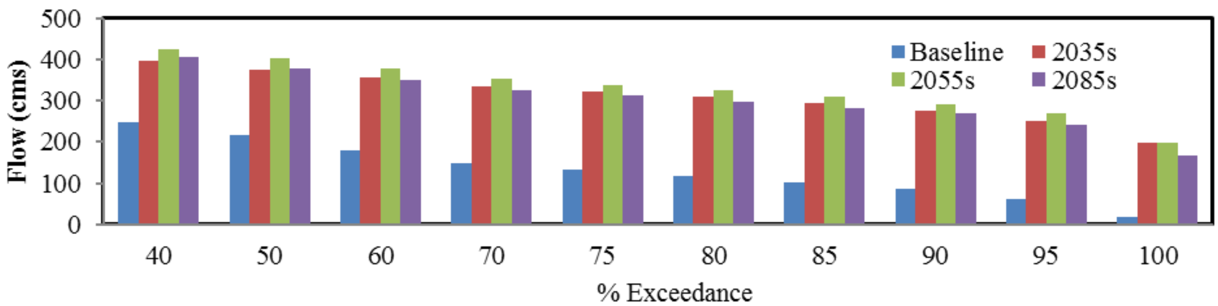


Figure 4-4: Squared correlation coefficient for 19 BCCA models under RCP 4.5 scenario of CMIP5 at precipitation station 00335747.

(a)



(b)



(c)

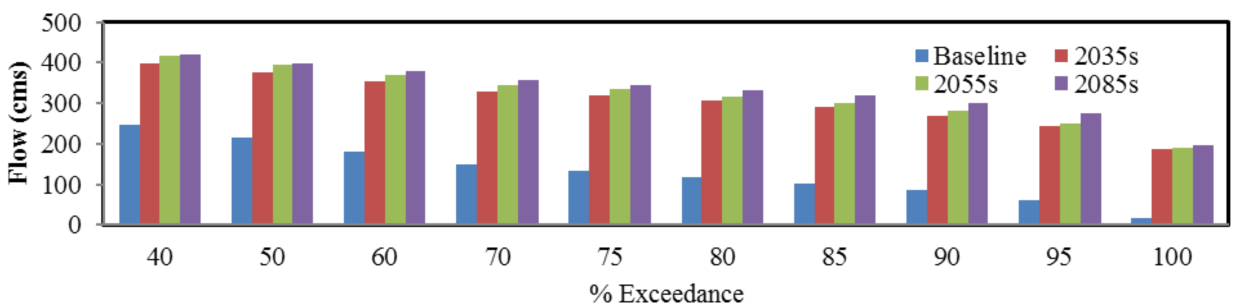
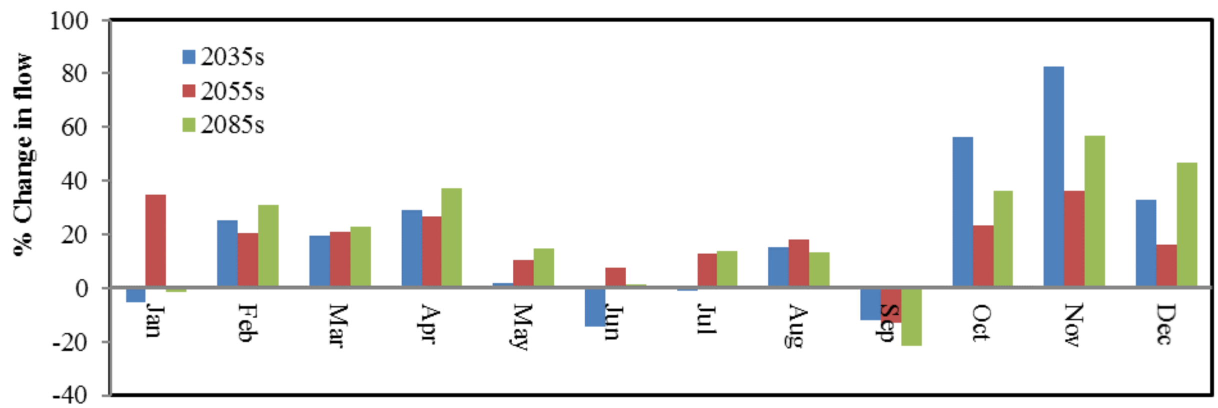
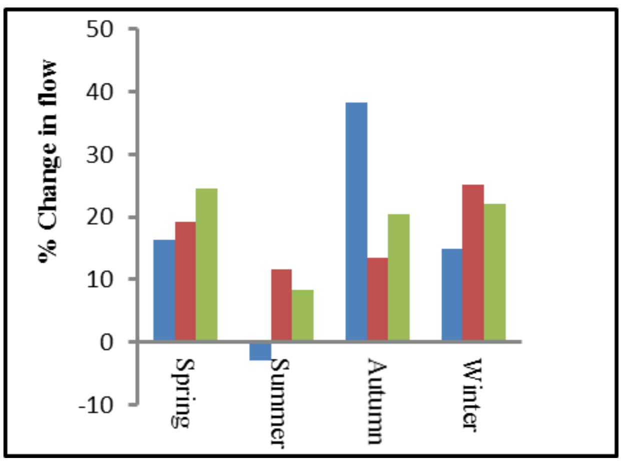
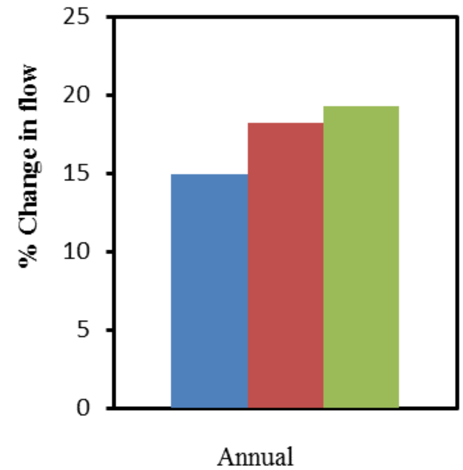


Figure 4-5: Percentage exceedance for mean flow volume for three future periods (2021-2050, 2051 to 2070 and 2070 to 2099) as compared to baseline period (1995-2009) at the outlet of the watershed using RCP 8.5 (a), RCP 4.5(b), RCP 2.6 (c).



(a)





(b) **(c)**

Figure 4-6: Percentage change in monthly mean flow volume for three future periods (2021 to 2050, 2051 to 2070 and 2070 to 2099) as compared to baseline period (1995-2009) at the outlet of the watershed, b) average seasonal flow, and c) average annual flow for similar three periods using RCP 8.5.

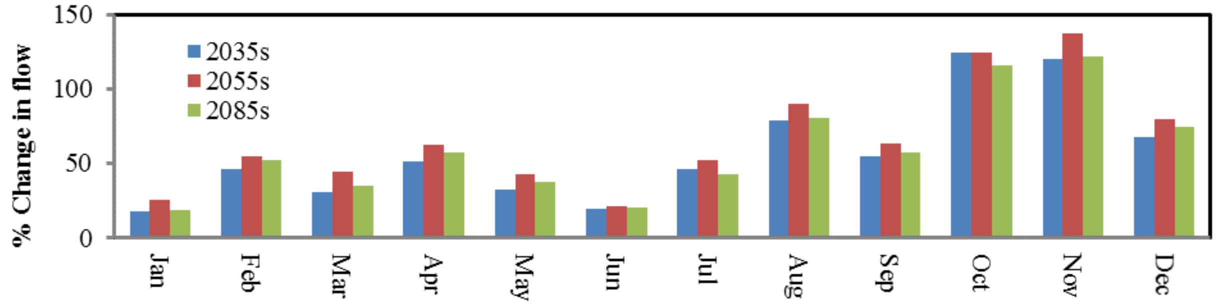
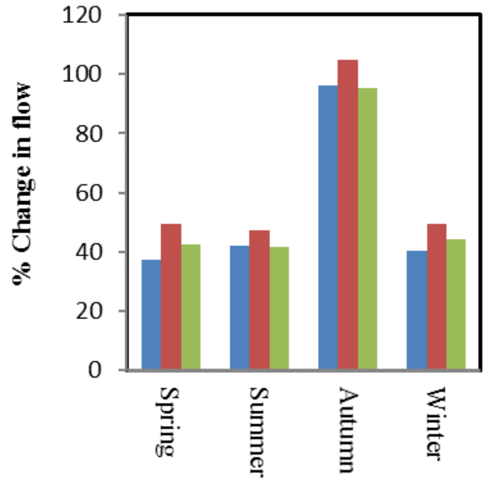


Figure - (a)



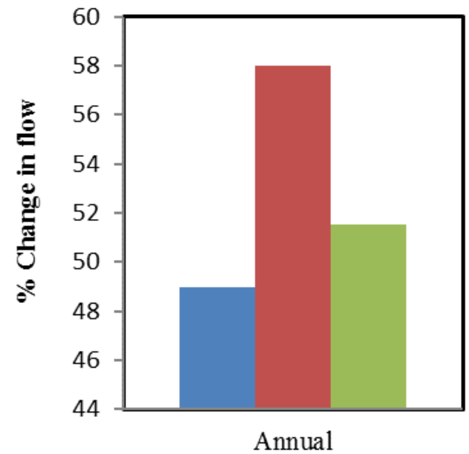


Figure - (b)

Figure - (c)

Figure 4-7: a) Percentage change in monthly mean flow volume for three future periods (2035s, 2055s and 2085s) as compared to baseline period (1995-2009) by MPI-ESM-LR-4.5 at the outlet of the watershed, b) average seasonal flow, and c) average annual flow for similar three periods using RCP 8.5.

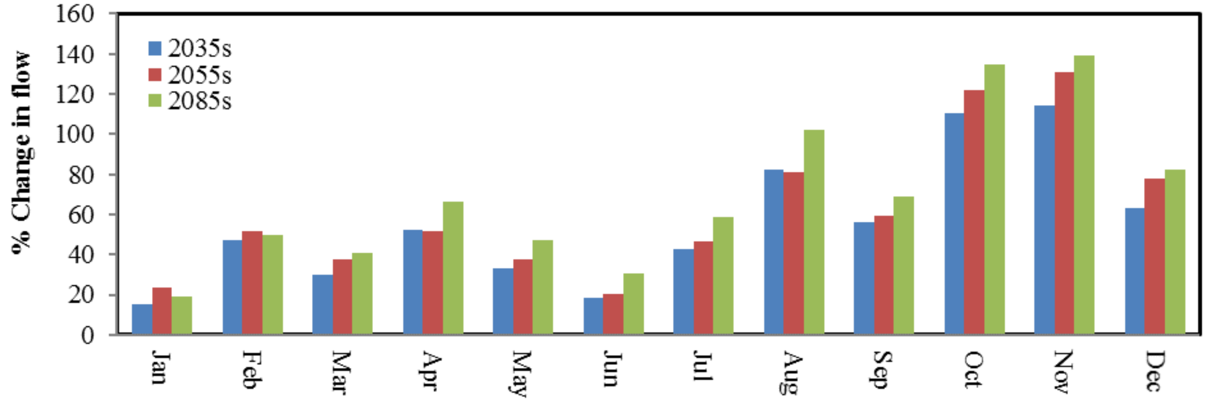
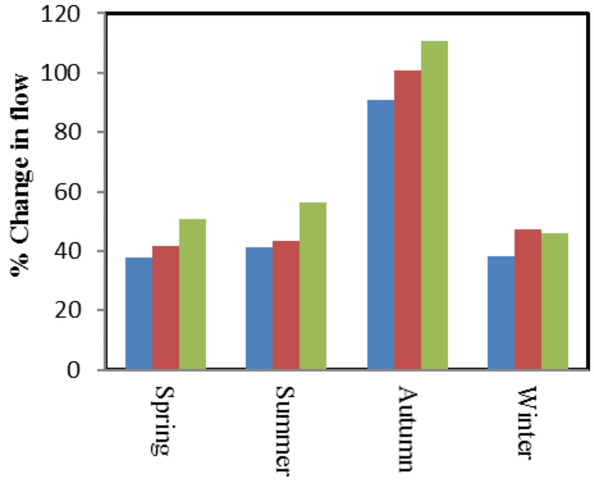


Figure - (a)



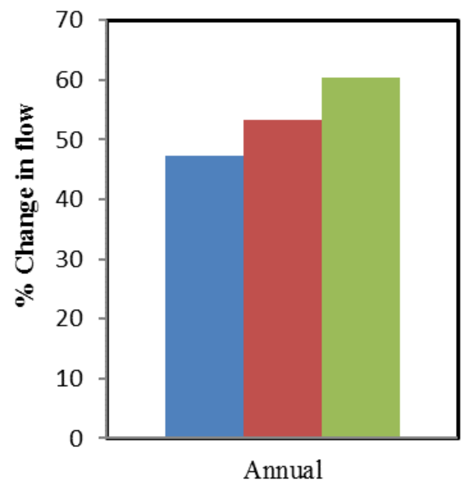


Figure - (b) **Figure - (c)**

Figure 4-8: a) Percentage change in monthly mean flow volume for three future periods (2035s, 2055s and 2085s) as compared to baseline period (1995-2009) by MPI-ESM-LR-2.6 at the outlet of the watershed, b) average seasonal flow, and c) average annual flow for similar three periods using RCP 8.5.

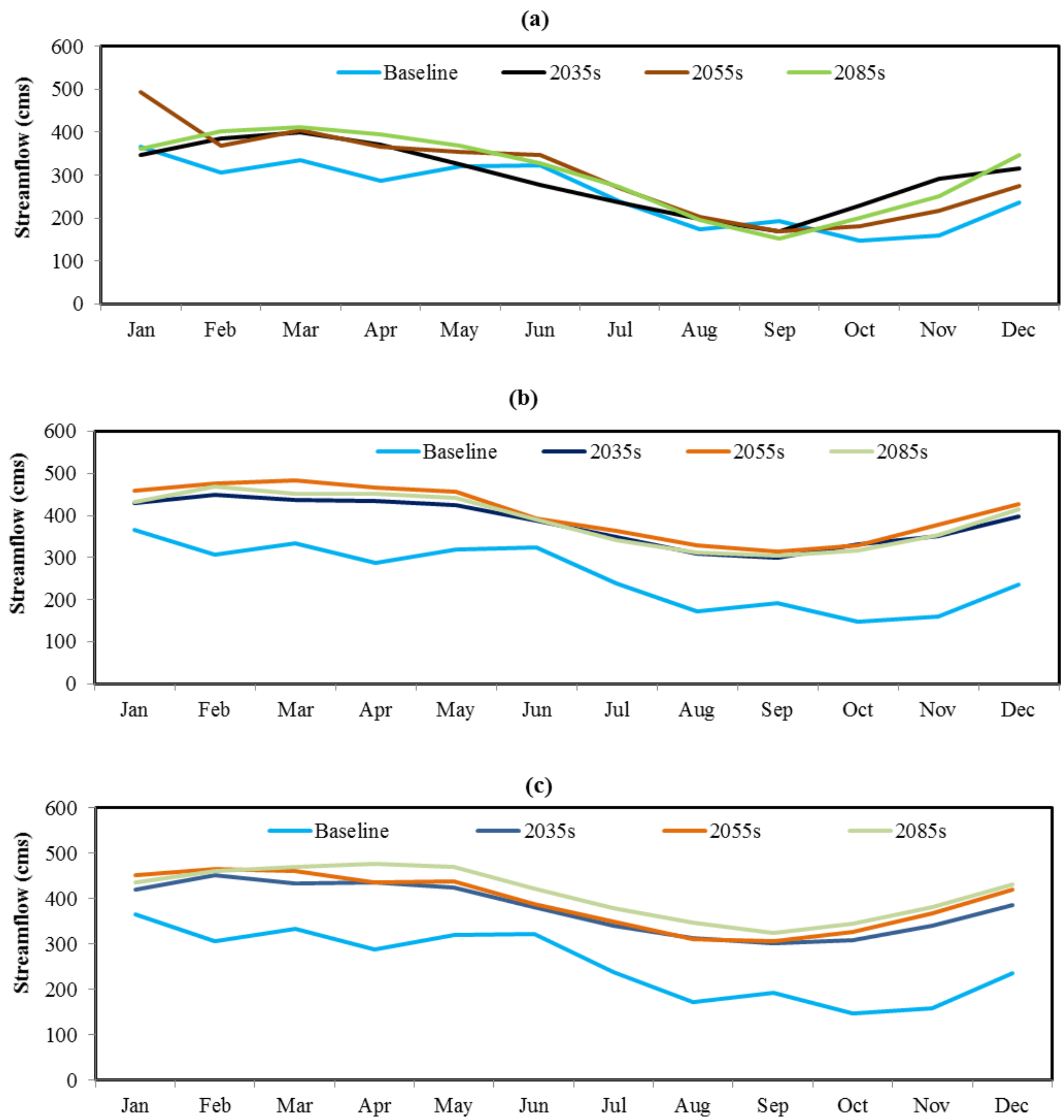


Figure 4-9: Monthly mean flow volume for three future periods (2035s, 2055s and 2085s) and baseline period (1995-2009) at the outlet of the watershed for RCP 8.5 (a), RCP 4.5 (b), RCP 2.6 (c).

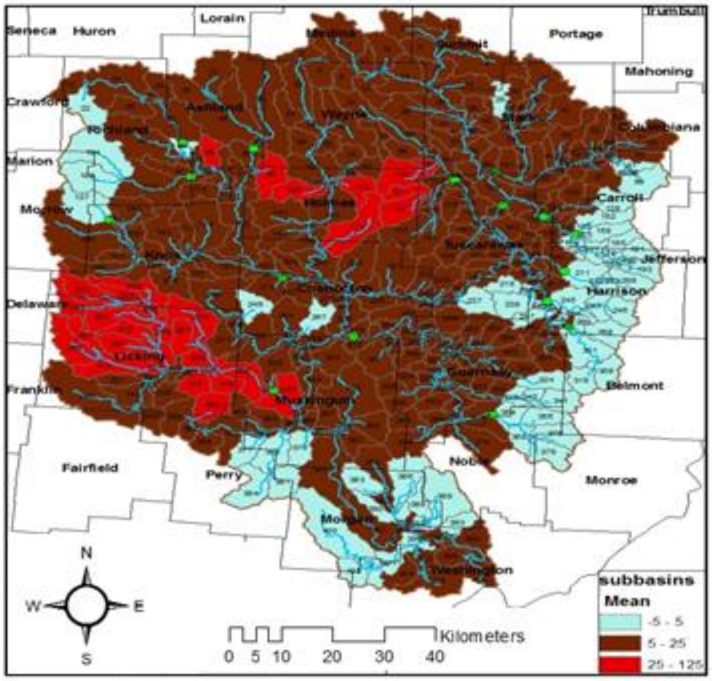


Figure 4-10: Percentage change in annual mean streamflow for 2035s period against baseline period (1995-2009).

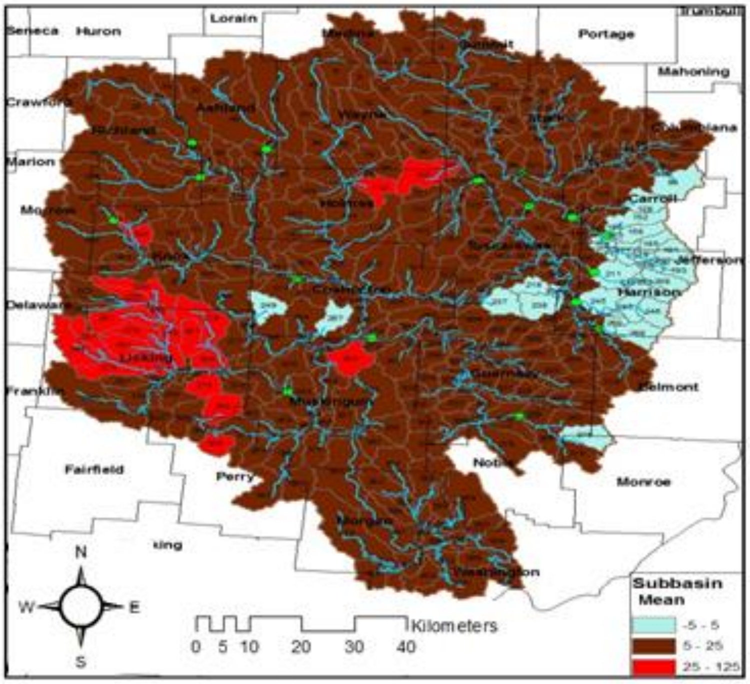


Figure 4-11: Percentage change in annual mean streamflow for 2055s against baseline period (1995-2009).

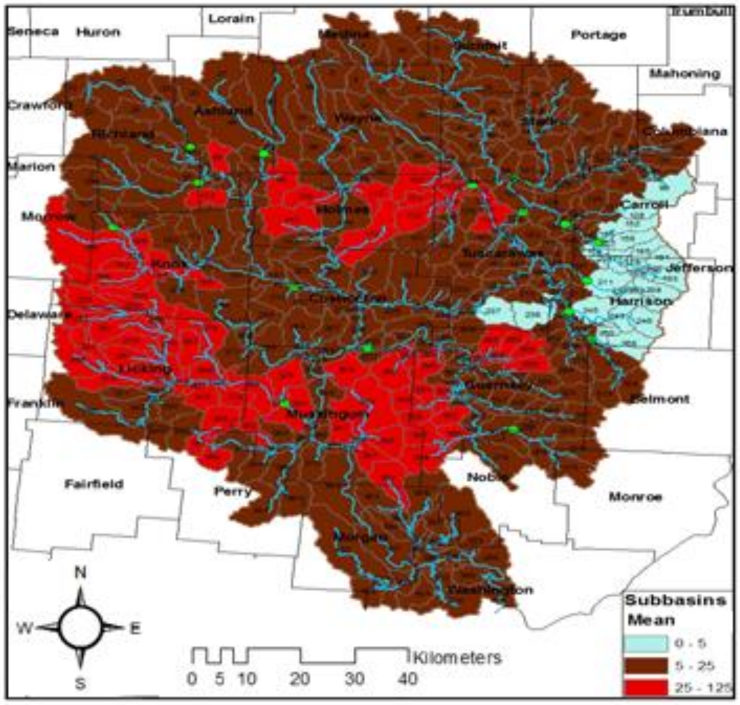


Figure 4-12: Percentage change in annual mean streamflow for 2085s against baseline period (1995-2009).

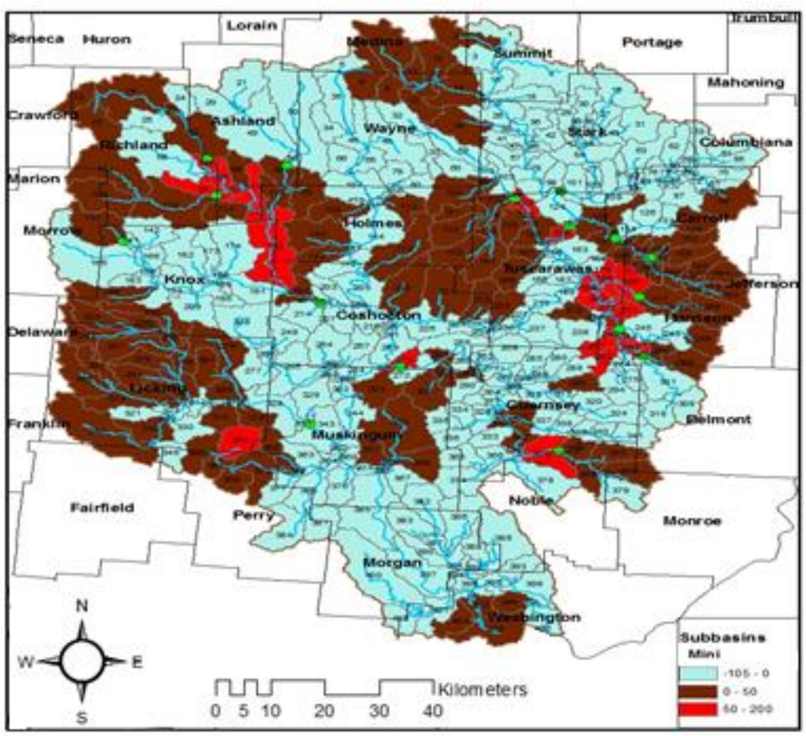


Figure 4-13: .Percentage change in annual minimum streamflow for 2035s against baseline period (1995- 2009).

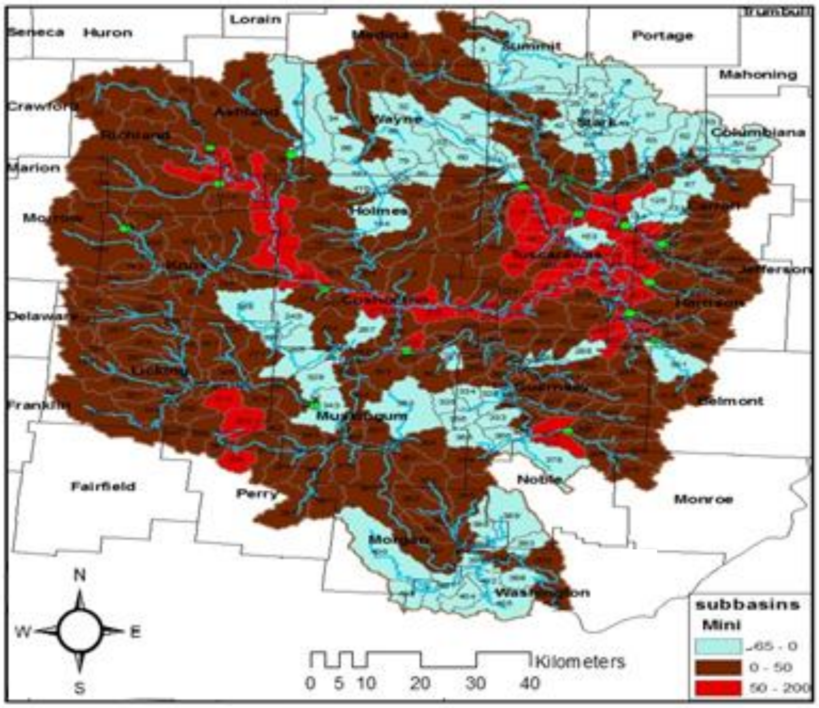


Figure 4-14: Percentage change in annual minimum streamflow for 2055s against baseline period (1995 - 2009).

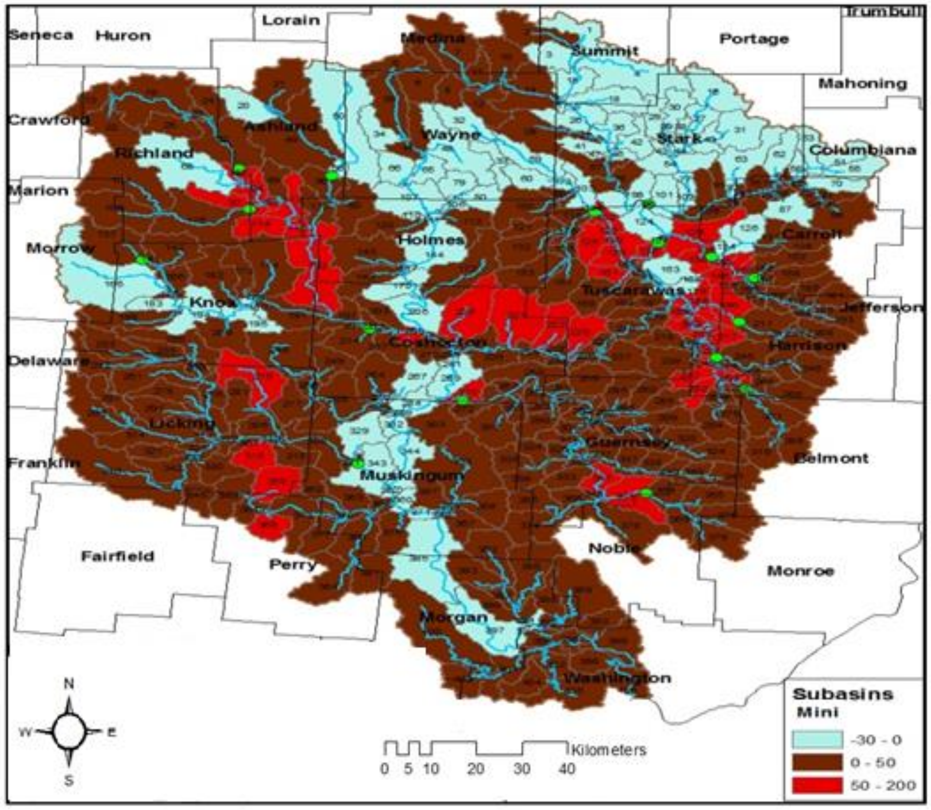


Figure 4-15: Percentage change in annual minimum streamflow for 2085s against baseline period (1995- 2009).

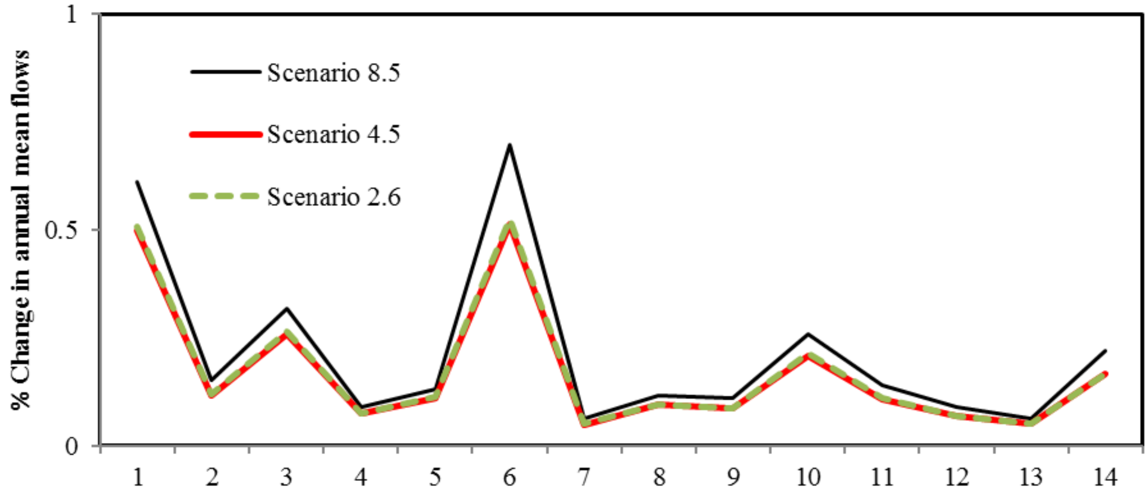


Figure 4-16: Percentage change in annual mean flow for current and baseline scenario during 2035s periods.

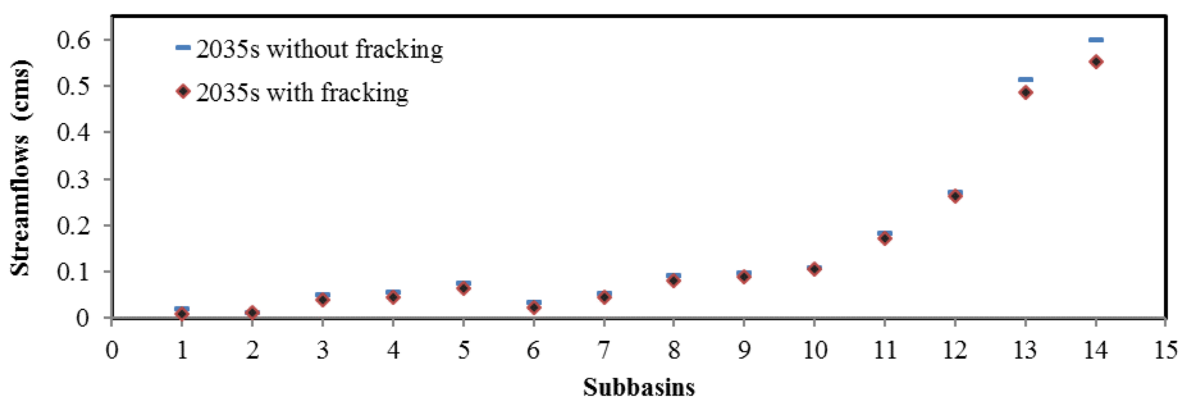


Figure 4-17: Seven days monthly minimum flow (considered the minimum value from each year) for current and baseline scenario during 2035s period using RCP 8.5.

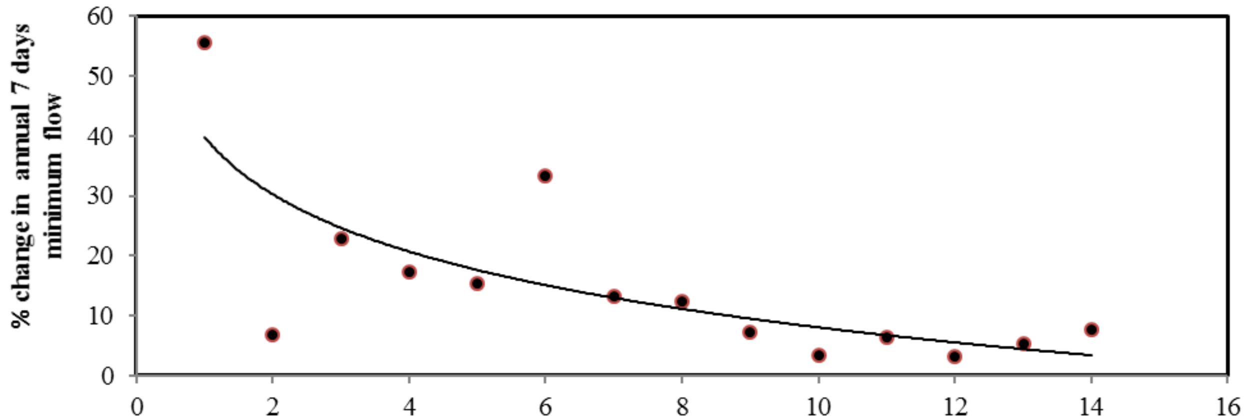


Figure 4-18: Percentage change in 7 days minimum flow for current and baseline scenario (considered one minimum value from each year) during 2021-2050 periods using RCP 8.5 over the 14 subbasins in downstream where the 14th subbasin was largest in area (1589 km2) and first subbasin was the smallest in area (42.88 km^2).

Surface water quality and ecosystem health with shale energy development

Basic Information

Title:	Surface water quality and ecosystem health with shale energy development
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End Date:	2/28/2016
Funding Source:	104B
Congressional District:	Ohio Congressional District 15
Research Category:	Water Quality
Focus Category:	Water Quality, Surface Water, Sediments
Descriptors:	None
Principal Investigators:	Elizabeth Myers Toman, Jiyoung Lee

Publications

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2. C. R. Sheban (student) E.M. Toman, J. Lee. “Surface Water Quality and Microbial Communities in Three Ohio Watersheds.” Poster Presentation. Denman Undergraduate Research Forum, The Ohio State University, Columbus, Ohio. March 25, 2015.
3. E.M. Toman. “Unpaved Rural Roads and Stream Water Quality.” Invited speaker, Water Luncheon Seminar presented by The Water Management Association of Ohio and The Ohio Water Resources Center. April 15, 2015.
4. Sheban, C. R. 2015. Surface Water Quality and Microbial Communities of Three Ohio Watersheds. Undergraduate Thesis for the Honors Program. The Ohio State University.
5. Silliman, B. A. 2015. Interactions and Relationships of Road Born Sediment and Total Sediment Production in a Small Agricultural Catchment. M.S. Thesis. The Ohio State University.

Surface water quality and ecosystem health with shale energy development

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Jiyoung Lee, Division of Environmental Health Sciences & Dept. of Food Science & Technology
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Issue

The development of shale energy in Ohio and throughout the world has dramatically increased in recent years without full understanding of the impacts to surface water quality and, in turn, total ecosystem health. Shale energy exploration and extraction activities, including the construction and traffic use of access roads and well pads, can produce sediment that may runoff to surface waterbodies. Sediments carry metals, chemicals, and nutrients that when released may disrupt aquatic microbial communities and encourage the growth of toxic algae. Besides the disturbance to aquatic ecosystems, increased sediments and associated algae, especially blue green algae, may affect the health of terrestrial systems as humans and animals interact with surface waters.

Research Objectives

The overall goal of this research is to characterize the impacts of shale development to surface waters and ecosystem health. We will accomplish this with two main objectives:
1) Determine the changes to stream water quality with development of shale energy within a watershed in eastern Ohio, and
2) Evaluate the response of microbial communities to environmental stresses, such as changes in sediment loading and water quality.

Methodology

Beginning in June 2014, we installed equipment in perennial streams at the mouth of two small watersheds on property owned and managed by The Ohio State University, OARDC in Noble County Ohio. This equipment continuously measures and records water quality parameters including temperature, dissolved oxygen, conductivity, pH, and turbidity. We surveyed the stream channel at each gauging site and installed equipment that continuously measures flow velocity and depth and uses the channel dimensions to determine stream flow. All collected data are sent via cellular modem to a publically accessible website for viewing or download. Beginning in March 2014 we took monthly grab samples at each site that were transported to an EPA certified lab and analyzed for total suspended solids (TSS). From June to October 2014, additional grab samples from the streams were analyzed for nutrients and microbes.

Findings and significance

This research was designed with the intent of comparing water quality measures and stream microbial communities in paired watersheds before, during and after shale energy development in one of the watersheds (treatment watershed). The timeline for drilling at the study site has been postponed. Although this development is a disappointment to the researchers, we feel confident that we have measured and recorded baseline data regarding water quality parameters and microbial communities in these watersheds and have the infrastructure ready to begin monitoring activities if or when shale energy development at the site begins. The work is summarized below.

ABSTRACT

Counties in Eastern Ohio within the Utica-Point Pleasant shale formation have experienced increased development of shale oil and gas. These operations have created areas of bare soil and dirt roads with high volumes of traffic. Soil erosion from these intensely used sites can lead to greatly increased sediment concentrations in waterways. The objective of this project was to examine how sediments in water affect the presence of microbes. The research also contributes baseline data to a larger project assessing the effects of shale development on water quality. Sedimentation has been recognized as the number one impairment of streams in the U.S., bringing with it negative environmental and human health consequences. Sediments may foster and transport bacteria, chemicals and other microbial organisms. The hypothesis of this project was that increased sediment in stream water would correlate with increased detection of *E. coli*. Baseline testing for cyanobacteria was also conducted. It was predicted that changing aquatic conditions including sediment levels caused by shale development would contribute to a shift in the aquatic cyanobacteria communities and be favorable to the species *M. aeruginosa*. The three adjacent watersheds for this study are located within The Ohio State University, Eastern Agricultural Research Station (EARS) in Noble County, Ohio. The site historically was mined for coal in the 1960s, but was reclaimed through the 1990s and is currently a site for livestock research. Most of the EARS property is contained within a larger basin that drains directly to the municipal water reservoir for the city of Caldwell, Ohio. Water samples were collected five times from each watershed and tested for turbidity, *E. coli*, phycocyanin and chlorophyll *a*. Additionally, real-time qPCR was used to detect genetic markers for human specific fecal contamination, ruminant specific fecal contamination, the phycocyanin intergenic spacer region (PC-IGS) of *Microcystis aeruginosa* and the microcystin synthetase gene A (*mcyA*) of

Microcystis aeruginosa. Finally, microcystin was tested using ELISA. Results show correlations between high turbidity and high *E.coli* counts (R^2=0.50, p = 0.032). A similar linear relationship between turbidity and chlorophyll *a* was also found (R^2 = 0.59, p = .04). *Microcystis aeruginosa* and microcystin were not detected. Further microbial profiling is recommended to gather more information on the overall microbial community structure in conjunction with their transport via sediments.

1. INTRODUCTION

1.1 Sediment

Sedimentation is a natural process that happens when particles settle to the bottom of a water body. Sediments are either suspended in the water column or deposited on the bed of the channel. The amount of suspended and bed load sediment is influenced by the energy and velocity of the water (FEM 2015). Most sediment in surface waters are soil particles (sand, silt and clay) and minerals from surface erosion processes, but may also include decomposing organic matter such as leaf litter or algae. Suspended solids refers to both minerals and organics while suspended sediments refers to just the mineral fraction of the solids load (Ongley 1996). Water bodies naturally contain sediments, but erosion and sedimentation rates have drastically increased due to modern agriculture practices, urban land-use changes, timber harvesting, construction and other anthropogenic disturbances. Sedimentation is the primary cause of stream impairment and the single greatest non-point source pollutant in the United States (EPA 2006a).

Suspended sediment concentration (SSC) is measured by weighing the dry mass of particulate in the water sample (Marquis 2005). This method of measurement requires lab equipment such as a vacuum filtration apparatus, filters and a drying oven. It is time consuming to transport, filter and weigh samples and is inconvenient for long term data. A more common and efficient way to estimate the amount of suspended sediment in water is with turbidity. Turbidity measures the degree to which light is scattered by particles suspended in a liquid in Nephelometric Turbidity Units (NTU) (USGS 2013). Different bodies of water such as streams, rivers or lakes will have different normal turbidity levels depending on the discharge, bedrock, and other conditions (CWT 2004). Dissolved organic materials such as decaying vegetation and humic matter can color the water and also contribute to turbidity (FEM 2015).

Sediments in water pose environmental, industrial and human health consequences. Suspended sediments can interfere with predator prey dynamics by reducing visibility in water (Vinyard et al. 1976) and can irritate fish gills (Marquis 2005). Increased sedimentation can divert traditional migration patterns and disrupt fish spawning by filling in the cracks of gravel beds where many organisms lay their eggs (Coen 1995). Contaminants attached to fine grained sediments are ingested by filter feeder species and biomagnified to higher trophic levels in the food chain (Schubel 1997). Suspended sediment also reduces the amount of sunlight that penetrates the water, allowing for less photosynthetic production of oxygen by underwater plants (Schubel 1997). Particles eroded from agricultural lands carry nutrients such as nitrogen and phosphorous that when released may encourage excessive algae growth. Subsequently, decomposition of organic material such as algae depletes oxygen levels and create hypoxic conditions detrimental to aquatic organisms (Burkholder 1992).

1.2 Sediments and Drinking Water

While it is possible to remove large sediments that cause physical damage to water treatment machinery, many small particles interfere with drinking water cleaning techniques. Sediments shield pathogens from disinfection processes like chlorination and ultraviolet irradiation (Marquis 2005). Turbidity in drinking water was once seen as harmless color, but studies have strongly linked suspended sediment to the presence of pathogens (WHO 2004). Sediments can promote the growth of pathogens by providing them food, shelter and transportation, and can lead to waterborne disease outbreaks (EPA 1999). Other studies have shown correlations between lowering turbidity and the removal of *Cryptosporidium* and *Giardia* as shown below in Figure 1 (LeChevallier et al. 1992).

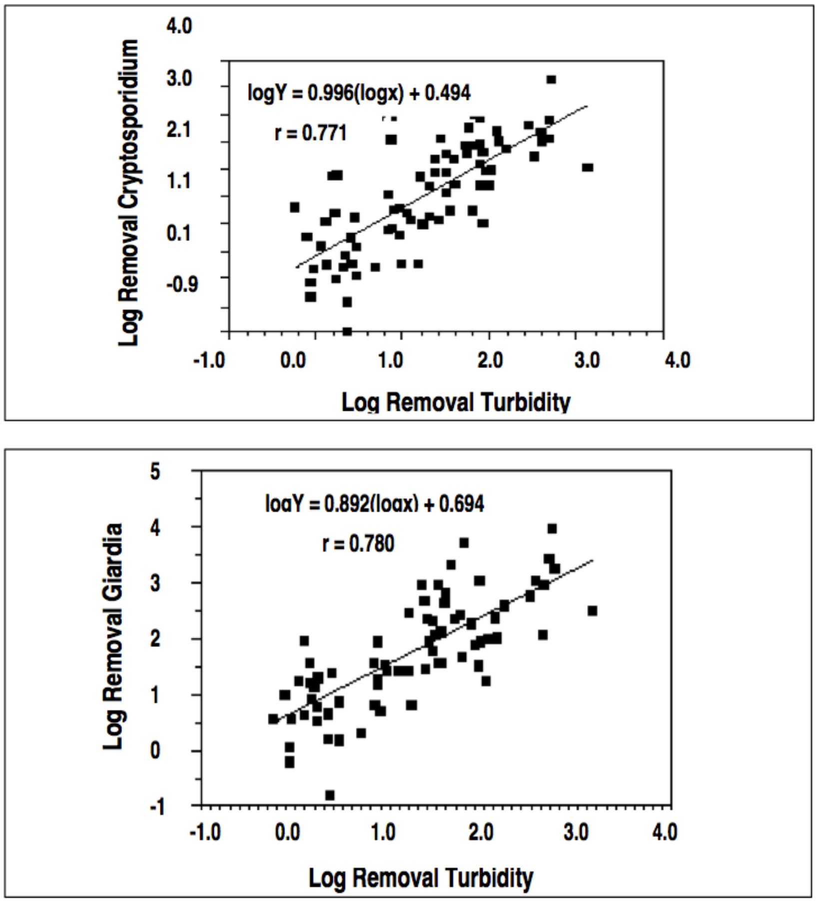

Figure 1. *Relationship between removal of turbidity and removal of Cryptosporidium, and removal of turbidity and removal of Giardia. From* LeChevallier et al. (1992).

1.3 *Escherichia coli*

During 2012, there were 831 foodborne disease outbreaks reported, responsible for 14,972 illnesses, 794 hospitalizations, and 23 deaths (CDC 2012). Some of the most common pathogens that cause outbreaks live in animal digestive tracts and are spread by the movement of water that has been contaminated with fecal matter. A safe and efficient way to test waters for fecal contamination is with an indicator organisms such as coliform bacteria or *Escherichia coli (E. coli)*. The presence of these nonpathogenic indicators proposes that other disease-causing bacteria, viruses and protozoa that also live in animal intestines may be there as well (EPA 2012). Though most *E. coli* strains are harmless, there are Shiga-toxin producing strains (STEC) that are pathogenic, such as O157:H7. These toxic strains originate in the guts of ruminant species, primarily cattle, and were responsible for 5% of the total etiological outbreaks in 2012 (CDC 2014; CDC 2012).

Total coliforms are a group of bacteria that are commonly found in nature and used as a broad indicator of bacterial contamination in drinking water sources. The EPA recommends that *E. coli* be used as an indicator in recreational waters because it is a more fecal-specific coliform bacterium (EPA 2012). Though *E. coli* is still commonly used across the United States, studies have shown that *E. coli* may be an active member of the microbial community and therefore not an effective indicator species. *E. coli* was thought to survive poorly exposed to stresses in the open environment, but has been found in soil, sediment and algal samples. One study repeatedly found isolated *E. coli* that had been outside of an animal host for longer than the bacteria has ever been expected to survive. This may provide evidence that *E. coli* have become "naturalized" members of the soil microbial community (Ishii & Sadowsky 2008).

Microbial source tracking methods have been designed to trace the origin of fecal contamination. There are genotypic, microbiological, phenotypic and chemical methods available. This project used the genotypic method of detecting host-specific genetic markers. This method is advantageous because it does not require culturing the organisms and can differentiate pathogenic properties of specific bacteria (Scott et al. 2002).

1.4 *Microcystis*

Another microbial concern in aquatic systems is cyanobacteria that, when grown in excess, can cause harmful effects (EPA 2015a). Cyanobacteria are generally referred to as 'blue-green algae' due to their coloring and appearance, but are not algae. The blue color comes from other accessory pigments such as phycocyanin, which work with the green pigment chlorophyll *a* to contribute to photosynthesis. Cyanobacteria produce a suite of secondary metabolites including hormones, allelochemicals and toxins that can also cause human health concerns (Carmichael 1992).

The most common cyanobacteria species come from the genus *Microcystis* and are capable of producing microcystin toxins (MC). The toxins are released into water when a bacterial cell dies and ruptures. Ingestion or topical exposure to MCs can have serious health effects on humans, aquatic organisms, birds and land animals. MCs accumulate in fish and bird livers and can be taken up by fish embryos that interfere with development and hatching (California EPA 2009). The ability of *Microcystis* species to utilize nitrogen gas and photosynthesize light has allowed them form symbiotic relationships with plants, animals and other organisms. It also allows them to outcompete other aquatic phytoplankton species. *Microcystis* can be found in extreme aquatic and terrestrial environments as well as a wide range of latitudes, making them the most dominant phytoplankton group in eutrophic freshwater bodies throughout the world (Oberholster et al. 2004).

An increase in sediment load in a body of water can enact a dangerous feedback system of cyanobacterial and algal growth (Vahtera et al. 2007). Sediments carry nutrients (particularly phosphates) in water and under low dissolved oxygen conditions, sediments release the nutrients into the water column (Schubel 1997; NSW 2009). These large loads of what is usually the limiting nutrient allow algae to grow in excess. Cyanobacteria blooms are caused by a variety of factors including nutrient ratios, temperature, light attenuation and organic matter availability and are difficult to predict (EPA 2015c).

Excess algae growth eventually reaches a carrying capacity. Large quantities of decomposing algae absorb oxygen, create even worse hypoxic (low dissolved oxygen) conditions, and allow for even more nutrients to be released from sediments. Hypoxic conditions have also been observed to cause decreased nitrogen levels. With nitrogen limited systems, nitrogen loads carried in urban and agricultural runoff water induce exponential algal growth (Vahtera et al. 2007). This feedback system can spiral to create large-scale algal blooms that have many devastating effects on aquatic ecosystems.

1.5 Quantification of Cyanobacteria and Cyanotoxins

Some species within the *Microcystis* genus produce toxins and some do not. Furthermore, not all *Microcystis* species produce the same levels of toxins, and toxicity can vary over time within a bloom. This means that the quantity of cyanobacteria cells does not directly correlate the quantity of toxin produced. *Microcystis* produces the most toxins of all cyanobacteria, and though there are various kinds of MCs from different strains of *Microcystis*, all have been found to be hepatotoxic with similar signs of poisoning (Sivonen et al. 1990). The most common and most studied freshwater cyanobacteria species is *Microcystis aeruginosa*. The MC that this species produces is a hepatotoxin peptide that causes liver failure and cancer in animals and humans (Carmichael 1992).

Directly measuring *Microcystis* and MC requires significant resources and time. Advanced methods have been developed over the years including a MC-specific enzyme-linked immunosorbent assay (ELISA) that directly detects MC (Chu et al. 1989). More recently, quantitative polymerase chain reaction (qPCR) methods have been used to amplify MC producing genes. The MC synthase complex that controls MC production can be found in a cluster of genes called the *mcy* operon, ranging from genes *mcyA* through *mcyJ* (Sevilla et al. 2008). Detection using qPCR analyses usually test for only one *mcy* gene, though not all *Microcystis* cells have them (Ouellette & Wilhelm 2003).

Advances in technology still require transport time and expensive lab equipment and are not practical for rapid screening of drinking or recreational waters. Indirect measures of MC such as bacteria biomass may be inaccurate since not all bacteria strains and cells produce toxins. Alternatively, non-molecular approaches have been developed such as quantification of phycocyanin pigment via flourometry (Marion et al. 2012). Phycocyanin, a blue pigment that harvests light, is more telling of the presence of *Microcystis* in water than the green pigment chlorophyll *a*, which other algae also contain (Ahn et al. 2002).

1.6 Regulation of Cyanobacteria and Cyanotoxins

Algal blooms and toxicity levels are water quality issues affecting every continent, and HAB occurrence is predicted to increase as the climate warms (EPA 2015c). This poses a human health concern as well as an ecological hazard for wildlife. Cyanobacteria and toxins have been studied for decades, dating back to the early 1980s, but continue to reveal contradictory results. Modern technology has allowed for more accurate detection of *Microcystis* and MC, but their ecological role and rates of growth are still unclear. Environmental conditions such as temperature, light intensity, soluble metals, suspended sediments, available nutrients, nutrient ratios, precipitation events, water flow, pH and seasonal variability have all been studied with many contrary outcomes (Van der Westhuizen and Eloff 1985; Kromkamp et al. 1989; Sevilla et al. 2008; Harding and Wright 1999; Pimentel and Giani 2014; Zhu et al. 2014; Schatz et al. 2007).

The EPA published its latest guidelines and recommendations about cyanobacteria and cyanotoxins in June of 2015. The drinking water standard was set at 1.0 μg/L, a recreational low risk threshold of 6 μg/L, and a recreational moderate risk standard at 20 μg/L (EPA 2015). These are low, short-term risk levels and exposure can cause skin irritation and acute gastrointestinal illness (EPA 2015b).

1.7 Sediments and Microbes

Microbes survive and are transported on sediment particles. The fluvial patterns and sediment type dictate the fate of microbes in alluvial systems. The literature reveals a general consensus that the majority of enteric bacteria in alluvial systems are associated with sediments (Wilkinson et al. 1995; Jamison et al. 2004; Costerson 1978; Auer and Niehaus 1993). There have been a number of field studies that have modeled the re-suspension of sediments during storm flow conditions and the subsequent degradation of water quality (McDonald et al.1982; Sherer et al. 1988; Nagels et al. 2002). Most of the models that have been developed track indicator fecal bacteria such as *E. coli*, but are limited by the many factors that need to be mathematically represented.

1.8 Sediments and Shale Development

Natural gas extracted from deep shale reserves is increasingly in demand in the United States and around the world. Recent technological developments in horizontal drilling have made extraction faster and easier in unconventional oil reserves (Vidic et al. 2013). The Marcellus and the Utica-Point Pleasant formations are the main targets of the approximately 15,000 wells in Ohio that have used hydraulic fracturing since 1990. The number of permits for Utica-Point Pleasant wells in Ohio has skyrocketed from one single well in 2010, to 1,443 wells today (ODNR 2015).

The development of shale can be detrimental to the quality of surface water bodies. Sediments erode from the construction of well pads, installation of pipelines, and from heavily trafficked roads (Kiviat 2013). Erosion is a natural process, but the activities of hydraulic fracturing can elevate levels of sediment in runoff water, thereby affecting surface water. Well pads, pipeline corridors, and gravel or dirt roads, especially used during wet weather, can be major sources of fine sediments in surface waters (Toman et al. 2011).

1.9 Objective and Hypothesis

The objective of this research was to identify a connection between environmental and public health concerns by correlating suspended sediment concentrations with microbial populations in three small streams in eastern Ohio. The hypothesis was that increased suspended sediment concentrations in stream runoff would cause an increased number of microbial cells found. This study measured turbidity at the mouth of three watersheds to see how strongly it correlated with *E. coli*, and how it would affect a shift in the cyanobacteria community. Results will reveal if other microbes associate with sediments in a way similar to *E. coli*. Furthermore, the DNA archived from the water samples will be saved as baseline data for further studies.

2. METHODS

2.1 Study Location

The research was located at The Ohio State University's Ohio Agriculture Research and Development Center's (OARDC) Eastern Agricultural Research Station (EARS) in Noble County, Ohio. Representative of the eastern Ohio landscape, the 2000-acre EARS site is comprised of agricultural land including grazing pastures for sheep and cattle, rolling hills, steep valleys and distinctive soils. The EARS property contains three watersheds that drain into Caldwell Lake and are surrounded by the cities of Caldwell and Belle Valley (Figure 2). This area has no history of cyanobacteria blooms, but the potential use of the site for hydraulic fracturing could create aquatic conditions favorable to their survival. The watershed on the eastern side of the property contains an intermittent headwater stream (Afton Creek) and is characterized by a variety of land uses including agricultural pastures, forest, roads and some residential homes. Middle Creek watershed running through the center of EARS property has similar land use but is much larger (2.13 km^2 as compared to 0.86 km^2 in Afton Creek Watershed). Middle Creek is a 3rd order perennial stream with about half of the drainage area (48%) outside of the EARS property. The watershed on the western edge of EARS (Hickory Creek) has the smallest by area (0.27 km^2) and is fed by springs that are located in an open cattle pasture. The watersheds differ in their size, road density and land use. These variables will be compared to the concentration of sediment measured.

Maps of the watersheds were made in ArcGIS using the NAD 1983 UTM Zone 17N projection, and the watersheds were defined using a USGS interactive map, Ohio StreamStats (USGS 2014). The EARS property boundary was found using Report All Real Estate Portal property parcel maps and confirmed by documents from the Nobel County Auditor, Mike Stritz.

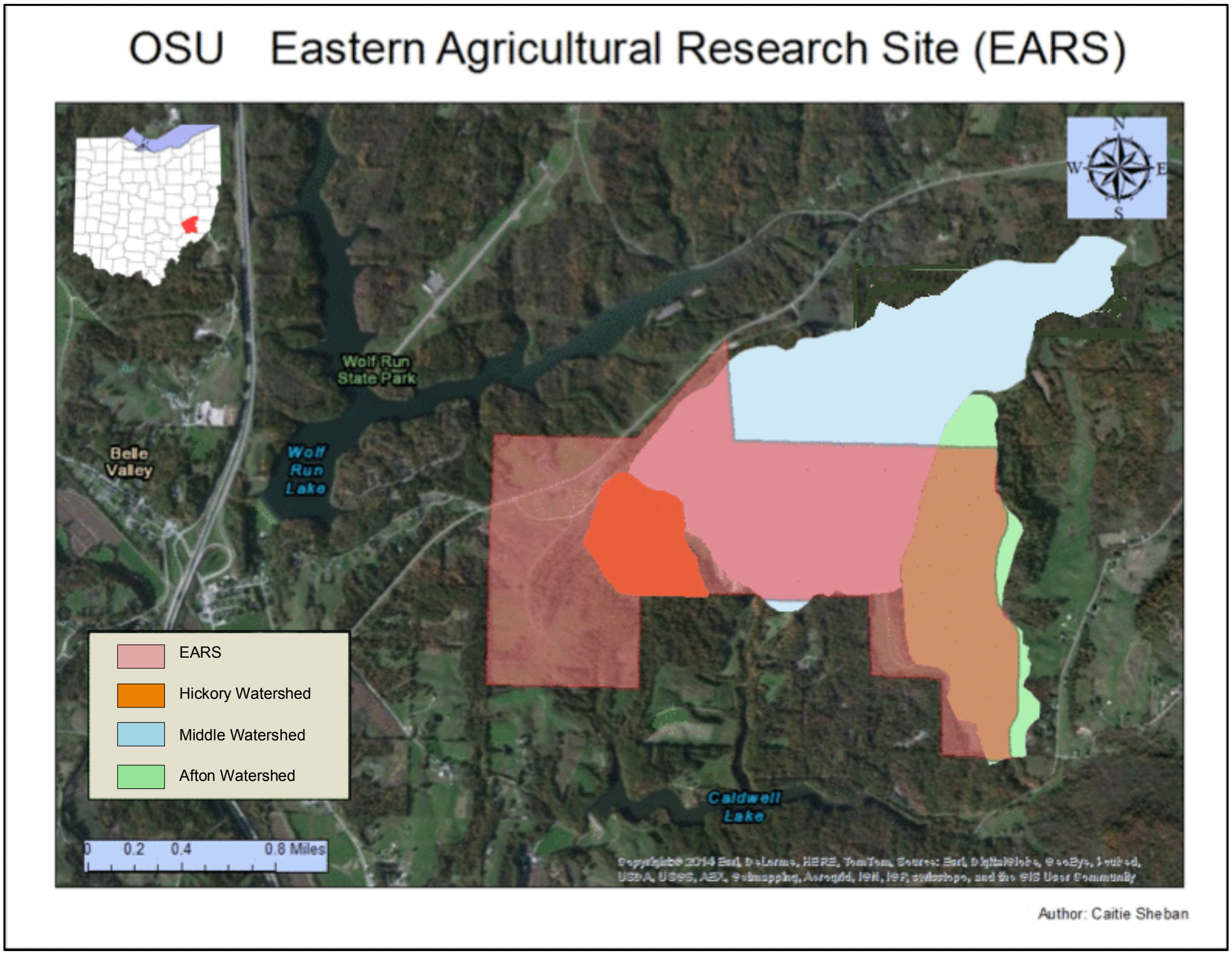

Figure 2. *Boundaries of EARS research site and the three watersheds of interest with an inset of Noble County, Ohio.*

Currently EARS is an active livestock operation with a research focus on management practices for beef cattle and sheep. The animal herds are rotated among pastures within all three watersheds. Service roads inside the property are constructed with unbound aggregate (sand, dirt and gravel with no binding agent) and are used by tractors, four-wheelers and trucks. Road location and land use for the watersheds are shown in Figure 3. Road length, watershed size, and land cover data are summarized in Table 1.

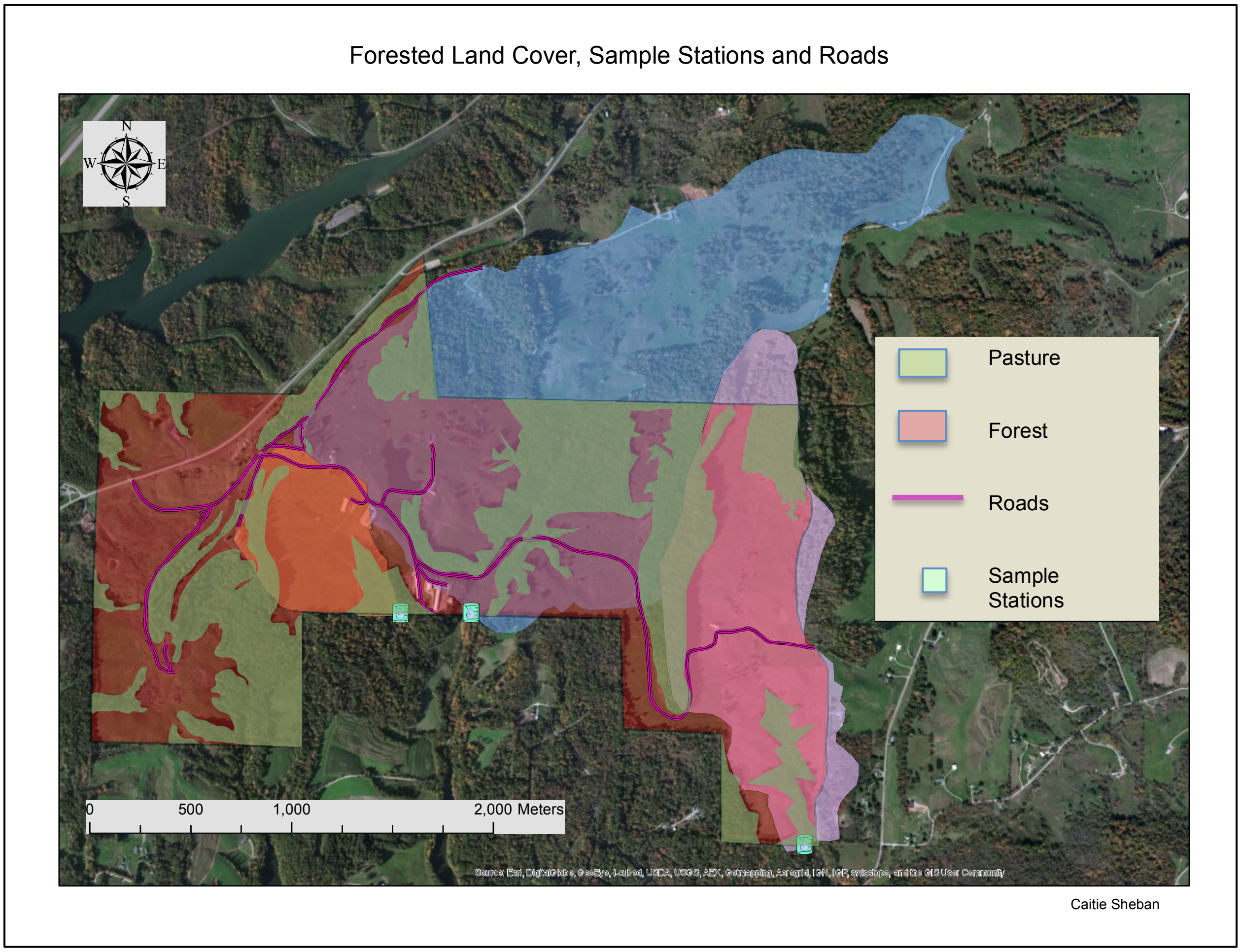

Figure 3. *EARS property boundaries with red highlighting pasture land and green highlighting forest cover. The pink lines represent the service roads and sample stations are marked at the bottom of each watershed.*

Table 1. *Length of road, total area, area in EARS and proportions of pasture and forested land cover for each watershed.*

Watershed	Length of Road (km)	Area (km^2)	Area inside EARS property (sq km^2)	% Forest	% Pasture
Afton	1.87	0.86	0.70	31	69
Middle	2.47	2.13	1.03	42	58
Hickory	0.68	0.27	0.27	24	76

While EARS is not currently being developed for horizontal hydraulic fracturing of shale oil and gas, there are many neighboring properties that have old and active wells. As of February 2015, there were 45 active wells in Noble County, 70 in neighboring Monroe County and 40 in Guernsey County. Figure 4 shows a map of the well sites (white squares) northeast of EARS that are: currently drilling (red), producing oil and gas (green), permitted (yellow) and drilled but not yet producing (gray). The lines represent underground horizontal well paths and the dots represent the bottom hole, or end of the well (uticashaleblog.com 2015).

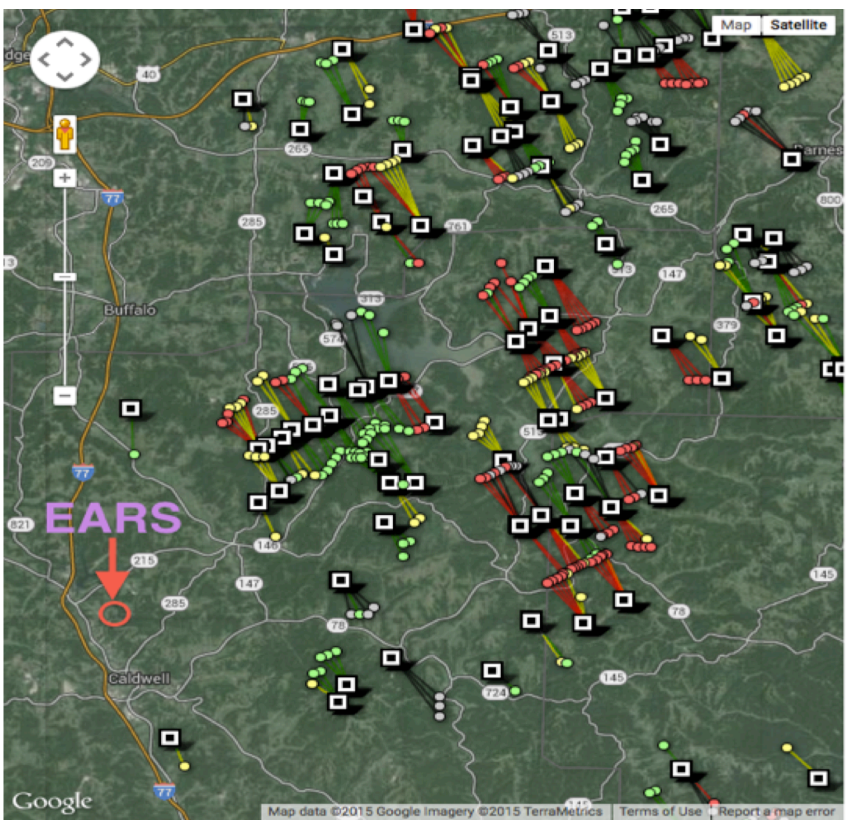

Figure 4. *Current horizontal well development near EARS. White squares represent well pads, lines represent underground horizontal well paths and dots represent the bottom hole. Red means the well is drilling, green means it is producing, yellow means permitted only and gray shows drilling rigs that are drilled but not yet producing. Data updated on February 14, 2015 (uticashaleblog.com 2015).*

2.2 Site Instrumentation and Sample Collection

A gauging station for each watershed was installed at the bottom of the watersheds and was contained by fencing to prevent disturbances from grazing animals. Each station contained a YSI Sonde that collected continuous turbidity, temperature, conductivity and pH of the stream that was used for a larger project. A weather station located on EARS provided precipitation information. Water samples at each gauging station were collected in the summer of 2014 between the months of May and September on six different dates, shown in Table 2. Each sample was collected between 10:00 AM and 2:00 PM.

There was a thunderstorm on sample date June 25, 2014, starting with a light rain in the morning while the samples from Afton and Middle Creeks were collected. It grew into a heavy downpour in the afternoon when the sample from Hickory Creek was collected.

The weather conditions towards the end of the summer were very dry. On the August 28 and September 28, 2014 sample dates, both Hickory and Afton Creeks were not flowing, but a sample was still collected from Middle Creek. The sample from Middle Creek on September 28, 2014 was taken from a stagnant pool near the sample site due to the lack of flowing water.

Table 2. *Precipitation at EARS on each sample date and for the three days before the sample date.*

Sample Date	72 hrs before (cm)	48 hrs before (cm)	24 hrs before (cm)	On Sample Date (cm)
6/18/14	0	0.69	0	0.76
6/25/14	0	0.30	0.05	3.28
7/10/14	0	0.36	1.14	0
7/22/14	1.91	0.03	0	0
8/29/14	0	1.70	0.03	0
9/28/14	0	0	0	0

2.3 Sample Analysis

Each grab sample was measured for turbidity in the field using a HACH turbidimeter. The samples from each watershed were then put on ice and transported to a lab at Ohio State for further testing. Grab samples were collected directly into autoclaved 1000 mL polypropylene bottles from the top of each stream. Due to the shallow depth of the streams, only one sample was collected as to not disrupt settled sediments from the streambed. The grab samples were taken back to the lab and examined within 24 hours using *in vivo* fluorometry for phycocyanin and chlorophyll *a* with a Turner Designs handheld AquaFluor fluorometer and filtered for *E. coli*. A 50 mL tube of each sample was also frozen at -40°C for future reference.

Fluorometers emit certain wavelengths of light and detect wavelengths that are emitted back from compounds. To detect chlorophyll *a*, fluorometers transmit a blue beam of light (460 nm) and detecting the amount of red light (685 nm) fluoresced by chlorophyll *a* in the sample. *In vivo* (meaning "within a living organism") fluorometry measures chlorophyll that are in living algal cells compared to *in vitro* methods that measure chlorophyll after extracting it from the living cells (Turner Designs 1999). Phycocyanin, a blue pigment found in cyanobacteria, absorbs orange and red wavelengths (620 nm) and emits different red wavelengths (660 nm). Studies have shown that the level of fluorescence of the pigment phycocyanin is strongly correlated to cyanobacteria biomass and microcystin concentration (Kim 2013, Marion et al. 2012, McQuiad et al. 2010).

For *E. coli* measurements, 50 mL and 10 mL of water from each sample were vacuum filtered through a 0.45 µm pore size membrane. The membrane was then rolled onto an mTEC agar and traced with forceps to seal it and prevent air bubbles. The agar plates were inverted and incubated at 35°C for 2 hours and then at 44.5°C for 22 hours. The plates were then removed and the magenta colored colony forming units (CFU) of *E. coli* that grew within the membrane boundaries were counted. The number of CFUs for each membrane was back calculated to units of CFU per 100 mL and results for each watershed were averaged. The mean *E. coli* value for each sample is referred to as the *E. coli* density in CFU/100 mL (Marion et al. 2012).

2.4 DNA Analysis

The water samples were stored on ice for approximately 24 hours before testing. The samples were vacuum filtered through a 0.4μm polycarbonate Isopore Membrane ™ that was folded over and stuffed into a capped 2 mL test tube and frozen at -80°C to be used for DNA extractions. DNA extractions from the frozen filters were completed using a Qiagen QIAmp Fast DNA Stool mini kit.

The extracted DNA was tested with real-time qPCR (quantitative polymerase chain reaction) that amplifies, detects and quantifies fluorescence emitted from a reporter molecule that represents a targeted genetic sequence.

To determine if the DNA extractions had any inhibitors such as physical, chemical or biological compounds that may interfere with DNA amplification, a Sketa22 assay was performed. The control consisted of five microliters (μL) of salmon sperm DNA that were added to the water samples. If the expected amount of salmon sperm DNA was not detected with the qPCR, the DNA extraction was diluted and tested again.

The appropriately diluted DNA and a series of specific primers specific were pipetted into a well strips that were centrifuged, cleaned and placed into the real-time qPCR detection machine. The machine measured the number of amplification cycles it takes to reach a set threshold of fluorescence where significant and specific amplification occurs. The threshold cycle number (C_t value) was compared to positive and negative controls to check for inhibition and recorded for each PCR run (Rulli 2012).

PCR analyses were run for DNA sequences of ruminant marker (Rum2Bac), human marker (HF183), a pathogenic toxin produced by a harmful *E. coli* strain (Shiga toxin 2), the phycocyanin pigment intergenic spacer (PC-IGS) of *Microcystis aeruginosa*, and a microcystin production gene (*mcyA*) of *Microcystis aeruginosa*. These procedures were conducted using the protocols found in Bernhard and Field (2000), Converse et al. (2009), Hauhland et al. (2010), Mieszkin et al. (2009), Kurmayer and Kutzenberger (2003), Tillett and Neilan (2000) and Yoshida et al. (2007). Table 3 summarizes the completed lab tests. Any positive data were compiled and analyzed in Excel.

To test for microcystin directly, an Abraxis Microcystin ADDA ELISA Test Kit was used following the procedure outlined in the kit. A series of three incubation periods using different solutions from the kit colored the detection wells if negative. The wells were then analyzed with a spectrophotometer.

Statistical analyses were run using IBM SPSS software (IBM Corp. Released 2013. IBM SPSS Statistics for Macintosh, Version 22.0. Armonk, NY: IBM Corp) and Microsoft Excel.

Table 3. *Summary of laboratory method used for each microbial marker in EARS water samples.*

To Identify	**Ruminant Feces**	**Human Feces**	***E. coli* Toxin**	**Phycocyanin**	***Microcystis aeruginosa***	**microcystin production gene**	**microcystin**
Gene Target	Rum2Bac	HF183	Stx 2	PC-IGS	NIES-843	*mcyA*	-
Method	qPCR	qPCR	qPCR	qPCR	qPCR	qPCR	ELISA

3. RESULTS

3.1 Turbidity

Turbidity measurements taken in the field ranged from 2.2 to 2,313 NTU and are summarized in Table 4. The Middle Creek sample from September 28, 2014 is included in Table 4 but was not used in the following statistical analyses. It was collected from a pool on a dry day during the end of the summer when the water in Middle Creek was not flowing. Afton and Hickory Creeks were dry on August 29 and September 28, 2014.

Table 4. *Turbidity measurements in NTU for Afton, Middle and Hickory Creek watersheds at EARS, Noble County, Ohio between June 2014 and September 2014.*

Sample Date	Watershed	Turbidity (NTU)
6/18/14	Afton	10.8
	Middle	3.1
	Hickory	8.5
6/25/14	Afton	71.2
	Middle	42.1
	Hickory	2313
7/10/14	Afton	11.9
	Middle	7.0
	Hickory	11.0
7/22/14	Afton	6.5
	Middle	3.9
	Hickory	11.3
8/29/14	Middle	2.2
9/28/14	Middle	22.0

3.2 Microbes

Only ruminant and human feces were detected in any of the samples and only ruminant feces could be quantified (Table 7). Results from the lab analyses are summarized in Tables 5 and 6. Human specific fecal contamination was detected in the June 18, 2014 Middle Creek sample and the August 29, 2014 Middle Creek sample, but in amounts below the quantifiable range (1.1×10^{2} – 1.1×10^{6} gene copies/mL) and are not used in the statistical analyses. Significant results were only found for ruminant specific fecal contamination (Rum2Bac). Ruminant specific fecal contamination was only detected in half of the samples (7 of 14) and two were also below the range of quantification (“detected, not quantifiable” denoted DNQ). The June 18, 2014 sample from Afton Creek and the July 22, 2014 sample from Hickory Creek showed much higher gene copy counts (586.2 and 682.2 gene copies/mL, respectively) than the other three creeks on July 10, 2014 (average of 41.4 gene copies/mL). *E. coli* was detected from every sample except the August 29, 2014 sample from Middle Creek due to laboratory complications.

3.3 Pigments

Phycocyanin was detected using the *in vivo* fluorometer (mean = 1.5 μg/L, maximum = 12.4 μg/L, minimum = 0.1 μg/L). Chlorophyll *a* was detected in every water sample (average = 3.0 mg/L, maximum = 13.1 mg/L, minimum = 1.0 mg/L). The three highest values for phycocyanin and chlorophyll *a* were recorded on June 25, 2014.

Table 5. *Summary of results for each microbial marker in EARS water samples.*

To Identify	Ruminant Feces	Human Feces	*E. coli* Toxin	Phycocyanin	*Microcystis aeruginosa*	microcystin production gene	microcystin
Gene Target	Rum2Bac	HF183	Stx 2	PC-IGS	NIES-843	*mcyA*	-
Detection	Yes	Yes	No	No	No	No	No
Quantification	Yes	No	No	No	No	No	No

Table 6. *Chlorophyll a, phycocyanin,* E. coli *and Rum2Bac counts for each sample. DNQ (detected, not quantifiable) for PCR analysis that detected gene copies below the quantifiable range.*

Sample Date	Watershed	*in vivo* chlorophyll *a* (mg/L)	*in vivo* phycocyanin (µg/L)	*E. coli* (CFU/100mL)	Rum2Bac (gene copies/mL)
6/18/14	Afton	1.6	0.7	9.52E+02	586.2
	Middle	1	0.5	3.18E+02	-
	Hickory	1.8	0.6	8.12E+02	12.0 (DNQ)
6/25/14	Afton	4.9	1.6	2.77E+04	-
	Middle	3.3	1.0	4.20E+03	1.0 (DNQ)
	Hickory	13.1	12.4	2.50E+05	-
7/10/14	Afton	1.9	0.5	8.50E+02	25.2
	Middle	1.8	0.6	6.93E+02	59.9
	Hickory	2.4	0.7	1.14E+03	39.1
7/22/14	Afton	2.3	0.2	1.09E+03	-
	Middle	1.4	0.3	2.26E+02	-
	Hickory	2.8	0.4	2.06E+03	682.2
8/29/14	Middle	1.0	0.1	-	-
9/28/14	Middle	22.1	3.1	1.13E+02	-

3.4 Statistical Analysis

Turbidity and *E. coli* densities were compared and found to have a positive relationship (R^2=0.50, p = 0.032). Table 7 shows the summary statistics for turbidity and *E. coli* densities for each sample. The three data points from June 25, 2014 were analyzed separately as rain weather data, and the data from June 18, July 10, July 22 and August 29, 2014 represent dry weather conditions. The median turbidity value during the rainy weather (8.08×10^2 NTU) was over 100 times greater than the dry weather turbidity average (7.6 NTU). The mean *E. coli* density for rainy weather (9.40×10^4 CFU/100 mL) was over 100 times greater than dry weather (9.05×10^2 CFU/100 mL).

Hickory Creek exhibited the highest *E. coli* density but Afton Creek had the highest turbidity (see Table 8). Overall, Hickory Creek had the highest average turbidity and *E. coli*. Figure 5a-e shows the relationship between maximum, minimum and median values for wet and dry data. The median values for the rain data are positively skewed. The box plots in Figure 5 show the different distributions of each data set.

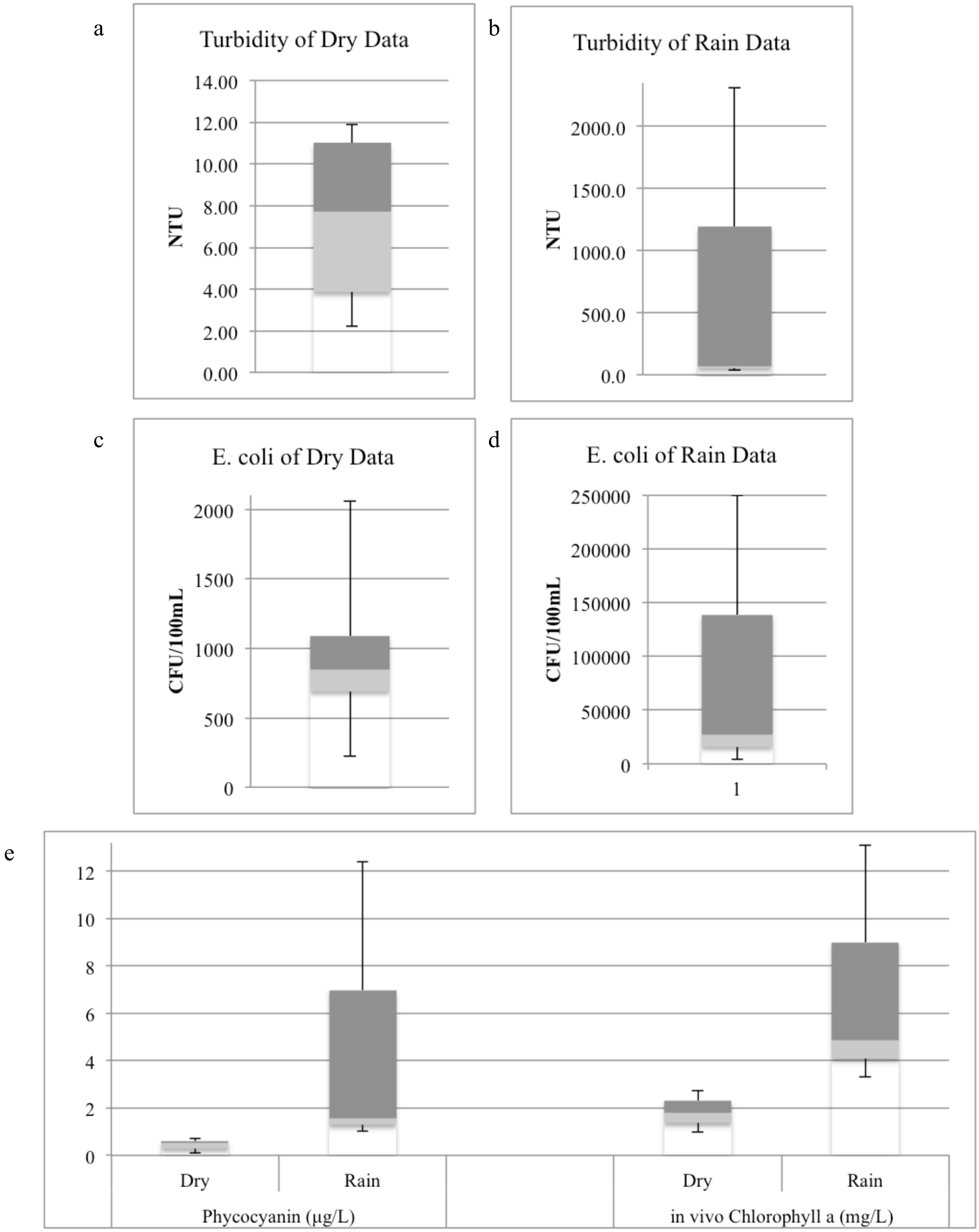

Figure 5. *Box blots of dry and rainy weather data for turbidity,* E. coli*, phycocyanin and chlorophyll* a*. The maximum, 3*rd *quartile, median, 1*st *quartile and minimum are represented by the top error bar, top of box, middle of box, bottom of box and bottom error bar, respectively. Figures 5a and 5b as well as 5c and 5d are plotted separately due to the difference in the y-axis scale*

Table 7. *Summary turbidity measurements in NTU units for all watersheds at EARS on each sample date. Data used for this table exclude the grab sample from Middle Creek on 9/28/14.*

Statistic	**All Data**		**Weather Regime**			
			Dry		**Rain**	
	Turbidity (NTU)	**Mean *E. coli* density (CFU/100mL)**	**Turbidity (NTU)**	**Mean *E. coli* density (CFU/100mL)**	**Turbidity (NTU)**	**Mean *E. coli* density (CFU/100 mL)**
Number of samples	13	12	10	9	3	3
Maximum	2313	2.50E+05	12	2.06E+03	2313	2.50E+05
Minimum	2.2	2.26E+02	2.2	2.26E+02	42	4.20E+03
Mean	192	2.42E+04	7.6	9.05E+02	809	9.40E+04
Median	10.8	1.02E+03	7.7	8.50E+02	71	2.77E+04
Standard deviation	637	7.15E+04	3.6	5.35E+02	1303	1.36E+05

Table 8*. Dry weather turbidity and E. coli measurements summarized by watershed.*

Statistic	**Dry Weather Data**					
	Afton Creek		**Middle Creek**		**Hickory Creek**	
	Turbidity (NTU)	**Mean *E. coli* density (CFU/100mL)**	**Turbidity (NTU)**	**Mean *E. coli* density (CFU/100mL)**	**Turbidity (NTU)**	**Mean *E. coli* density (CFU/100mL)**
Number of samples	3	3	4	3	3	3
Maximum	12	1.09E+03	7	6.93E+02	11	2.06E+03
Minimum	6.5	8.50E+02	1.8	3.18E+02	8.5	8.12E+02
Mean	9.7	9.64E+02	3.9	4.12E+02	10	1.34E+03
Median	11	9.52E+02	3.5	3.18E+02	11	1.14E+03
Standard deviation	2.8	1.20E+02	2.2	2.47E+02	1.6	6.47E+02

3.4.1 Dry Weather Data

Turbidity correlated with *E. coli* the strongest (Figure 6) with a significant positive linear relationship (R^2=0.50, p = 0.032). A similar relationship between turbidity and chlorophyll *a* can also be seen in Figure 7 (R^2 = 0.59, p = 0.04).

3.4.2 Rain Weather Data

The rain data from June 25, 2014 were considered on a logarithmic scale to account for their large variation. The linear relationship between turbidity and *E. coli* for the rain data was not significant (p = 0.95) and had a large R^2 value of 0.98 even after a logarithmic transformation (Figure 8).

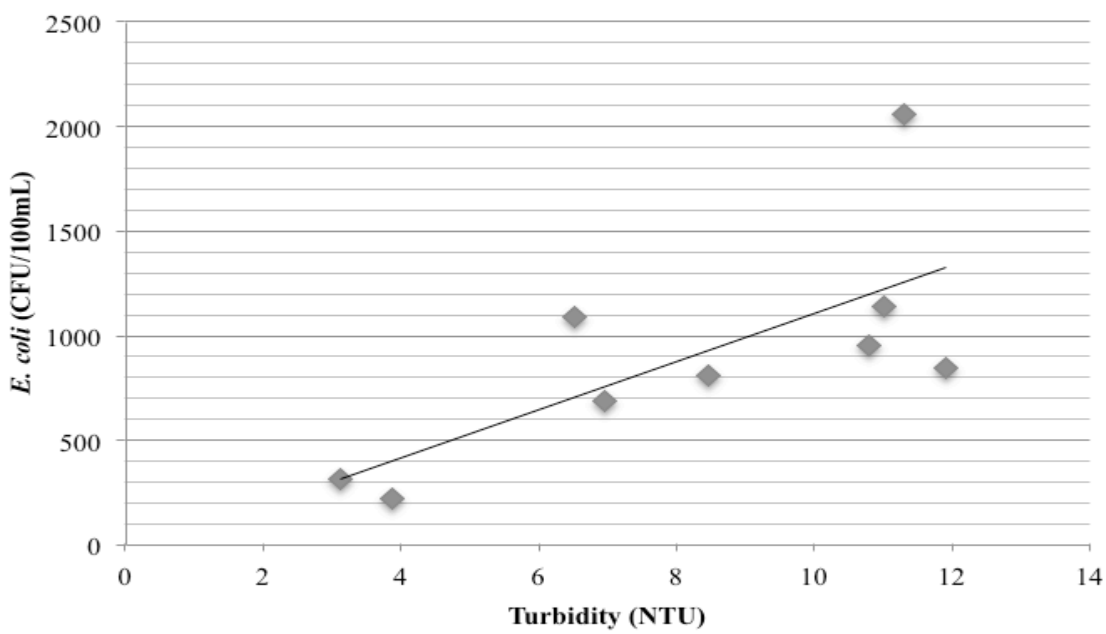

Figure 6*. Dry weather data for turbidity from all three watersheds at EARS plotted against E. coli densities.*

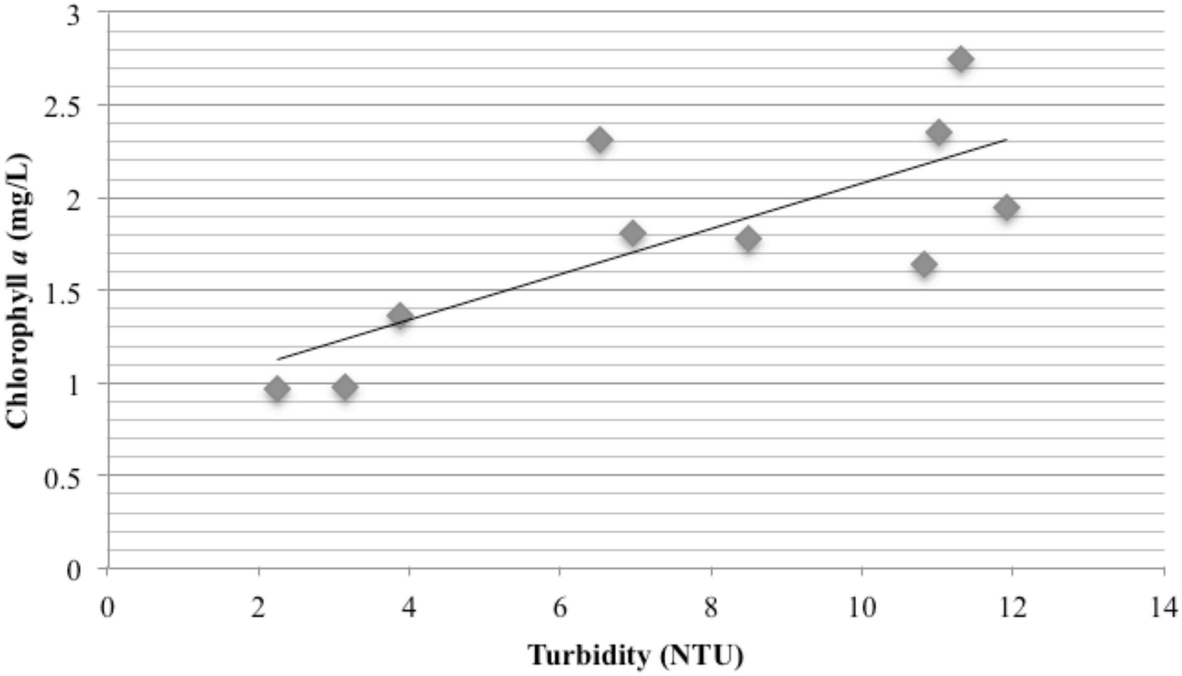

***Figure** 7. Dry weather data for turbidity from all three watersheds at EARS plotted against chlorophyll a.*

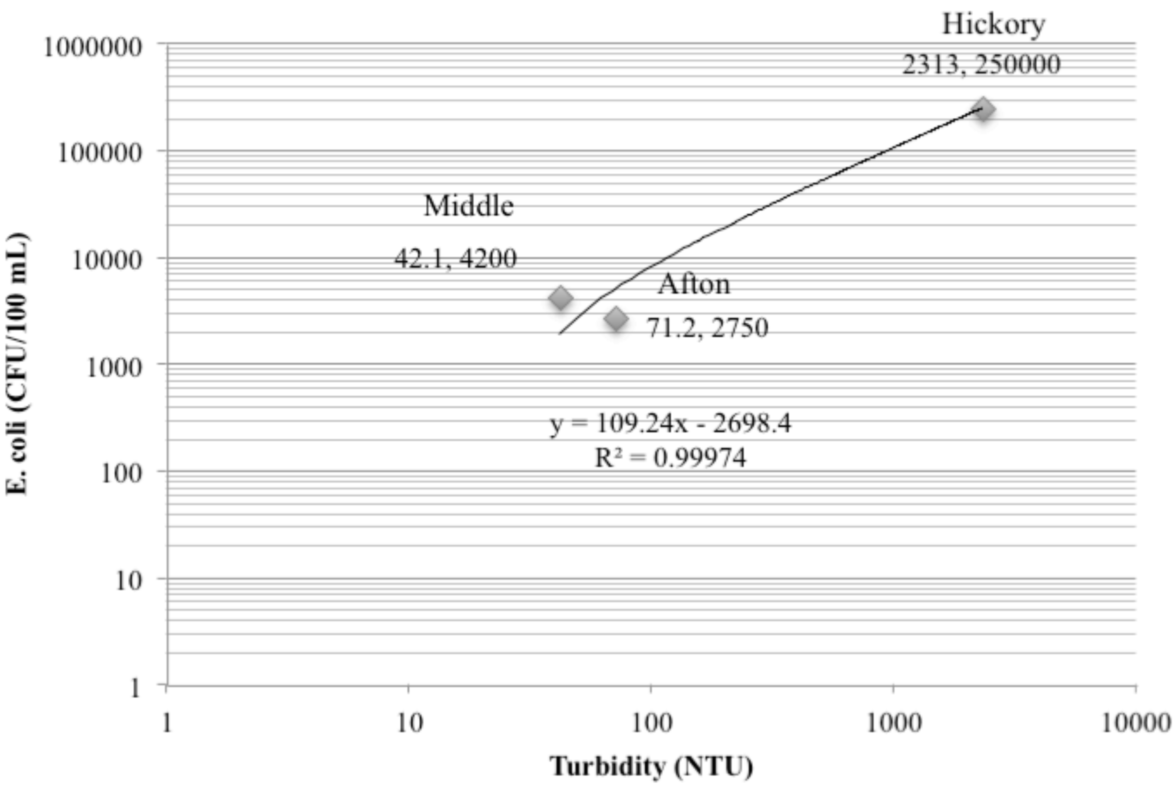

***Figure 8.** Three data points for water samples taken on June 25, 2014 from Afton, Middle and Hickory Creeks representing wet weather data. The Hickory Creek sample was collected after a heavy downpour and flash flood resulting in enormously increased* E.coli *and turbidity. The data are shown on a log-log scale to show the variability during rain events.*

3.5 Land Cover

Middle Creek watershed has an area of 2.13 km^2 and is the largest of the three. Afton Creek watershed is 0.86 km^2 and Hickory Creek watershed is 0.27 km^2. Each watershed was split into pasture and forest land cover in ArcMap software using aerial photos. The percentage pasture of each watershed compared to average turbidity is shown in Figure 9.

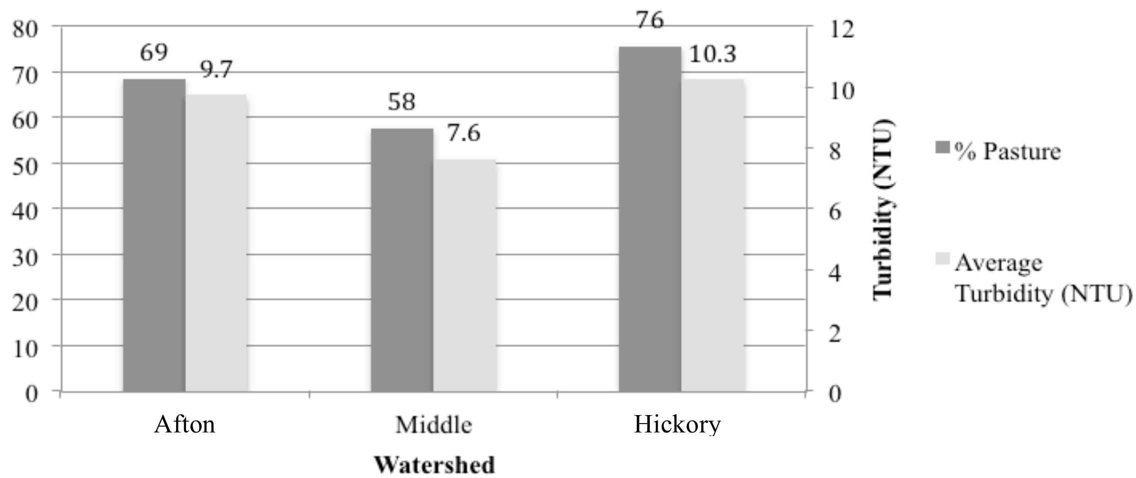

Figure 9. *Average turbidity and percent pasture land cover for each watershed inside EARS. The land cover that is not pasture is predominately forested.*

4. DISCUSSION

4.1 Turbidity, *E. coli*, and Ruminant Associated Fecal Contamination

The water samples collected from the three watersheds inside EARS shows a range of turbidity values from 2 to $2.31x10^3$ NTU, and *E. coli* ranged from $2.26x10^2$ to $2.50x10^5$ CFU/100 mL. The vastly different conditions the creeks experienced during a rain event made the separation of the data into wet and dry weather conditions the most appropriate way to draw conclusions. Sediment concentration has been shown in many studies to be strongly correlated with fecal coliforms like *E. coli* and is divided into stormflow and dry flow (USGS 2012). Linear relationships between *E. coli* and turbidity are site specific, but may be developed so that turbidity measures can be used to screen for elevated contamination. There are many other environmental factors that influence the survival of cyanobacteria and make it much more difficult to model and predict. While summary statistics were provided for the 14 grab samples, they are limited in their meaning. So few data points are not representative of all Ohio weather conditions or flow regimes. There is not enough data during rain events to confirm the relationship between wet and dry results, but due to the variability in precipitation intensities and occurrences in Ohio, much more variability in data for both weather regimes would be expected. Some studies have shown that turbidity and *E. coli* data during storm flow showed the most uniformity regardless of the rain event intensity. This is due to a mixing effect caused by high velocity water in the stream channel (USGS 2012).

Eastern Ohio gets an average of 11.6 cm (4.6 inches) of rain during the month of June, which is 0.39 cm a day (Current Results Nexus 2015). On June 25, 2014, 3.3 cm of rainfall was recorded in an event characteristic of an Ohio summer storm. Ohio experiences thunderstorms approximately 35-45 days a year, most commonly between April and September. Ohio thunderstorms can drop many inches of rain in a few hours producing flash floods that can cause serious damage in the southeastern Ohio hilly terrain (Schmidlim 1996). The elevated turbidity and *E. coli* data collected on June 25, 2014 give insight to the effects of a typical rain event at EARS on the amount of sediment discharged into the streams. More data during high flow rain events would help confirm what the relationship is between turbidity and *E. coli* at such extreme levels. Data for a variety of rainfall amounts is important to create a regression equation that can be used to predict *E. coli* densities.

Wet weather samples are expected to have highly elevated turbidity values caused by water that runs off over the landscape and carries soil particles. *E. coli* organisms living in animal feces in the pastures attach to sediment particles that drain into the streams. High velocity flow in the stream channels may also re-suspended sediments and organisms along the streambeds and contribute to increased turbidity and *E. coli*.

E. coli is an indicator of general fecal contamination and was found in all of the tested samples, but the ruminant specific genetic marker was only detected in half of the samples. Ruminant specific fecal contamination was detected in all three watersheds on July 10, 2014, and the concentrations detected were within a range of 25 to 60 gene copies/mL (whereas other detected concentrations on June 18 and July 22, 2014 were 5.86×10^2 and 6.82×10^2 gene copies/mL, respectively). There is no information recorded about the animal rotations that certainly have an impact on fecal concentrations, but there was 1.14 cm of precipitation at EARS the day before that may have washed and settled over the landscape, contributing to the similar concentrations in each watershed.

4.2 Turbidity, Phycocyanin and Chlorophyll *a*

Chlorophyll *a* was also found to have a significant relationship with turbidity ($R^2 = 0.59$, $p = 0.04$), which supports the idea that floating photosynthetic organisms may also largely contribute to water cloudiness. It may also support the suggestions that sediments (represented by turbidity) carry nutrients that catalyze phytoplankton growth.

Chlorophyll *a* is the primary pigment for all photosynthetic organisms and can be used to represent total algal biomass (USGS 2008). Cyanobacteria exclusively produce the pigment phycocyanin in conjunction with chlorophyll *a*. For this reason, phycocyanin can be used in water monitoring indices as an alternative measure for microcystin levels (Marion et al. 2012). By measuring both chlorophyll *a* and phycocyanin, the relative amount of total algae and cyanobacteria could be compared to the direct measures of *M. aeruginosa* and microcystin. If a stable relationship between the two could be confirmed, chlorophyll *a* and phycocyanin could be used as a rapid and practical index to screen for microcystin levels in the watersheds. These indices may be of interest to the City of Caldwell that uses Caldwell Lake as a municipal drinking water source, and Wolf Run State Park for recreational monitoring.

Turbidity and phycocyanin had a much weaker and insignificant correlation (R^2=0.19, p = 0.24). Turbidity may not be the sole indicator of cyanobacteria growth due to the sensitivity of cyanobacteria to other environmental conditions such as temperature, light intensity, nutrient ratios and pH, as discussed in the introduction. Chlorophyll *a* and phycocyanin exhibited a weak linear relationship (R^2=0.06, p = 0.50) and were likely occurring in separate species. Phycocyanin was measured at much lower quantities (µg/L) compared to chlorophyll (mg/L), and may have been degraded to quantities below the detection range.

4.3 Microbes and qPCR

The phycocyanin intergenic spacer gene (PC-IGS) was not detected. The PC-IGS sequence is specific to *M. aeruginosa*, but can be found in both toxic and non-toxic strains of the organism. The ELISA that tested for the toxin microcystin directly also showed no detection.

Inhibitors in the DNA extracts may explain the lack of detection of PC-IGS. There are many steps during the process of DNA extraction that are susceptible targets for inhibitors. Inhibition can be caused by the loss of sample nucleic acid via absorption to surfaces of tubes and pipets, degradation of primers by nucleases, or by contamination of ethanol used for cleaning. Some samples may also have naturally occurring inhibitors such as polyphenols (tannic acid) or humic substances (fulvic and humic acids) from decomposing plant matter (Pennington 2014). Table 8 below shows the dilutions of the DNA extractions that were necessary for qPCR analyses. The high dilutions needed for some of the samples may have hindered detection of the desired DNA sequences, but the samples that were not diluted also showed no detection. The samples that needed to be diluted did not correlate with high turbidity values, which might imply the inhibitions were not physical. It is most likely that there were no *Microcystis aeruginosa* cells in the water.

Table 8. *Turbidity values for each grab sample and the DNA extract dilution necessary to eliminate inhibitors for qPCR analysis. There is no obvious correlation between turbidity and inhibition.*

Sample Date	Watershed	Turbidity (NTU)	Dilution
6/18/14	Afton	10.8	-
	Middle	3.1	1/500
	Hickory	8.5	-
6/25/14	Afton	71.2	-
	Middle	42.1	-
	Hickory	2313.0	-
7/10/14	Afton	11.9	-
	Middle	7.0	-
	Hickory	11.0	-
7/22/14	Afton	6.5	1/1000
	Middle	3.9	1/5000
	Hickory	11.3	-
8/29/14	Middle	2.2	1/5000
9/28/14	Middle	22.0	1/500

4.4 Land Cover

The highest average turbidity matches the highest relative pasture cover and the lowest average turbidity in the watershed with the lowest percent pasture. Highest and lowest average turbidity also match the smallest and largest watersheds, respectively. Watershed size and land cover type likely both contribute to turbidity.

Hickory Creek watershed had the highest average turbidity. This is likely because it was the smallest watershed with the shortest stream. The water deposited into the stream likely experienced less natural filtration over the landscape and the sediments had less residence time in the water. It also had the highest percent pasture, which means the landscape has smaller, more uniform vegetative cover and likely more animal traffic.

4.5 Possibilities for Further Studies

Analysis of the general microbial community would have been a more appropriate starting point before testing for specific species. Microbial profiling is recommended going forth to better understand what genera are most prevalent in the EARS watersheds.

Further information about the materials being transported and suspended in the water could be obtained from a soil survey. This would help explain the naturally existing microbial community and give insight to the minerals and chemicals expected at EARS. It would also be helpful to include information about the rotations of animals grazing between watersheds. Currently, livestock movement is controlled by the farm managers at EARS who do not log such specific daily activity.

Finally, more samples over a variety of weather conditions would help to draw more concrete conclusions about the patterns and relationships of turbidity and *E. coli* at EARS. This study used turbidity to represent sediment, but it would be more accurate to measure total suspended solids (TSS). By measuring both parameters for each water sample, a relationship can be confirmed so that turbidity measures can be transformed into more accurate TSS measurements.

5. CONCLUSIONS

This study supports existing research that sediments in water foster microbial species. The linear relationship between turbidity and *E. coli* confirm the hypothesis that increased sediment concentrations in water correlate to an increased number of microbes and potentially enteric pathogens. There are many different environmental factors that affect the natural microbial community in a watershed that could also be explored. The results of this study focused on *E. coli*, a common indicator species of fecal contamination. Fecal contamination is not only a problem in waterways near cattle farms, but in agricultural areas all around Ohio. Many disease-causing pathogens are transported via the fecal to oral route and travel far distances in waterways. Animal manure is commonly used as a soil fertilizer, but the nutrients and organic matter that are beneficial to crop growth also pollute the surrounding waterways. Nutrients can catalyze algal growth, ammonia is toxic to fish and the decomposition of organic matter reduces dissolved oxygen in the water necessary to support other aquatic life (Mancl & Veenhuizen 2015).

Though specific microbial species were not identified, this study demonstrated the patterns of sediment production due to precipitation and road use within a small watershed. This research was part of a larger project that focused on how road use produces sediments that wash into the creeks. This highlights the seriousness of road quality, material and structure. Construction projects in the area including shale development activities should be acutely aware that roads are a large source of stream contamination in a typical rural Ohio watershed.

There was no detection of Microcystis in the three watersheds, but harmful algae blooms are still a concern for many inland lakes in Ohio. Lakes in Eastern Ohio do not typically experience HABs, but the potential increase in salinity, sediment concentration and temperatures caused by shale activities may provide conditions favorable for HABs. The proximity of EARS to recreational water and drinking water sources make it an important study location to monitor for the growth of harmful species. EARS provides a unique opportunity for many studies to monitor baseline conditions of an Ohio watershed.

REFERENCES

Ahn, C.Y., Chung, A.S., Oh, H.M. (2002). Rainfall, phycocyanin, and N:P ratios related to cyanobacterial blooms in a Korean large reservoir. *Hydrobiologia* 474:117-124.

Bernhard, A. E., Field K. G. (2000). A PCR assay to discriminate human and ruminant feces on the basis of host differences in bacteroides-prevotella genes encoding 16S rRNA. *Applied and Environmental Biology*. 66(10): 4571–4574.

Burkholder, J.M. (1992). Phytoplankton episodic suspended sediment loading: phosphate partitioning and mechanisms for survival. *Limnology and Oceanography*. 37(5): 974-988.

California Environmental Protection Agency (CEPA). (2009). Microcystins: a brief overview of their toxicity and effects, with special reference to fish, wildlife, and livestock. Ecotoxicology Program, Integrated Risk Assessment Branch, Office of Environmental Health Hazard Assessment. Available at < http://oehha.ca.gov/ecotox/documents/microcystin031209.pdf>

Carmichael, W.W. (1992). Cyanobacteria secondary metabolites—the cyanotoxins. *Journal of Applied Bacteriology* 72: 445–459.

Centers for Disease Control and Prevention (CDC). (2002). Survelience for Foodborne Disease Outbreaks 2012 annual report. Available at <http://www.cdc.gov/foodsafety/pdfs/foodborne-disease-outbreaks-annual-report-2012-508c.pdf> Accessed March 30, 2015.

Centers for Disease Control and Prevention (CDC). (2014). *E.coli (Escherichia coli)* general information. Available at <http://www.cdc.gov/ecoli/general/> Accessed March 30, 2015.

Chu, F.S., Huang, X., Wei, R.D., Carmichael, W.W. (1989). Production and characterization of antibodies against microcystins. *Applied and Environmental Microbiology* 55(8): 1928-1933.

Clean Water Team (CWT). (2004) Turbidity Fact Sheet, FS-3.1.5.0 (Turb). The Clean Water Team Guidance Compendium for Watershed Monitoring and Assessment, version 2.0, Division of Water Quality, California State Water Resources Control Boad, Sacramento, CA. Available at <http://www.waterboards.ca.gov/water_issues/programs/swamp/docs/cwt/guidance/3150en.pdf> Accessed December 11, 2014.

Coen, L. D. (1995). A review of the potential impacts of mechanical harvesting on subtidal and intertidal shellfish resources. *Marine Resources Research Institute*, SC Department of Natural Resources, Charleston, SC.

Converse, R. R., Griffith, J. F., Noble, R. T., Haugland, R. A., Schiff, K. C., & Weisberg, S. B. (2012). Correlation between quantitative PCR and culture-based methods for measuring enterococcus spp. over various temporal scales at three California marine beaches. *Applied and Environmental Microbiology 78*(4): 1237–1242. DOI:10.1128/AEM.07136-11

Current Results Nexus. (2015). Average rainfall for Ohio in June. Data collected from 1981 to 2010 for the NOAA National Climatic Data Center. Available at <http://www.currentresults.com/Weather/Ohio/precipitation-june.php> Accessed April 15, 2015.

Entrekin, S., Evans-White, M., Johnson, B., Hagenbuch, E. (2011). Rapid expansion of natural gas development poses a threat to surface waters. *Frontiers in Ecology and the Environment* 9(9): 503-511.

EPA. (1999). Guidance manual; turbidity provisions. Available at <http://www.epa.gov/ogwdw000/mdbp/pdf/turbidity/chap_07.pdf> Accessed April 3, 2015.

EPA. (2006a). Wadeable stream assessment: a collaborative survey of the nation's streams. Washington, DC: EPA. EPA 841-B-06-002.

EPA. (2006b). Toxicological reviews of cyanobacterial toxins: anatoxin-a, cylindrospermopsin and microcystins (LR, RR, YR and LA) (external review drafts). Available at <http://cfpub2.epa.gov/ncea/cfm/recordisplay.cfm?deid=160548#Download> Accessed January 28, 2014.

EPA. (2012). Water: monitoring and assessment; 5.11 fecal bacteria. Available at <http://water.epa.gov/type/rsl/monitoring/vms511.cfm> Accessed March 30, 2015.

EPA. (2015a). Nutrient policy and data: guidelines and recommendations. Available at < http://www2.epa.gov/nutrient-policy-data/guidelines-and-recommendations#what2> Accessed July 25, 2015.

EPA. (2015b). Health effects support document for the cyanobacterial toxin microcystins. EPA Dcument Number 820R15102. Available at < http://www2.epa.gov/sites/production/files/2015-06/documents/microcystins-support-report-2015.pdf> Accessed July 25, 2015.

EPA. (2015c). Nutrient policy and data: causes and prevention. Available at < http://www2.epa.gov/nutrient-policy-data/causes-and-prevention> Accessed July 25, 2015.

Fundamentals of Environmental Measurements. (2015). Turbidity, total suspended solids & water clarity. Available at <http://www.fondriest.com/environmental-

measurements/parameters/water-quality/turbidity-total-suspended-solids-water-clarity/#> Accessed March 30, 2015.

Harding, W. R., Wright, S. (1999). Initial findings regarding changes in phyto- and zooplankton composition and abundance following the temporary drawdown and refilling of a shallow, hypertrophic South African coastal lake. *Lake and Reservoir Management* 15(1):47-53.

Haugland, R. A., Varma, M., Sivaganesan, M., Kelty, C., Peed, L., Shanks, O.C. (2010). Evaluation of genetic markers from the 16S rRNA gene V2 region for use in quantitative detection of selected Bacteroidales species and human fecal waste by qPCR. *Systematic and Applied Microbiology* 33(6): 348-357.

Hudnell, H. K. (2010). The state of U.S. freshwater harmful algal blooms assessments, policy and legislation. Toxicon. 55: 1024-1034.

Ingrid, C.; Bartram, J. (1999). Toxic cyanobacteria in water: a guide to their public health consequences, monitoring and management; E & FN Spon on behalf of WHO: London, p 416.

Ishii, S., Sadowsky, M. J. (2008) *Escherichia coli* in the environment: implications for water quality and human health. *Microbes and Environments*. 23(2): 101-108.

Kim, H. Y. (2013). Correlation of microcystin with phycocyanin in cyanobacteria. Bowling Green State University. Available at <http://www.bluewatersatellite.com/resources/js/tinymce/jscripts/tiny_mce/plugins/uploaded/Correlation%20of%20Microcystin%20with%20Phycocyanin%20in%20Cyanobacteria%20final.pdf > Accessed March 17, 2014.

Kiviat, E. (2013). Risks to biodiversity from hydraulic fracturing for natural gas in the Marcellus and Utica shales. *Ann. N.Y. Acad. Sci.* 1286: 1–14. DOI: 10.1111/nyas.12146

Kromkamp, J., Van Den Huevel, A., Mur, L.R. (1989). Formation of gas vesicles in phosphorus-limited cultures of Microcystis aeruginosa. *Microbiology* 135(7):1933-1939.

Kurmayer R., Kutzenberger T. (2003) Application of real-time PCR for quantification of microcystin genotypes in a population of the toxic cyanobacterium *Microcystis* sp. *Applied and Environmental Microbiology* 69(11): 6723-6730. DOI:10.1128/AEM.69.11.6723-6730.2003.

LeChevallier, M.W. and W.D. Norton. (1992). Examining relationships between particle counts and *Giardia, Cryptosporidium,* and turbidity. *Journal American Water Works Association.* 84(12): 54-60.

Mancl, K., Veenhuizen, M. A. (2015) Avoiding stream pollution from animal manure. Ohio State University Extension Fact Sheet AEX-708-91. Available at <http://ohioline.osu.edu/aex-fact/0708.html> Accessed April 2, 2015.

Marion, J.W., Lee, J., Wilkins, III, J.R., Lemeshow, S., Lee, C., Waletzko, E.J., Buckley, T.J. (2012). In vivo phycocyanin flourometry as a potential rapid screening tool for predicting elevated microcystin concentrations at eutrophic lakes. *Environmental Science and Technology* 46:4523-4531.

Marquis, P. (2005). Turbidity and suspended sediment as measures of water quality. *Streamline Watershed Management Bulletin* (9) 21–23. Available at <http://www.forrex.org/sites/default/files/publications/articles/streamline_vol9_no1_art4.pdf> Accessed April 3, 2015.

McQuaid, M., Zamyadi, A., Prevost, M., Bird, D. F., Dorner, S. (2010). Use of in vivo phycocyanin fluorescence to monitor potential microcystin-producing cyanobacterial biovolume in a drinking water source. *Journal of Environmental Monitoring,* 13(2): 455-463.

Mieszkin, S., Yala, J. F., Joubrel, R., Gourmelon, M. (2009). Phylogenetic analysis of Bacteroidales 16S rRNA gene sequences from human and animal effluents and assessment of ruminant faecal pollution by real-time PCR. *Journal of Applied Microbiology* 108: 974-984.

NSW Department of Primary Industries. (2009). What causes algal blooms. Available at <http://www.water.nsw.gov.au/Water-Management/Water-quality/Algal-information/What-causes-algal-blooms/What-causes-algal-blooms/default.aspx>

Oberholster, P.J., Botha, A-M., Grobbelaar, J.U. (2004). Microcystis aeruginosa: source of toxic microcystins in drinking water. *African Journal of Biotechnology* 3(3):159-168.

Ohio Agriculture Research and Development Center (OARDC). (2015). Weather system: Eastern station. Available at <http://www.oardc.ohio-state.edu/newweather/stationinfo.asp?id=3> Accessed November 5, 2014.

Ohio Department of Natural Resources (ODNR). (2015). Shale well drilling & permitting. Available at <http://oilandgas.ohiodnr.gov/shale#SHALE> Accessed April 3, 2015

Ohio Environmental Protection Agency (OEPA). (2010). Ohio harmful algal bloom advisories in 2010. Online document. Available at <http://www.epa.state.oh.us/LinkClick.aspx?fileticket=hVW3RnXIGBI%3D&tabid=4802>

Ongley, E. (1996). Water Quality Monitoring - A practical guide to the design and implementation of freshwater quality studies and monitoring programmes: chapter 13 – sediment measurements. *Published by UNEP and WHO.* Available at <http://www.who.int/water_sanitation_health/resourcesquality/wqmchap13.pdf>

Ouellette, A. J. A.; Wilhelm, S. W. (2003). Toxic cyanobacteria: the evolving molecular toolbox. *Frontiers in Ecology and the Environment* 1(7): 359-366.

Pennington, R. (2014). Dealing with amplification inhibitors: reagent choice matters. Promega Corporation, powerpoint. Available at <http://www.promega.com/~/media/files/promega%20worldwide/north%20america/promega%20us/webinars%20and%20events/2014/dealing%20with%20amplification%20inhibitors%20webinar%20feb%202014.pdf?la=en>

Pimentel, J.S.M., Giani, A. (2014). Microcystin production and regulation under nutrient stress conditions in toxic *Microcystis* strains. *Applied and Environmental Microbiology* 80(18):5836-5843.

Rulli, S. (2012). Introduction to real-time qualitative PCR (qPCR). SabioSciences, a QIAGEN company. Available at <http://www.sabiosciences.com/manuals/seminars/qPCR%20Introduction_April%202012.pdf >

Schatz, D., Keren, Y., Vardi, A., Sukenik, A., Carmeli, S., Börner, T., Dittmann, E., Kaplan, A. (2007). Towards clarification of the biological role of microcystins, a family of cyanobacterial toxins. *Environmental Microbiology* 9(4):965-970.

Schmidlim, T. W. (1996) "Climate and weather". *A geography of Ohio*. Peacefull, L. Kent: Kent State University Press, 1996. 16-30. Print.

Schubel, J. R. (1977). Sediment and the quality of the estuarine environment: Some observations. Fate of pollutants in the air and water environments. Part 1. Mechanism of interaction between environments and the mathematical modeling and the physical fate of pollutants. *John Wiley and Sons*, New York, NY. I. H. Suffet (ed.), 399-424

Scott, T.M., Rose, J.B., Jenkins, T.M., Farrah, S.R., Lukasik, J. (2002). Microbial source tracking: current methodology and future directions. *Applied Environmental Microbiology* 69(12): 5796-5803.

Sevilla, E., Martin-Luna, B., Vela, L., Bes, M. T., Fillat, M. F., Peleato, M. L. (2008). Iron availability affects mcyD expression and microcystin-LR synthesis in *Microcystis aeruginosa*. *Environmental Microbiology* 10(10): 2476-2483.

Sivonen, K., Carmichael, W.W., Namikoshi, M., Rinehart, K.L., Dahlem, A.M., Niemelä, S.I. (1990). Isolation and characterization of hepatotoxic microcystin homologs from the filamentous freshwater cyanobacterium Nostoc sp. strain 152. *Applied and Environmental Microbiology* 56(9): 2650-2657.

Tillett, D., Neilan, B. A. (2000). Xanthogenate nucleic acid isolation from cultured and environmental cyanobacteria. *Journal of Phycology* 36(1): 251-258. DOI: 10.1046/j.1529-8817.2000.99079.x

Toman, E.M. and Skaugset, A.E. (2011). Reducing sediment production from forest roads during wet-weather hauling. *Transportation Research Record: Journal of the Transportation Research Board,* 2203(1): 13-19.

Turner Designs. (1999). Application note: cyanobacteria detection in water using *in vivo* fluorometry. Revision A. S-0061. p 1-10. Available at <http://www.turnerdesigns.com/t2/doc/appnotes/S-0061.pdf> Accessed 18 March 2014.

USGS. (2012). *Escherichia coli* bacteria density in relation to turbidity, streamflow characteristics, and season in the Chattahoochee river near Atlanta, Georgia, October 2000 through September 2008—description, statistical analysis, and predictive modeling. Available at <http://pubs.usgs.gov/sir/2012/5037/pdf/sir2012-5037.pdf> Accessed 15 April 2015.

United States Geological Survey (USGS). (2013). Oregon water science center, turbidity, units of measurement. Available at <http://or.water.usgs.gov/grapher/fnu.html> Accessed March 5, 2014

United States Geological Survey (USGS). (2014). StreamStats Interactive Map. Available at <http://water.usgs.gov/osw/streamstats/ohio.html> Accessed February 2014

uticashaleblog.com. (2015). Utica shale maps, data, resources, news, analysis, and commentary. Available at <http://www.uticashaleblog.com/>

Vahtera, E., Conley, D.J., Gustafsson, B.G., Kuosa, H., Pitkänen, H., Savchuk, O.P., Tamminen, T., Viitasalo, M., Voss, M., Wasmund, N., and Wulff, F. (2007). Internal Ecosystem Feedbacks Enhance Nitrogen-fixing Cyanobacteria Blooms and Complicate Management in the Baltic Sea. *AMBIO: A Journal of the Human Environment* 36(2): 186-194.

Van der Westhuizen, A.J., Eloff, J.N. (1985). Effects of temperature and light on toxicity and growth of the blue-green alga Microcystis aeruginosa. *Planta* 163:55-59.

Vidic, R., Brantley, S.L., Vandenbossche, J.M., Yoxtheimer, D., and Abad, J.D. (2013). Impact of Shale Gas Development on Regional Water Quality. *Science* 340, 1235009. DOI: 10.1126/science.1235009

Vinyard, G. L., O'Brien W. J. (1976). Effects of light and turbidity on reactive distance of bluegill (*Lepomis macrochirus*). *Journal of the Fisheries Research Board of Canada* (33) 2845-2849.

USGS. (2008). Scientific Investigations Report 2008–5201: nutrient and chlorophyll a water samples. Available at < http://pubs.usgs.gov/sir/2008/5201/section5.html> Accessed July 16, 2015.

World Health Organization (WHO). (2003). Guidelines for safe recreational water environments; coastal and fresh waters; Geneva; Vol. 1, p 219.

World Health Organization (WHO). (2004). Water treatment and pathogen control: process efficiency in achieving safe drinking-water. chaper 2: removal processes. Available at < http://www.who.int/water_sanitation_health/dwq/en/watreatpath2.pdf?ua=1>

Zhu, L., Wu, Y., Song, L., Gan, N. (2014). Ecological dynamics of toxic Microcystis spp. and microcystin-degrading bacteria in Dianchi Lake, China. *Applied and Environmental Microbiology* 80(6):1874-1881.

Characterizing stream restoration’s water quality improvement potential through hyporheic exchange enhancement

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6. Baker, S.B., & Jefferson, A. Development of hyporheic exchange and nutrient uptakefollowing stream restoration. Consortium of Universities for the Advancement of Hydrologic Sciences International Biennial Meeting, Shepherdstown, WV. August, 2014. (poster)
7. Baker, S.B., & Jefferson, A. Development of hyporheic exchange and subsurfaceprocesses following stream restoration. Geological Society of America Annual Conference, Vancouver, B.C. October, 2014. (poster)
8. Baker, S.B., & Jefferson, A. Development of hyporheic exchange and subsurface processes following stream restoration. Kent State Water Research Symposium, Kent, OH. October, 2014. (poster)
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Characterizing stream restoration's water quality improvement potential through hyporheic exchange enhancement

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Problem and Research Objectives

Restoration of streams degraded by urban or agricultural runoff is a multi-million dollar industry in the state of Ohio (Mecklenberg and Fay, 2011). Restoration ultimately seeks to return streams to their pre-disturbance physical and biological conditions, though pragmatic goals include improving geomorphic stability, habitat diversity, stormwater management, and water quality. Despite these goals, restoration often falls short of significant biological improvements, as quantified by post-restoration fish and macroivertebrate monitoring. However, these biotic metrics may be limited by other watershed factors or by lack of connected habitat from which recolonization could occur (Spanhoff et al., 2007; Bond and Lake, 2003). Instead, direct measurements of physiochemical processes may be more useful indicators of a stream restoration project's long-term potential to improve water quality and facilitate ecosystem services. The way the restored reach moves and stores water and changes the water chemistry is what sets the template for the biotic communities to return and populate the stream successfully in the long term. Unfortunately, the focus on macrobiology as a monitoring tool means that data and scientific understanding are limited with respect to how stream restoration alters underlying physiochemical and ecosystem processes.

The research undertaken focuses on a specific question: ***How does stream restoration affect hyporheic flow over time and does this improve water quality?*** This question addresses multiple levels of stream function (Harman *et al.*, 2012), but focuses on one of the most important processes affecting stream water chemistry: *hyporheic exchange*. Hyporheic exchange moves water into, through, and out of the streambed sediment matrix, in the zone of stream water and groundwater mixing known as the hyporheic zone (Bencala, 2006). Hyporheic flowpaths allow for stream water to participate with the streambed substrate in biogeochemical cycling of nutrients and pollutants, buffering stream water temperature, and supporting important benthic microhabitats for invertebrates and fish. Thus, reestablishment of hyporheic exchange is critical to overall restoration success (Lawrence et al., 2013).

Hyporheic exchange is controlled by pressure gradients generated by streamflow over and around geomorphic structures and by the hydraulic conductivity of the streambed sediment (Buffington, 2009). Thus, restoration alters both of the controls on hyporheic exchange, by reshaping the channel bed and by changing the size or compaction of stream sediments (Boulton, 2007). Further, post-restoration flushing of fine sediments into or out of the streamed and geomorphic structures may alter hydraulic conductivity and therefore hyporheic exchange (Figure 1, Nowinski, 2011), even in the absence of visible changes in stream morphology. Yet, there is limited research on how restoration affects hyporheic exchange, and there is no research on how hydraulic conductiviy and

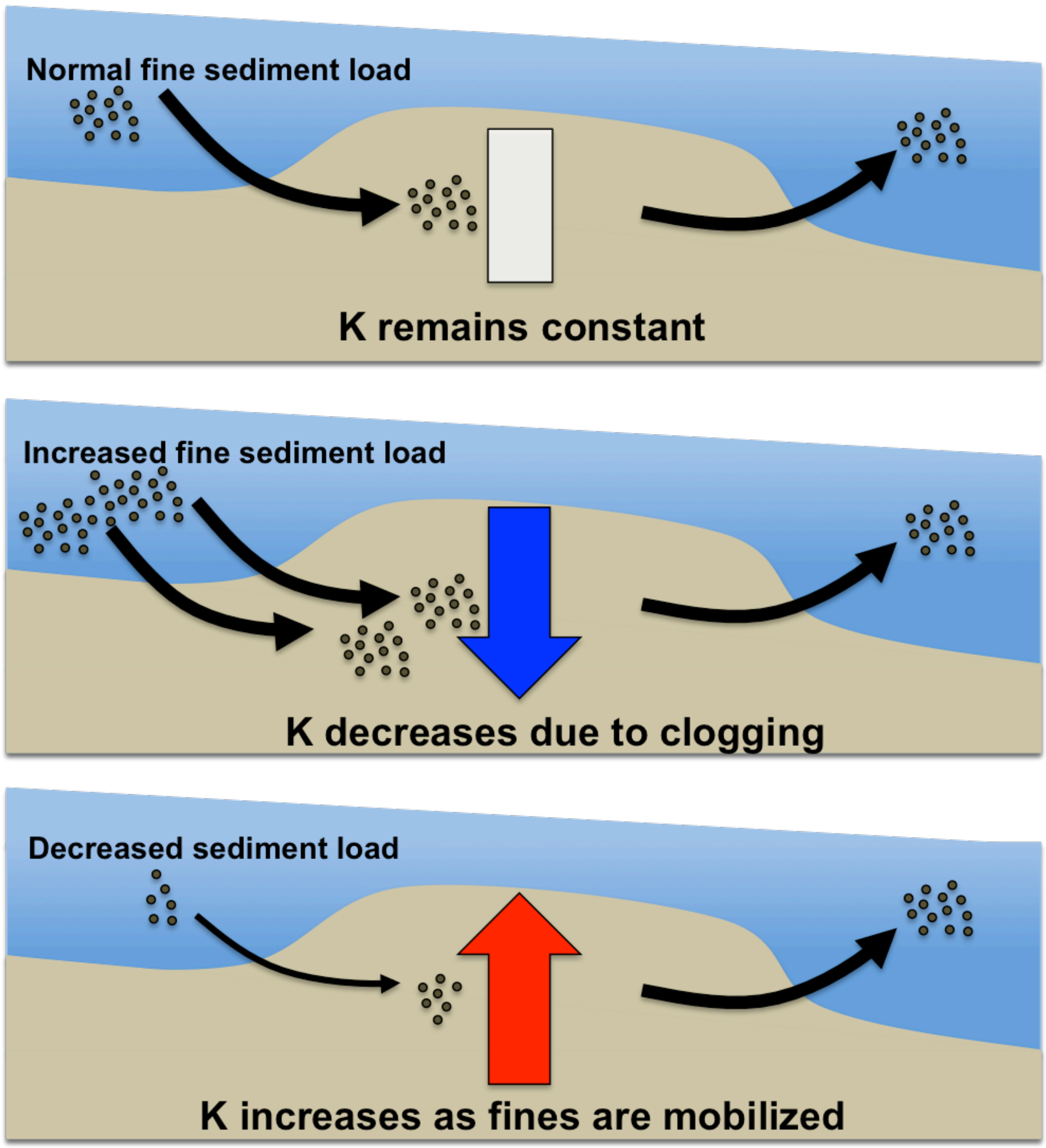


Figure 1. The fine sediment load in a stream can affect streambed hydraulic conductivity. If fine sediment is in balance with the energy to mobilize it within and from the streambed, no change is likely. If the load is increased, flowpaths may become blocked and reduce permeability. If the load is less than what the stream can move than the flow of water may winnow out fine material from the bed and increase hydraulic conductivity.

hyporheic exchange evolve following stream restoration. However, research on these topics has significant potential to inform and improve stream restoration practices, since restoration designers and construction crews can control or influence many of the factors that set the template for hyporheic exchange.

If stream restoration is to be successful it is important that hyporheic flowpaths are generated by the constructed channel form and sustained over the project lifetime.

Despite the significance of hyporheic exchange and the associated biogeochemical cycling, these processes are not specifically designed for nor monitored following completion of a project. Without an understanding of the how these processes function in a restored stream, restorations will continue to lack ability to fully develop to an ecologically beneficial state.

The objectives of the project are to:

(1) evaluate changes in hyporheic exchange over time following stream restoration; and
(2) assess the effects of physical on nutrient uptake and water quality in restored stream reaches

Methodology

Study Sites

Kelsey Creek at Kennedy Park in Cuyahoga Falls, OH has an 8.36 km^2 suburban watershed with 18% forested area (Figure 1). The stream was restored in August 2013 to address head cutting and bank erosion propagated upstream after a low head dam was removed adjacent to Munroe Falls Ave. Biohabitats, Inc. conducted the restoration, building floodplains and long pools and riffles, intended to mitigate stormwater flows. The construction used two layers of large cobbles in the riffles to ensure geomorphic stability, while the rest of the material used was sourced from the banks. Willow stakes and a sterile cover of rye grass was planted after construction was completed in 2013. Native plants, shrubs, trees, and seedmix were all planted during the fall of 2014.

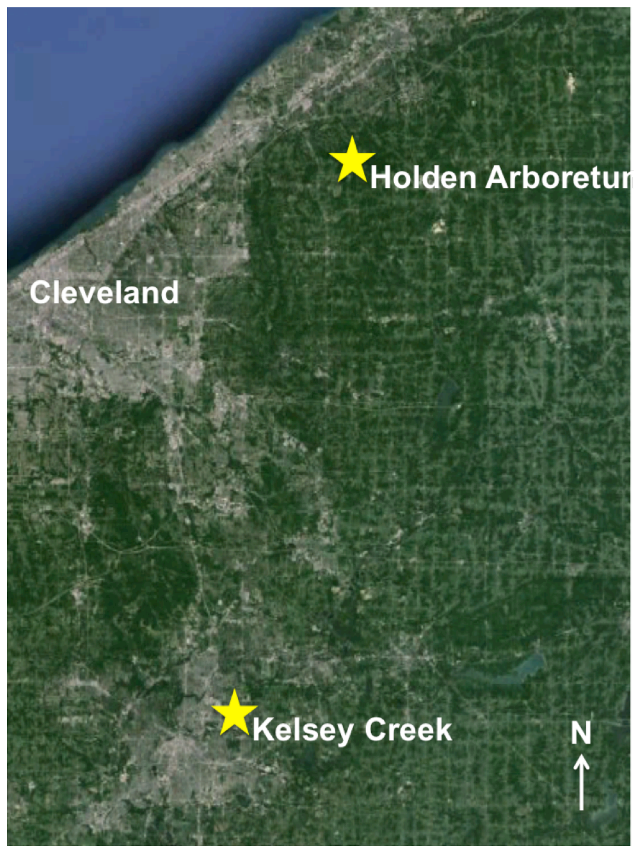


Figure 2. Map of field sites in northeast Ohio.

An unnamed tributary to Pierson Creek at **Holden Arboretum** near Kirtland, OH was restored in two sections (October 2013 and April 2014) at the outflow of Heath Pond to accommodate a new outlet (Figure 1). The restoration implemented boulders for geomorphic stability and externally sourced bank run material for the streambed. The stream is ephemeral, flowing when the pond overtops a constructed lip. The drainage is 0.39 km^2 and is 58% forested. Freeze coring of the sediment found that the streambed is predominantly clay and silt, with some gravel near the surface.

Hydraulic head and hydraulic conductivity

Piezometers were installed at Holden Arboretum in three stream reaches, one restored in April 2014, one restored in October 2013, and one which was unrestored. At each reach, 12 piezometers were arrayed in a longitudinal transect over a riffle, with the upstream and downstream most positions in a pool, and some lateral positions within the riffle. At Kelsey Creek, piezometers were removed after each set of measurements was completed, and then reinstalled in the same position every six months to make ensuing measurements.

Piezometers were constructed from 1 meter long 1¼" inner diameter schedule 40 PVC pipe with a rounded end cap, screened with 80+ 1.5mm holes over the bottom 15 cm including through the sides of the end cap. Piezometers were then installed in longitudinal arrays through riffle structures, driven to a depth of 30 cm with a small sledge hammer. This placed the screened interval at 15-30 cm below the top of the streambed. A peristaltic pump was used to develop piezometers and evacuate any infilling sediment from the pipe. Hydraulic head was measured with an electric tape to find the depth from the top of the pipe to water inside the piezometer and to the stream surface outside.

Hydraulic conductivity was determined using the Hvorslev slug test method. HOBO Water Level Loggers (Onset Computer Corporation, Bourne, MA) were placed in the bottom of a piezometer to measure pressure at the bottom of the pipe and water was rapidly poured into the top of the piezometer to raise the water level suddenly and then allow for recovery. Atmospheric pressure was measured with an additional HOBO hung in the air, and used in the HOBOWare Barometric Pressure Assistant function to correct pressure during slug tests and determine depth of water over time.

Water Chemistry

Water samples were collected from the upstream end of each study area, transitions between restored and unrestored reaches, and at the downstream end of each study area. Samples were collected using 50 mL syringes which were first rinsed three times with stream water. The collected volume was filtered through a glass fiber filter and into a 50 mL Falcon tube. In situ measurements of pH, conductivity, water temperature, and dissolved oxygen were made concurrent with water grab sampling, using a YSI Professional Plus multiparamter sonde (Yellow Springs, OH). Samples were also collected from the hyporheic zone by drawing water from piezometers. For these samples, rinses of the syringe with sample water was conducted when available volumes allowed. All samples were chilled on ice in coolers during collection in the field and then frozen for storage.

For laboratory analyses, water samples were thawed and immediately prepared for analyses. One milliliter from each sample was diluted 1:10 with 2% nitric acid and analyzed on a Perkin-Elmer ICP-OES 8000 for calcium, potassium, magnesium, manganese, sodium, nickel, and zinc. Samples were also analyzed on a Dionex ICS-2100 chromatography system for chloride, fluoride, nitrate, nitrite, sulfate, bromide, and phosphate, at both full strength and at 1:10 dilution with deionized water in order to bring some constituents into a readable range.

Freeze Coring and Sediment Cation Extraction

Streambed sediments were sampled using freeze coring methods. A ¾" galvanized steel pipe with an end cap was driven 30 cm vertically into the streambed, 5-15 cm from each piezometer location, at a position as close to directly downstream as possible. In some instances the presence of cobbles required that the core be taken to the side of the piezometer. Once the pipe was in position, dry ice pellets were dropped into the pipe and isopropyl alcohol was added. After 5-20 minutes the saturated stream sediments around the pipe had frozen and the core was lifted out. A hammer and chisel were used to break the sediment and ice away from the pipe and subdivide it into the upper portion from 0-15 cm depth, and the lower portion from 15-30 cm depth. The frozen sediment was stored in plastic Ziploc bags, allowed to melt, and dried in an oven at 105˚C. Aggregates were broken apart gently using a mortar and pestle to separate grains without altering them. Sediment was then analyzed for grain size using a Retch Camsizer P4 Video Particle Size Analyzer (Haan, Germany) with a detection range of 30-30,000 microns in 50 bins distributed logarithmically. Binned data was interpolated linearly to derive percentiles of the sediment size distribution for each sample.

Sediment from freeze cores was ground in a SPEX Sample Prep 8000M Mixer/Mill (Metuchen, NJ) for 5 minutes to create a powder. Approximately 2.5 g of milled soil was weighed and put in preweighed 50 mL centrifuge tubes. 25 mL of 0.1 M $BaCl_2$-$NaCl_2$ was added and the tubes were placed on a rotator for 30 min at 180 rpm. Samples were then centrifuged for 30 min at 4000x g and the supernatant was collected in a new tube, diluted at 1:10 with 10% HNO_3 and analyzed on an ICP-OES for Ca, Na, Mg, K, Fe, and Mn.

Principal Findings

Hyporheic exchange

During the study period (June-November 2014) at Holden Arboretum, the overall variability in hydraulic conductivity (K) across each reach did not change significantly (Figure 3), but K at most individual piezometer positions did change over time (Figure 4). Generally, piezometers in pools had a decrease in K, while those in riffles had an increase in K. This likely due to the steeper channel slope in the riffles allowing sediment mobilization, which is then deposited in the more gently sloping pool areas. These two patterns approximately balance each other out within a given reach, as sediment mobilized from one area, increasing K, is then deposited shortly down gradient, decreasing K. The lack of change in the overall variability substantiates this at Holden Arboretum, showing no reach-wide shift over time (Figure 3), regardless of restoration age. Hydraulic head measurements showed very small gradients suggesting localized upwelling and downwelling was not strong. In this stream, hyporheic exchange may be limited by lack of well-defined geomorphic structures that promote downwelling and upwelling rather than by clogging of bed sediments.

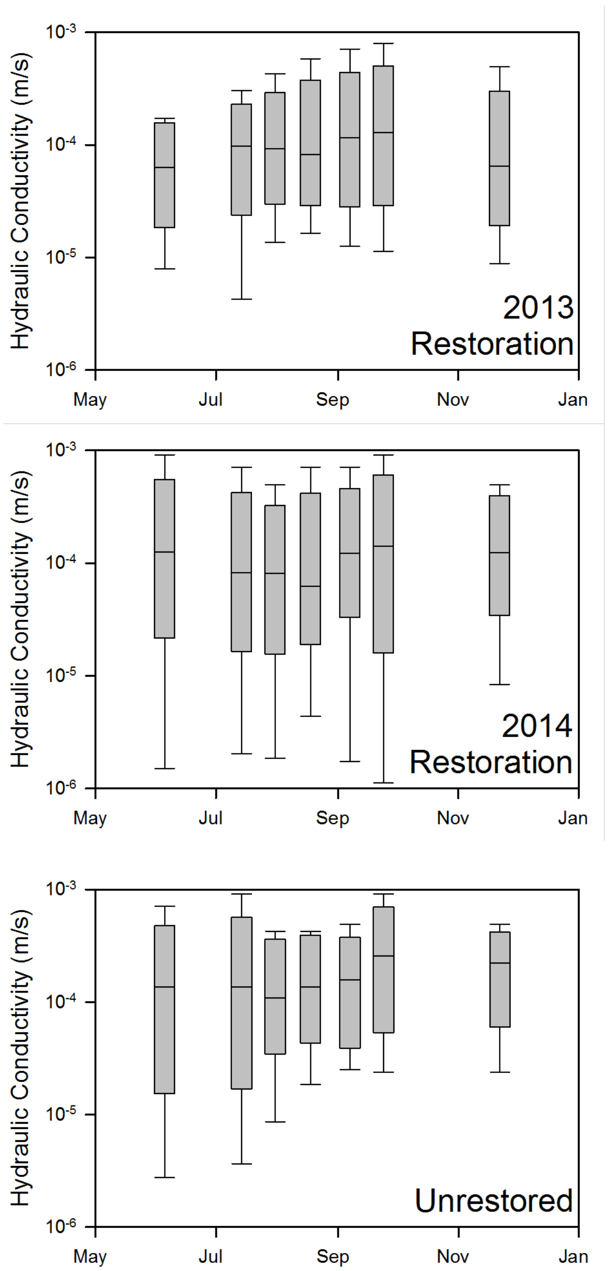
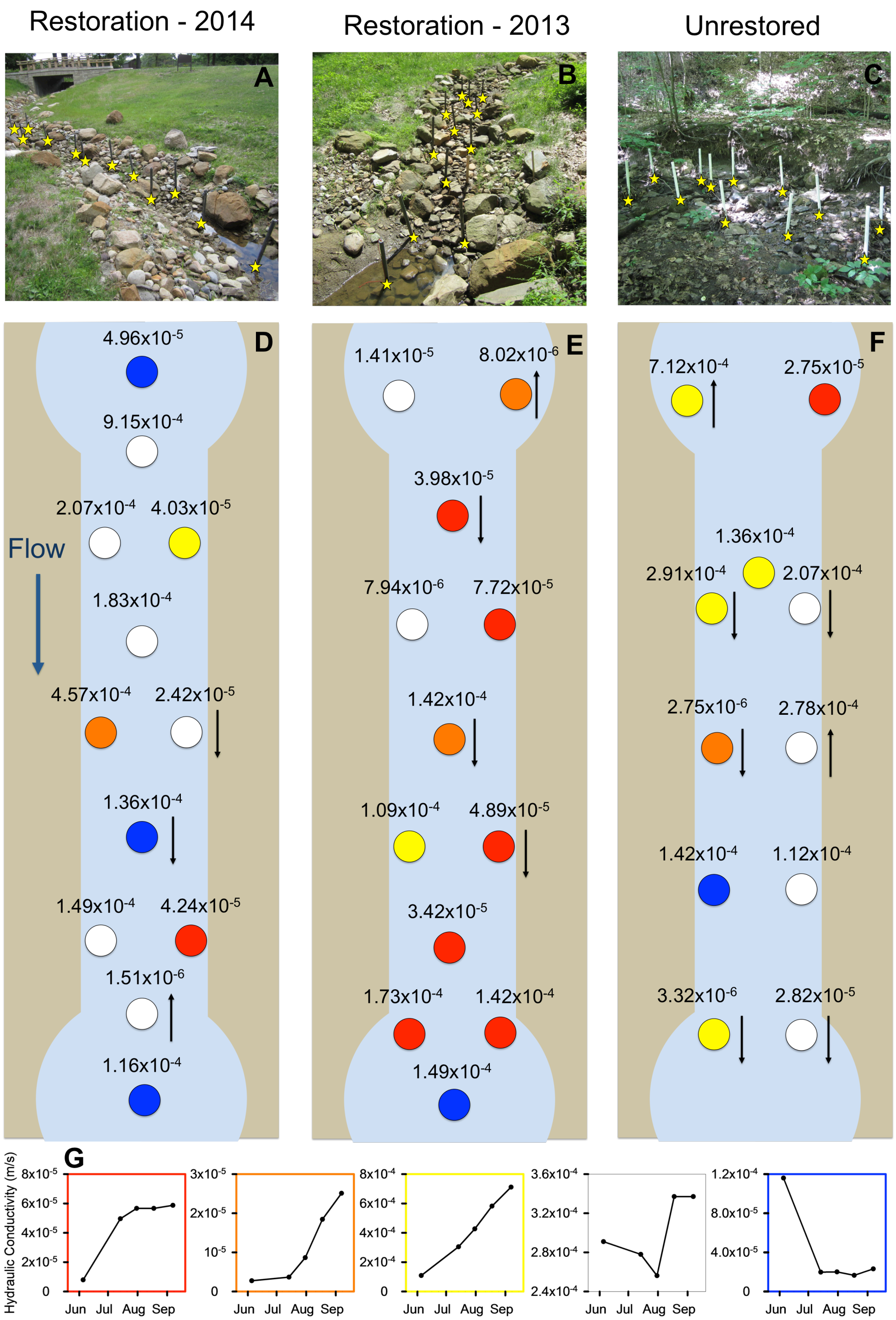


Figure 3. Distribution of hydraulic conductivities for all measured points at each stream reach through time.



At three reaches (A, B, C) at Holden Arboretum, 12 piezometers were installed (screened at a depth of 15-30 cm) to monitor hydraulic head and measure hydraulic conductivity with slug tests. Representative patterns in hydraulic conductivity change from June 5th 2014 to November 22nd 2014 (G) are indicated by colored circles at each piezometer position (D, E, F). Initial hydraulic conductivity is noted next to each (m/s) and arrows indicate upwelling or downwelling where hydraulic head was not neutral.

↑ Positive hydraulic head

↓ Negative hydraulic head

☆ Piezometer position

Figure 4. Change in hydraulic conductivity over time for each piezometer at Holden Arboretum.

Hydraulic conductivity measurements at Kelsey Creek reveal a different trend. The mean K for each set of measurements decreases over time as does the lowest quartile, particularly for the November 2014 data (Figure 5). Hydraulic gradients at Kelsey Creek were also typically not strong. The overall trend toward a less permeable streambed at this site suggests that hyporheic flowpaths are being blocked and restricted by infilling of the sediment matrix with fine material. This may be due to sediment input from the stream reach directly upstream of the restored reach, which suffers from severe bank erosion as well. It is therefore likely that the hydraulic conductivity in the riffle structures of the restored reach will continue to decrease until the bank erosion upstream is stabilized. This may require additional restoration.

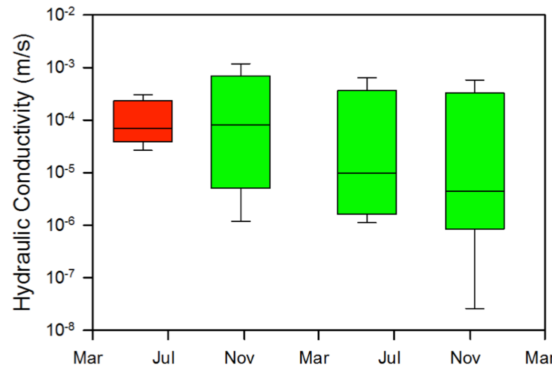


Figure 5. Hydraulic conductivity distribution at Kelsey Creek before (red) and after (green) the August 2013 restoration.

Water Quality

No significant changes to water quality were observed across the restored reaches. Water grab samples were analyzed for anions including nitrate, nitrite, chloride, phosphate, fluoride, bromide, and sulfate, and cations including calcium, magnesium, manganese, iron, nickel, and sodium. The chemistry from the top of the restored reaches at both Holden Arboretum and Kelsey Creek did not vary significantly from that at the downstream end of the reaches and thus the restorations are judged to have no major impact on the chemical quality of the water in the channel. The only notable change to water quality was a decrease in temperature across each reach at Holden Arboretum. This is largely due to elevated water temperature in the pond, which then cools off as it flows through the restored reaches. Such decreases in water temperature could result from hyporheic exchange, or from increased shading of the stream reaches relative to the pond.

Pore water collected from piezometers at Kelsey Creek revealed one important trend. Manganese concentrations were greatest in the head or upstream end of riffle 2 and 3, and in the middle of riffle 1, in each case as much as ten times higher than the surface water (Figure 6). Over the length of the each riffle, these high manganese levels dropped off, returning to levels similar to the surface water by the end of each riffle. Concentrations were also high at positions near the side of the channel. These results suggest that redox chemistry is active within the constructed riffles in the restoration and is likely caused by dissolved oxygen gradients along flowpaths through these structures. Nutrient levels did not change significantly within riffle sediments though this may be due to low initial concentrations. These restoration structures may instead have more impact on dissolved metal loads in the stream, serving as a source for redox species like manganese. This may change over time as manganese is depleted from the hyporheic zone and other biogeochemical processes become more important.

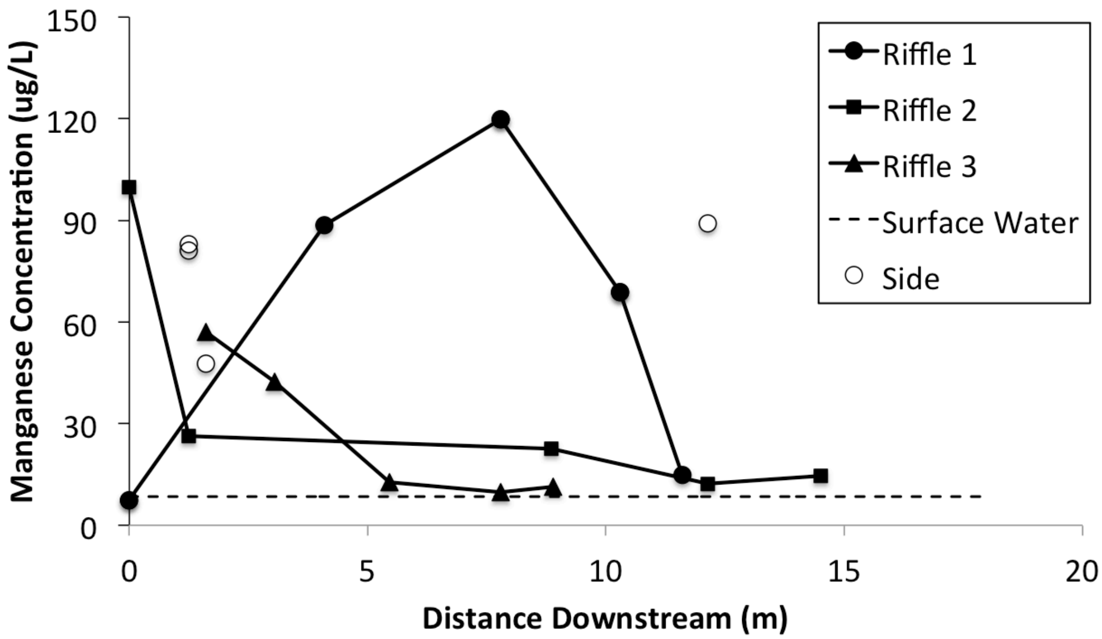


Figure 6. Manganese concentrations from pore water 15-30 cm below the streambed surface at Kelsey Creek.

<u>On-going analyses</u>

Freeze coring demonstrated a range of sediment sizes and sorting as well as colors indicative of redoximorphic features (Figure 7). Grain size analysis, sediment cation extractions and nutrient uptake measurements have been completed and analysis of these data will inform the hydraulic conductivity and porewater chemistry interpretations. Beyond the original scope of the grant, salt and dye tracer tests were conducted in the study reaches and 2-D model simulations will be completed later in 2015. These data and simulations will help generalize the findings.

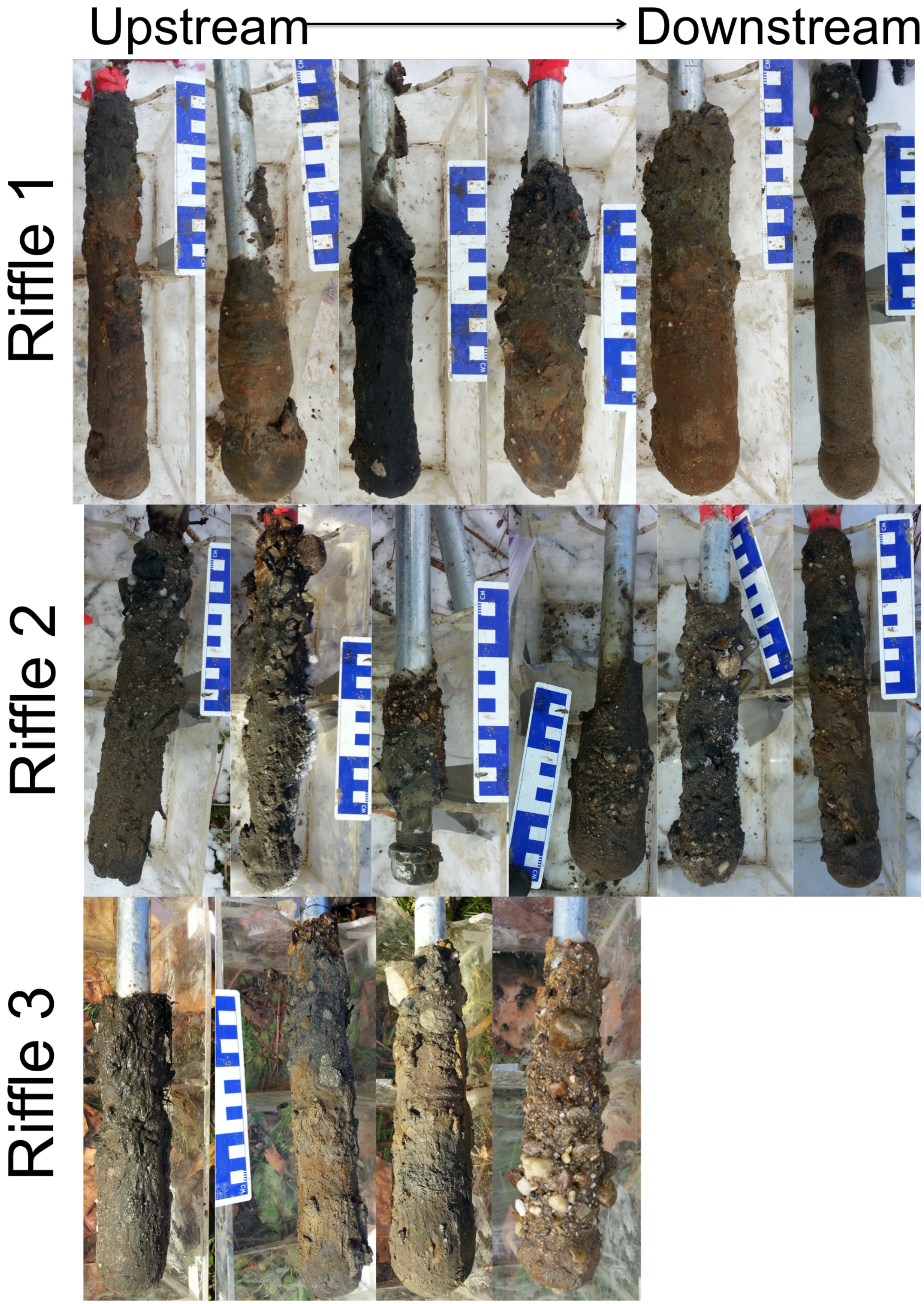


Figure 7. Freeze cores from riffles at Kelsey Creek. Sediment was sampled to a depth of 30 cm adjacent to each piezometer.

Significance

Overall this study has discovered a dynamic environment in the hyporheic zone of restored streams, with changing hydraulic conductivity and strong chemical gradients. In the stream where sediment inputs were restricted by the upstream dam (Holden Arboretum), hydraulic conductivity did not change at the reach scale over a 5 month period. Changes at individual points in the streambed, however, were substantial both in restored and unrestored reaches. In the stream where sediment inputs were unrestricted, and likely substantial, hydraulic conductivity at the reach scale declined over 15 months following restoration. Despite these dynamics, neither restored reach effected a change in surface water chemistry, as measured by baseflow grab samples analyzed for nitrate and other anions.

Hyporheic exchange was not significant enough to modify the water quality signal resulting from upstream land use and geology. This could be either be the result of insufficient hydraulic conductivity; the observed weak upwelling and downwelling; or short restored reach length. The observed weak upwelling and downwelling at both sites is notable because it belies claims that the geomorphic structures built during stream restoration are sufficient to generate substantial hyporheic exchange. Hydraulic conductivity was relatively high at Holden Arboretum and initially post-restoration at Kelsey Creek, so limiting hydraulic conductivity is unlikely for those sites and timepoints. However, the lowered hydraulic conductivity at Kelsey Creek by November 2014 may further impede already limited hyporheic exchange at this site. Finally, the short reach lengths in this study, which are typical of stream restorations, limit the residence time of water in the reach and the probability that water will spend sufficient time in the hyporheic zone to undergo biogeochemical processes such as denitrification.

While the study was limited to two sites and approximately one year of data, the results suggest that stream restoration practices may not induce sufficient hyporheic exchange to improve downstream water quality. Further, for reaches with fine sediment inputs, hyporheic exchange may become limited over time post-restoration.

References

Bencala, K. E. (2006). Hyporheic exchange flows. *Encyclopedia of hydrological sciences*, John Wiley & Sons, Ltd.

Bond, N. R., & Lake, P. S. (2003). Local habitat restoration in streams: Constraints on the effectiveness of restoration for stream biota. *Ecological Management & Restoration, 4*(3), 193-198.

Boulton, A. J. (2007). Hyporheic rehabilitation in rivers: Restoring vertical connectivity. *Freshwater Biology, 52*(4), 632-650.

Buffington, J. M., & Tonina, D. (2009). Hyporheic exchange in mountain rivers II: Effects of channel morphology on mechanics, scales, and rates of exchange. *Geography Compass, 3*(3), 1038-1062.

Harman, W., R. Starr, M. Carter, K. Tweedy, M. Clemmons, K. Suggs, C. Miller. 2012. *A Function-Based Framework for Stream Assessment and Restoration Projects.* US Environmen- tal Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, DC EPA 843-K-12-006.

Lawrence, J. E., Skold, M. E., Hussain, F. A., Silverman, D. R., Resh, V. H., Sedlak, D. L., et al. (2013). Hyporheic zone in urban streams: A review and opportunities for enhancing water quality and improving aquatic habitat by active management. *Environmental Engineering Science, 30*(8), 480.

Mecklenberg, D. A., & Fay, L. A. (2011). *A functional assessment of stream restoration in Ohio* (Technical Report, Ohio Department of Natural Resources.

Nowinski, J. D., Cardenas, M. B., & Lightbody, A. F. (2011). Evolution of hydraulic conductivity in the floodplain of a meandering river due to hyporheic transport of fine materials. *Geophysical Research Letters,38*(1), - L01401.

Linked geomorphic and ecological responses to river restoration: Influence of dam removal on river channel structure and fish assemblages

Basic Information

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End Date:	2/29/2016
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Descriptors:	None
Principal Investigators:	Kristin L Jaeger, Mazeika Sullivan

Publications

1. Dorobek, A.C. and S.M.P. Sullivan. Fish community dynamics in the short-term following lowhead dam removal in an urban river system. Oral. Ohio Biodiversity Conservation Partnership Annual Meeting, Feb. 2015, OSU 4-H Center, Columbus, OH. Oral Presentation
2. Vent, D. and S.M.P. Sullivan. Restoration of sensitive and rare fish in altered landscapes. Poster. Ohio Biodiversity Conservation Partnership Annual Meeting, Feb. 2015, OSU 4-H Center, Columbus, OH.
3. Dorobek, A.C., and S.M.P. Sullivan. Lowhead dam removal in an urban river system leads to shifts in fish assemblages in the short term. Oral. American Fisheries Society 145th Annual Meeting, August 2015, Portland, OR.
4. Dorobek, A.C. and S.M.P. Sullivan. Shifts in Fish Food Webs Following Lowhead Dam Removal in an Urban River System. Poster. Ohio Biodiversity Conservation Partnership (OBCP) – Terrestrial Wildlife Ecology Laboratory Meeting (TWEL), Nov. 2015, OSU 4-H Center, Columbus, OH.
5. Dorobek, A., Sullivan, S.M.P., and A. Kautza. 2015. Short-term consequences of lowhead dam removal for fish assemblages in an urban river system. River Systems 21: 125-139.

Linked geomorphic and ecological responses to river restoration: Influence of dam removal on river channel structure and fish assemblages

Final report

PI: Dr. Kristin Jaeger, School of Environment and Natural Resources, The Ohio State University, Wooster, OH. Current address: USGS Washington Water Science Center, Tacoma, WA.

Co-PI: Dr. Mazeika Sullivan, School of Environment and Natural Resources, The Ohio State University, Columbus, OH. Note that the PI role was assumed by Dr. Sullivan in August 2015 when Dr. Jaeger left OSU.

Problem and Research Objectives

Over half of the large rivers in the world are affected by dams (Nilsson et al. 2005). As of 1999, 75,000 dams existed in the continental United States (Graf 1999), and Ohio contains thousands small dams <4 m in height. As these small and lowhead dams (< 7.5 m) age or their upstream reservoirs fill with sediment thus limiting their ability to store water, their removal is becoming an increasingly popular restoration method to reestablish connectivity of upstream and downstream streamflow, sediment regimes, and movement of organisms (Poff and Hart 2002). Yet despite the increasing trend towards dam removal there is an alarming lack of data relative to the ecological impacts of small dams and dam removal (Hart et al. 2002, Stanley and Doyle 2002, 2003). Dam removal results in upstream and downstream changes to both the channel morphology (the physical shape of channel) and the streamflow velocity regime, with subsequent consequences for the aquatic ecosystem including fish assemblages. However, the particular character of geomorphic change associated with dam removal and its subsequent influence on ecosystem processes remain poorly resolved. Thus, there is an urgent need to evaluate ecosystem consequences of dam removal (Gangloff 2013). In particular, fish communities represent an important component of aquatic ecosystems and play important social and economic roles in Ohio. Recreational fishing is a major revenue generator within the state. Therefore, how fish assemblages respond to dam removal reflects a critical knowledge gap in the burgeoning dam removal and river restoration research.

The removal of the 77 year-old “5th Avenue” lowhead dam (145-m long, 2.5-m high) on the Olentangy River presents an opportunity to investigate linked ecological-geomorphic consequences of dam removal as they relate to fish assemblage structure. In addition, these dams are located on rivers flowing through urban areas of Columbus, Ohio, thus providing additional opportunities to evaluate response within a land use that has not previously been evaluated in the context of dam removal.

The removal of the dam is part of the Lower Olentangy River Restoration project to restore the river channel to free-flowing conditions found upstream and downstream of the dam, reestablish floodplain features and vegetation, and develop pockets of wetlands. Because the restoration project is designed to accelerate long-term river change associated with dam removal (Figure 1), it creates an extraordinary scientific opportunity to monitor and investigate changes in linked ecological-geomorphic processes, particularly within the context of evaluating changes in the fish assemblage over time as well as overall evaluation of the function of a re-created river system.

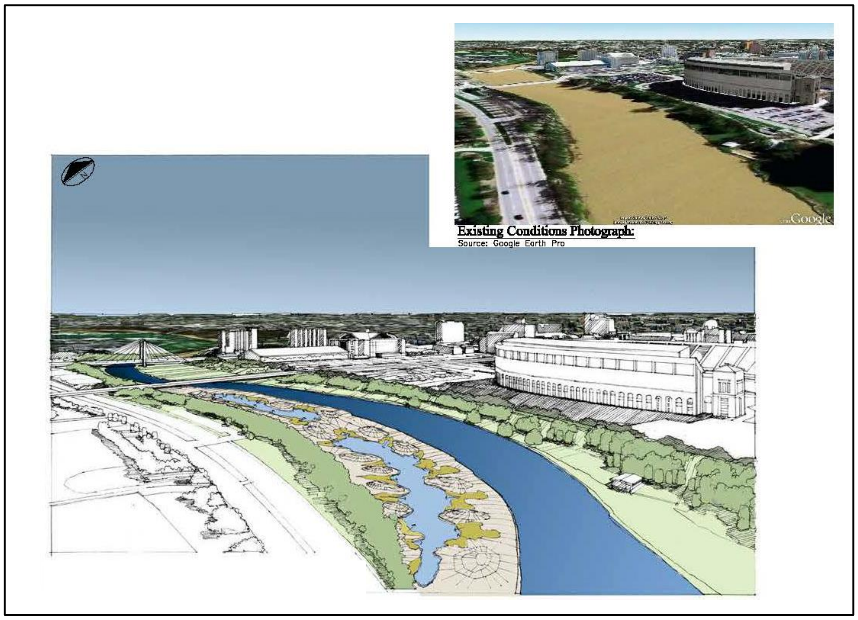

Figure 1. Proposed river corridor upstream of dam ~2 years after dam removal. Downstream of dam will not be actively restored. Dam was removed in August-September 2012.

<u>Research Objectives</u>

The overarching research objective was to investigate linked hydrogeomorphic and ecological short term response in a river following dam removal with a focus on fish community assemblages. We quantified geomorphic and ecological response following dam removal in (1) a downstream reach, (2) two upstream reaches, and tested that (3) riffle development following dam removal would be an important factor driving benthic macroinvertebrate and fish community responses. In addition, we also evaluated differences between restored and unrestored upstream reaches.

Methods

Study Design: The major component of the study followed a modified BACI (before-after, control-impact design) (Stewart-Oaten et al. 1986, Downes et al. 2002). Through ongoing research in the Olentangy River system (2010-present), Dr. Sullivan has "before" data relative to fish community assemblages from river reaches upstream and downstream of the 5th Avenue Dam, as well as at "control" sites. Pre-dam removal geomorphic data (Stantec, Columbus, OH) were available for use in this project. The control site represents an upstream (impounded) reach (OR1) of an intact lowhead dam of comparable size and age in the same study river system. Therefore, the proposed work focused on the "after" component of the design, and include data collection at study reaches upstream (two, one in unrestored [OR2] and one in restored [OR3] section) and downstream (one [OR4]) of the former 5th Avenue Dam, and one upstream of the intact "control" dam (OR1) (n = 4, Figure 2). Study reaches are ~300m long. All data were collected between 2013-2015.

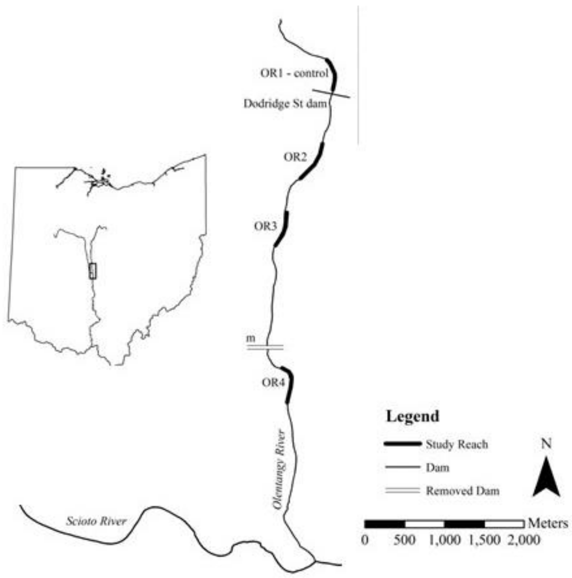

Figure 2. The Olentangy River study area in Columbus, Ohio. Picture for OR1-3 is facing upstream; picture for OR4 is in downstream direction. Figure adapted from (Dorobek et al. (2015).

For the second component of the study (3rd research objective), we surveyed five riffles at or upstream of the previous dam location, one riffle below the previous dam location, and one riffle downstream of an existing lowhead dam in the same river. Within each riffle, three quadrats were established at the top, middle, and bottom portions of the riffle to characterize representative microhabitats based on flow and substrate characteristics. Fish, benthic macroinvertebrate, chemical water-quality, substrate, and flow surveys were conducted within each quadrat at six time intervals: late spring (June), summer (August), and late fall of 2014; early spring (March), late spring (June), and summer (August) of 2015.

Geomorphic Data Collection: Changes in geomorphic complexity were quantitatively evaluated using repeated fine-resolution (0.5-m spacing array) bathymetric surveys in a grid array within each of the study reaches using an Acoustic Doppler Current Profiler (ADCP) to characterize variability in streambed elevation and quantify pool density. Digital Elevation Models (DEMs) were generated using krigging procedures that interpolate streambed topography based on this bathymetric data. Data collected during each bathymetric surveys were converted to a DEM, which was then differenced from DEMs of different survey periods to quantify change in streambed topography (Wheaton et al. 2015, Williams et al. 2015). In addition, changes in streambed sediment substrate size were evaluated through sieve analyses of bulk sediment samples at each study reach (Tullos and Wang 2014).

Sampling occurred at approximately 3-month intervals for a total of 4 sampling campaigns over the course of the year beginning in June 2013 and extending until September of 2015. Sampling included surveys of all study reaches using an Acoustic Doppler Current Profiler (ADCP) and streambed substrate sampling. The June 2014 sample period is one exception during which substrate sampling did not occur. The current data set include a range of six to eight hydrogeomorphic sample periods that roughly co-occur with biological data collection.

Biological Data Collection: Fish were sampled using a combination of standard boat- and backpack electrofishing (e.g., Brousseau et al. 2005, Kautza and Sullivan 2012) protocols. Individuals were identified to species, weighed (mg) and measured (mm). Species were classified by ecological and life-history traits following Frimpong and Angermeier (2009). Sampling occurred during the summer for each year of the study. See Dorobek et al. (2015) for additional details related to fish survey methodology. We also assessed relationships between riffle structure and benthic macroinvertebrate (using Surber samplers) and fish assemblages two years following the lowhead dam removal at six time intervals from late spring 2014 through summer 2015. In addition to the biological and geomorphic data, temperature, conductivity, dissolved oxygen (DO), and pH were measured using a YSI 650 MDS® (YSI Inc., Yellow Springs, Ohio) with attached 600R® sonde at each quadrat during each sampling period. In addition, one 500-ml water sample was collected from each riffle (at middle of the thalweg) during each sampling period for total mercury (Hg), total nitrogen (N), total phosphorus (P), nitrate (NO_3), phosphate (PO_4) and ammonia nitrogen plus phosphate (NH_4 + PO_4). The samples were stored at 4°C and sent for analysis at the The Ohio State University Service, Testing, and Research (STAR) Laboratory, Wooster, Ohio.

Data analysis: For objectives 1 and 2, hydrogeomorphic and fish-assemblage changes through time were first evaluated individually. For objective 3, water-chemistry and hydrogeomorphic data were used as potential predictors of biological responses.

Topographic changes in riverbed morphology, specifically the magnitude and spatial patterns of erosion and deposition were quantified using differencing of DEMs from individual survey periods. We also evaluated changes in streambed substrate character using the D_{50} and D_{84}, representing the grain size diameter of the 50^{th} and 84^{th} percentile substrate, respectively, and which typically are used to represent reach-scale hydraulic conditions of bedload sediment transport (e.g, the D_{84} grain size is generally the upper end clast size that is transported as bedload during relatively low flood magnitude events, for example the 2-year recurrence interval event).

Changes in fish assemblages were evaluated in terms of species richness (*S*), diversity (*H*') and evenness (*E*). Two-sample *t*-tests were conducted to test for potential differences in fish assemblage *S*, *H*', and *E* before and after dam removal, +1 and +2 years after dam removal, upstream and downstream of dam removal, and between the upstream restored and unrestored Olentangy River experimental reaches. We used Non-Metric Multidimensional Scaling (NMS) followed by analysis of similarities (ANOSIM) to test for differences in fish assemblage composition (1) between control and experimental reaches upstream from dams (e.g., OR1 and OR3), (2) before and after dam removal and, (3) between Olentangy River control and experimental reaches (e.g., OR1, OR2, OR3, OR4) across successive years following dam removal.

Subsequent analyses will include general linear models (GLM) and multivariate regression to relate continuous variable descriptors of hydrogeomorphic change (e.g., spatial variability in streamflow velocity, streambed bathymetry, pool and bedform spacing) to measures of changes in fish assemblage characterstics (e.g., density, community diversity, relative abundance of foraging groups).

For Objective 3, a mixture of analysis of variance (ANOVA), linear mixed models, regression, and Non-metric Multidimensional Scaling (NMS) were used.

Principal Findings

Hydrogeomorphology

We report on findings for the study period extending from June 2013 through September 2015.

Streamflow Conditions

Streamflow conditions during the study period were characterized by an overall larger mean annual discharge but relatively small peak flows (e.g., less than a flow of a 2 year recurrence interval) in the Olentangy River indicating relatively mild to moderate hydraulic conditions to carry out geomorphic change. Summers 2013-2015 experienced elevated streamflows for at least some portion of this season (Figure 3). In addition, Winter 2015 experienced lower than normal streamflow magnitudes.

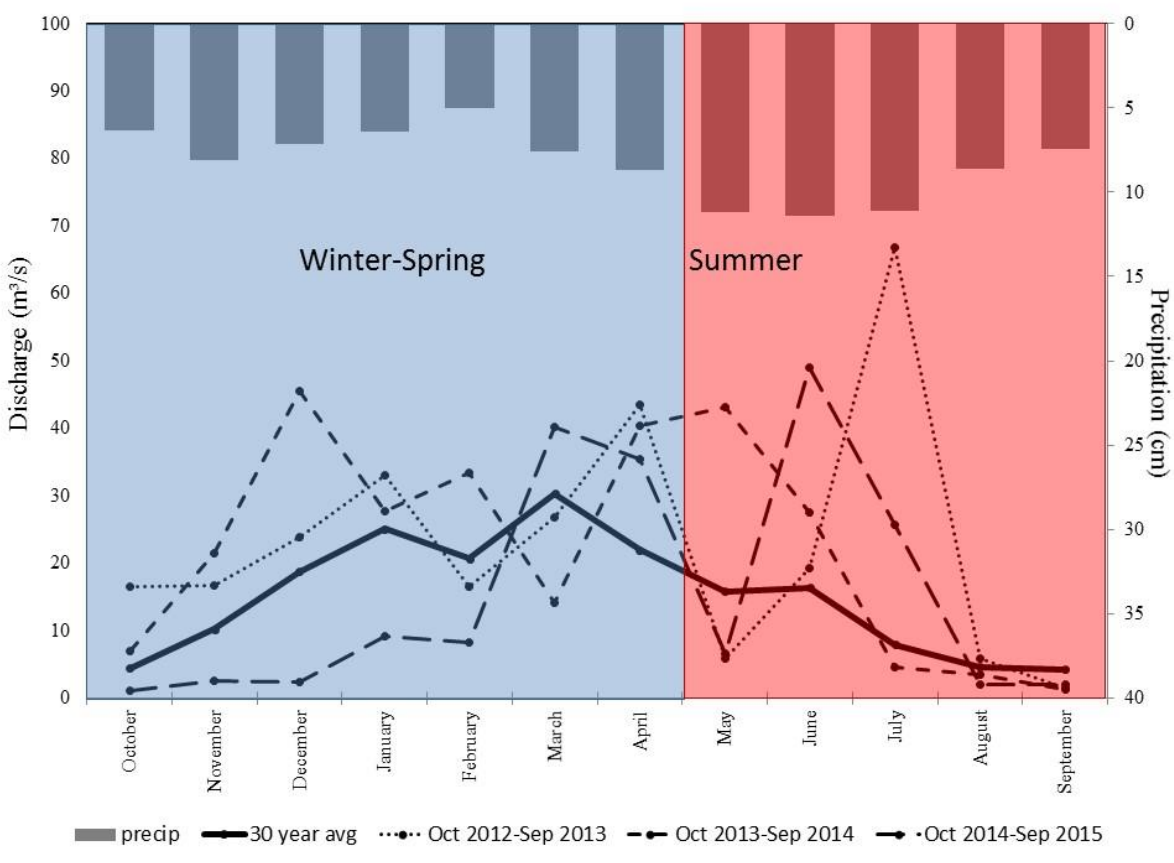

Figure 3. Monthly precipitation total and mean daily streamflow for each month over a 30-year period (1981-2010). Precipitation data are monthly normals from the NOAA National Centers for Environmental Information climate station at Columbus Ohio State University Airport, OH US. Discharge data is from USGS gage 03226800 Olentangy River near Worthington, OH monthly statistics from 1981-2010. Dashed lines are for Water Years 2013-

2015. Blue shading indicates winter and spring season (October-April) and summer (May-September), which approximately corresponds to the timing of the bathymetry surveys. Discrepancy between high summer precipitation normal values but lower normal streamflow values reflect regulation from upstream dams in the Olentangy River.

Changes in substrate composition

Changes in sediment grain size were observed over the study period (June 2013 – September 2015) and are reach specific. All study reaches are characterized by gravel (2 to 64mm) and finer material. Fines less than 2mm dominate all study sites with the exception of OR2 (the upstream, un-restored reached), which consistently had D_{50} larger than 2mm throughout the study period.

In general, reaches upstream and downstream of the removed dam are out of phase in terms of coarsening and fining behavior. In addition, coarsening and fining appear to correspond to different seasonal flows (Figure 4). In particular, coarsening in upstream reaches are concurrent with fining in the downstream reach and occur during winter and spring flows (e.g., Sep 2013-Apr 2014 and Sep 2014-June 2015). Conversely, fining in the upstream reaches are concurrent with coarsening in the downstream-of-the-removed dam reach and occur during summer flows (e.g., Apr 2014-Sep 2014 and Jun 2015-Aug2015). It is important to note that summertime flows involve several high flow events including the annual peak for the 2015 water year.

The abrupt increase in grain size diameter for the D_{84} at OR1 and OR4 in the second half of the study period beginning in September 2014 is attributed to changes in field sampling methods at these study sites (Figure 4). Prior to September 2014, sampling at OR1 had been limited to the right bank of the study reach but was extended to other coarser areas of the reach, notably a gravel bar located in the lower reach on river left. Similarly, the sample location at OR4 changed from one side of the river to the opposite side due to access issues, which was composed of coarser material. However, the out-of-phase pattern of OR4 relative to OR2 and OR3 occurs during consistent sampling methods beginning in September 2014 and therefore is interpreted as reflecting change in river sediment substrate character. Quantitative tests to evaluate change in sediment grain size through time will include comparison of observed change to background variability in grain size following Kibler et al. (2011). However, preliminary findings suggest that the upstream and downstream sites are behaving differently.

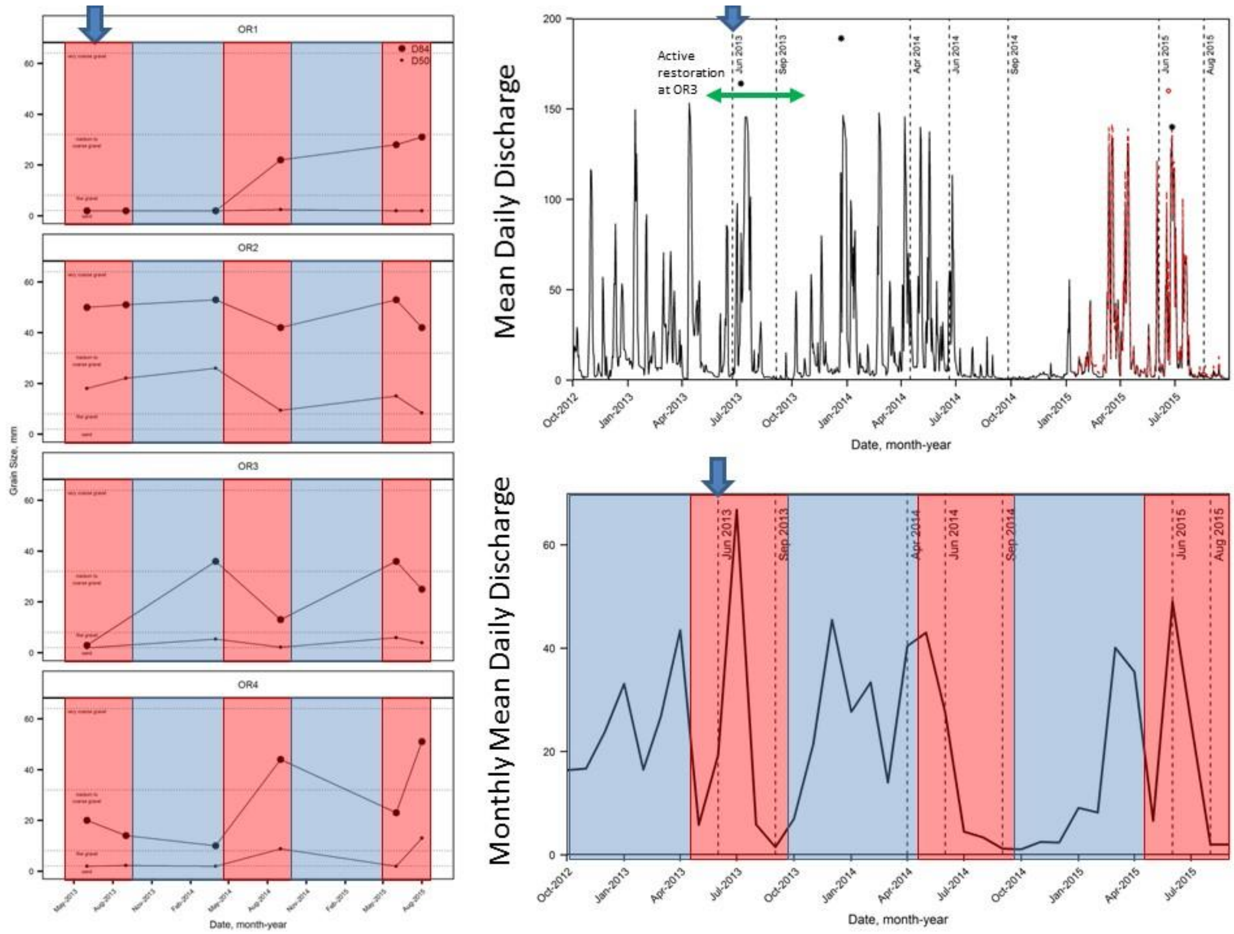

Figure 4. Changes in D50 and D84 at OR1-OR4 compared to streamflow for the study period. Left panels illustrate changes through time of D50a dn D84 clast sizes (mm). Top right panel illustrates mean daily discharge ($m^{-3}s^{-1}$) for the study period, vertical dashed lines indicate bathymetry studies. Bottom right panel illustrates mean daily discharge ($m^{-3}s^{-1}$) for each month for the study period. Red and blue shading reflect summer and winter-spring seasons, respectively. Blue arrow indicates start of first bathymetry and sediment data collection sampling period.

Changes in bathymetry

Analysis of bathymetric change is still in progress. DEMs have been created from bathymetric surveys and initial DEMs of difference have been generated. Initial analysis of DEMs of difference (DoD) indicate that observed changes in streambed bathymetry are reach specific. Changes in streambed bathymetry in the upstream unrestored reach, OR2, was concentrated along the thalweg, which occurs along the outside of a gentle bend on the left side of the river. Sequences of net erosion and net deposition occurred throughout the study period, and loosely correspond to seasonal patterns in streamflow discharge and substrate grainsize observed at this reach. In particular, net deposition tended to occur during winter/spring streamflows; net erosion corresponded to summer streamflows. These findings suggest that winter and spring streamflows deposit new material within this reach, which may also be the reason for observed coarsening of streambed substrate during this time period. Observed net erosion of the streambed and concurrent fining of the streambed substrate during summertime flows was unexpected, since erosion and coarsening tend to be linked, suggesting that observed changes were a result of upstream streamflow and sediment delivery and not downstream dam removal. Collectively, the erosion and depositional behavior at this upstream reach does not suggest a strong influence of the downstream removed dam.

The upstream restored reach, OR3, experienced net erosion during most bathymetric surveys that do not coherently correspond to substrate grainsize changes. This site experienced substantial in-channel engineering activities, including heavy machinery within the channel and riverbed regrading. Observed erosion may reflect ongoing adjustment of the newly engineered channel, which may overwhelm any potential influence of the removed downstream dam.

In the downstream reach OR4, DoDs indicate patterns of concurrent deposition and erosion within a given survey instead of dominant reach-scale erosion or deposition for a given survey observed in the unrestored and restored upstream reaches. Finally, OR1 experienced some periods of net deposition and erosion underscoring the background variability of this river system that seem to correspond to seasonal differences in streamflow magnitudes and sediment supply.

Coarsening may be a result of either (1) within-reach fines being winnowed leaving coarser clasts in place or (2) new coarse material being transported into and deposited within the reach. Interestingly, the out-of-phase behavior between upstream and downstream of the removed dam reaches suggests that response to streamflow and sediment transport dynamics are not the same system-wide, but are influenced by reach scale hydraulic conditions. For example, we attribute fining in the upstream reaches during summertime flows to sediment laden streamflows during this period that are causing deposition of fines, but this does not necessarily translate to the same response in the downstream reach.

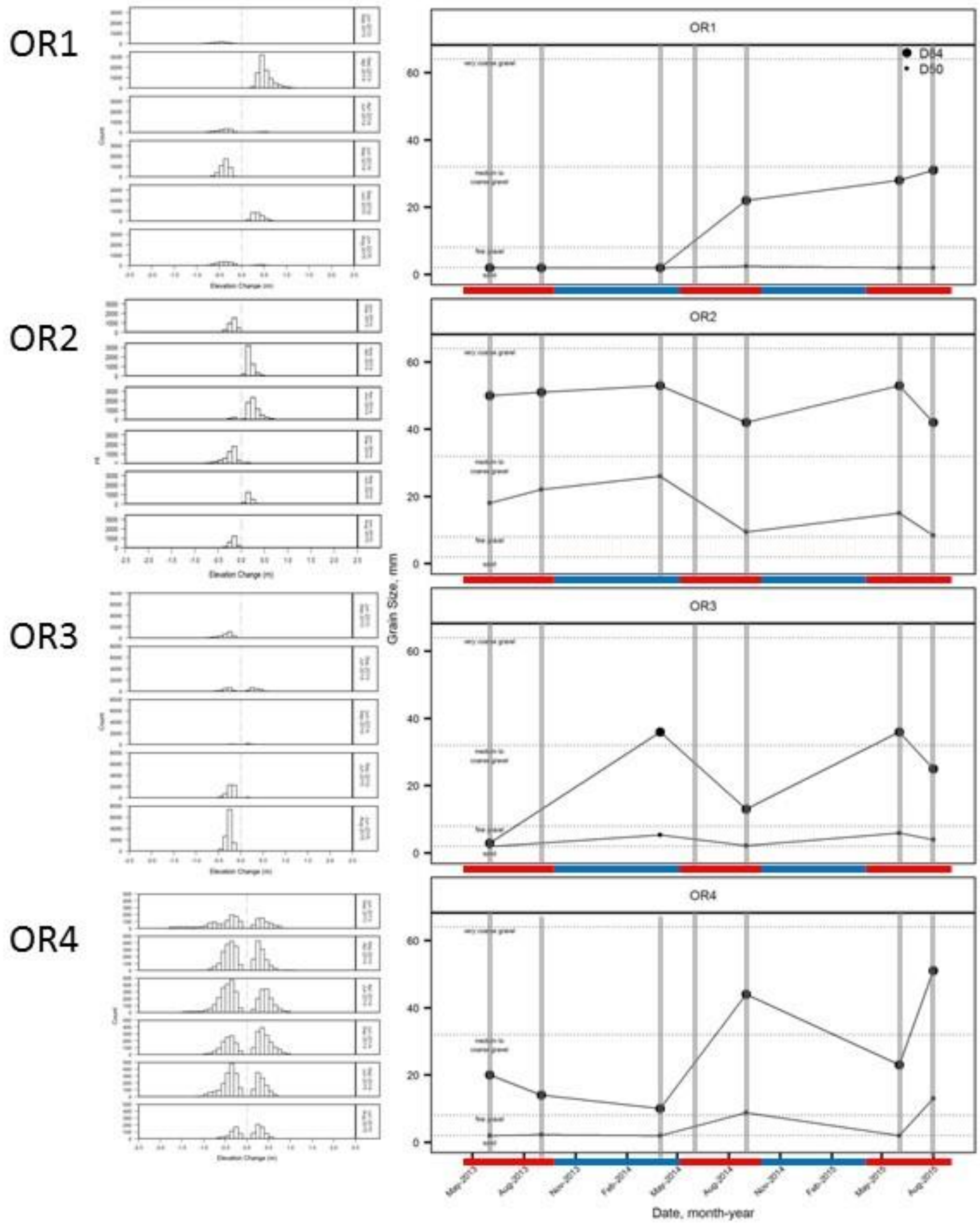

Figure 4. Comparison of reach-scale bathymetric changes and representative substrate clast sizes during the study period at four study sites (OR1-OR4) in the Olentangy River. Left panels are frequency distribution of DEM of Differencing (DEM_{Time2}-DEM_{Time1}) values for each 1-m grid cell within each study reach. Negative values indicate erosion. Positive values indicate deposition. Right panels are reach-averaged changes in D50 and D84 substrate sizes for the study period. Vertical lines indicate bathymetry surveys. Red and blue bars represent summer and winter-spring seasons.

Ecological component

Ecological sampling was conducted prior to and for three years following dam removal. This project investigated the consequences of lowhead dam removal for (1) fish assemblage structure and (2) the responses of benthic macroinvertebrate and fish community to riffle development following dam removal. For the purposes of this report, we have included data thru August 2014 for (1) and thru August 2015 for (2).

Ecological sampling was conducted prior to and after dam removal during the summer and early fall. Upstream fish assemblage (at restored reach) composition shifted significantly and was accompanied by a significant decrease in species richness and diversity. These changes represented changes in the relative abundance of taxa within different feeding guilds. Specifically, reductions in species richness and diversity at the upstream reaches were accompanied by the loss of large-bodied omnivorous species. Between year 1 and year 2 post-dam removal, diversity increased significantly at the upstream restored and downstream reaches. Species richness increased significantly at the upstream restored reach and showed an increasing trend at the upstream unrestored and downstream reach. Shifts in fish assemblages as the upstream restored and downstream reaches were accompanied by a substantial increase in insectivorous species including an increase in darter (*Etheostoma* spp.) species.

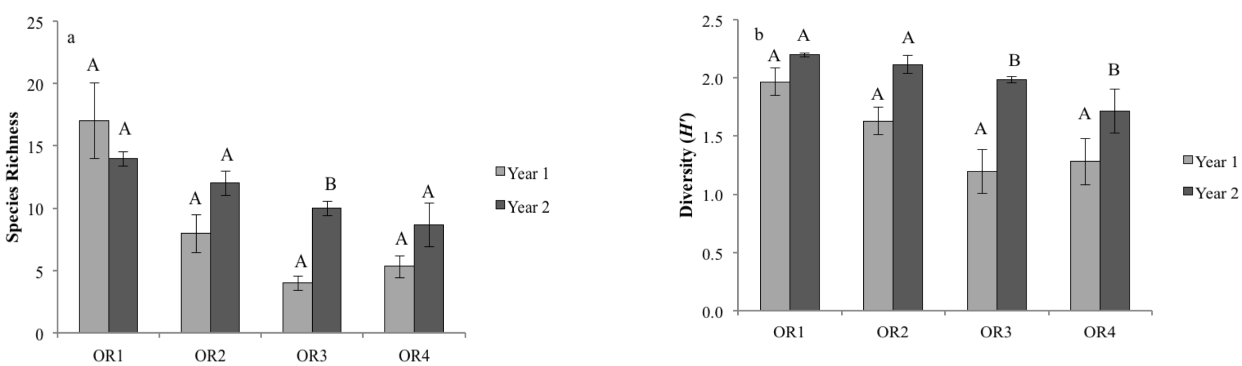

Figure 5. Fish assemblage (a) species richness and (b) diversity (*H*') in years 1 & 2 following dam removal of the Olentangy River study reaches. OR1 is the upstream of an existing dam control reach; OR2 is the upstream of the removed dam, unmanipulated experimental reach; OR3 is the upstream of the removed dam, restored experimental reach; and OR 4 is the downstream of the removed dam experimental reach. Significant differences based on *t*-tests are indicated by different letters ($p < 0.05$). Error bars represent $\pm$1 SE from the mean. From Dorobek, Sullivan, and Kautza (2015).

For the component of the study related to riffle development following dam removal, we found that the density and diversity of macroinvertebrates and fish were significantly different over time, largely as a function of season (lowest densities in early spring, greatest in summer). Nevertheless, we also found within-season changes (e.g., macroinvertebrate density declined by 50% from summer 2014 to late spring 2015). Macroinvertebrate assemblage composition was different by time but not riffle, whereas fish assemblages were similar irrespective of time or riffle. Sediment-size distributions, water depth, and streamflow velocities varied by riffle and over time. Overall, chemical water quality (e.g., phosphate [PO_4] and dissolved oxygen [DO]) was more strongly related to macroinvertebrate communities and hydrogeomorphic parameters (e.g., streamflow velocity and substrate size) were more strongly related to fish.

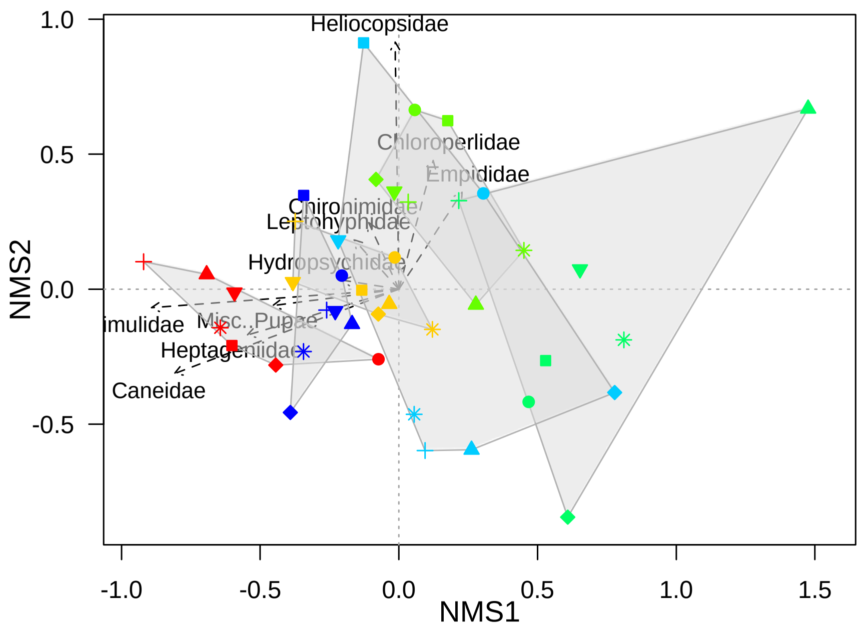

Figure 6. Non-metric Multidimensional Scaling (NMS) ordination plots of benthic macroinvertebrate assemblage compositions grouped by date (scaled by variance). The stress level was 22%, respectively. The different shapes indicate the different riffles and the different colors indicate the different sampling time periods; only the most significant families are indicated. Dates: June 2014 = red, August 2014 = yellow, November 2014 = green, March 2015 = cyan, June 2015 = blue. Riffles are shown by symbol: Riffle 1 = circle, Riffle 2 = square, Riffle 3 = diamond, Riffle 4 = triangle (up), Riffle 5 = triangle (down), Riffle 6 = asterisk, Riffle 7 = plus. From Cook and Sullivan (In Review).

Significance

This project quantifies linked geomorphic and ecological response to removal of a lowhead dam with respect to fish community assemblages, as well as how lowhead dam removal influence riffle development and both fish and macroinvertebrate communities. The geomorphic influence of dam removal on fish community assemblages builds on documented community shifts in aquatic biota following dam removal. Collectively, our results indicate that geomorphic alterations, in conjunction with water-quality changes, may be key mechanisms related to restoring biological diversity following dam removal. This study represents a higher level of system integration with intensive coupled geomorphic and ecological metrics than is common, and the results will be useful in informing future dam removal efforts.

Literature Cited

Brousseau, C., R. Randall, and M. Clark. 2005. Protocol for boat electrofishing in nearshore areas of the lower Great Lakes: transect and point survey methods for collecting fish and habitat data, 1988 to 2002. Canadian Manuscript Report of Fisheries and Aquatic Sciences **2702**.

Downes, B. J., L. A. Marmuta, P. G. Fairweather, D. P. Faith, M. J. Keough, P. S. Lake, B. D. Maptson, and G. P. Quinn. 2002. Monitoring ecological impacts: concepts and practice in flowing waters. Cambridge University Press, Cambridge, UK.

Frimpong, E. A. and P. L. Angermeier. 2009. FishTraits: A Database of Ecological and Life-history Traits of Freshwater Fishes of the United States. Fisheries **34**:487-495.

Gangloff, M. M. 2013. Taxonomic and ecological tradeoffs associated with small dam removals. Aquatic Conservation-Marine and Freshwater Ecosystems **23**:475-480.

Graf, W. L. 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. Water Resources Research **35**:1305-1311.

Hart, D. D., T. E. Johnson, K. L. Bushaw-Newton, R. J. Horwitz, A. T. Bednarek, D. F. Charles, D. A. Kreeger, and D. J. Velinsky. 2002. Dam removal: Challenges and opportunities for ecological research and river restoration. Bioscience **52**:669-681.

Kautza, A. and S. M. P. Sullivan. 2012. Using a process-based catchment-scale model for enhancing field-based stream assessments and predicting stream fish assemblages. Aquatic Conservation-Marine and Freshwater Ecosystems **22**:511-525.

Poff, N. L. and D. D. Hart. 2002. How Dams Vary and Why It Matters for the Emerging Science of Dam Removal: An ecological classification of dams is needed to characterize how the tremendous variation in the size, operational mode, age, and number of dams in a river basin influences the potential for restoring regulated rivers via dam removal. BioScience **52**:659-668.

Stanley, E. H. and M. W. Doyle. 2002. A geomorphic perspective on nutrient retention following dam removal. Bioscience **52**:693-701.

Stanley, E. H. and M. W. Doyle. 2003. Trading off: the ecological removal effects of dam removal. Frontiers in Ecology and the Environment **1**:15-22.

Stewart-Oaten, A., W. W. Murdoch, and K. R. Parker. 1986. ENVIRONMENTAL-IMPACT ASSESSMENT - PSEUDOREPLICATION IN TIME. Ecology **67**:929-940.

Soil development on coal mine tailings; impact on trace metal sources and mobility to acid mine drainage

Basic Information

Title:	Soil development on coal mine tailings; impact on trace metal sources and mobility to acid mine drainage
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End Date:	2/28/2016
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Congressional District:	OH-013
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Focus Category:	Geochemical Processes, Non Point Pollution, Water Quality
Descriptors:	None
Principal Investigators:	David M Singer

Publications

1. Zemanek, L., Herndon, E., Singer, D.M., 2015. A Geochemical and Mineralogical Comparison of Soil Formation on Mine Tailings and a Shale Hill and their Contribution to Stream Chemistry, Huff Run Watershed, Ohio, 250th ACS National Meeting, Boston, MA.
2. Herndon, E., Singer, D.M., Zemanek, L., 2016. Metal(loid) leaching from soils developing on coal mine waste, 251st ACS National Meeting, San Diego, CA.

FINAL REPORT

Problem and Research Objectives

Mine spoil is an obligate waste product of coal mining and energy production. These materials are chemically similar to the parent material but are no longer economically viable to continue removing coal. Although chemically similar, the physical properties of the solid phase have changed dramatically, primarily due to a large increase in surface area and porosity. The treatment and storage of these materials is an inherent cost of coal-based energy production. The primary environmental impact of coal mine spoil is the generation of acid mine drainage (AMD). AMD is the result of the oxidation of exposed sulfide minerals in abandoned coalmines and unreclaimed coal refuse piles in circulating rain- and groundwater, generating highly acidic, metal-rich fluids that are then discharged into the local environment. AMD is estimated to impair more than 12,000 km of streams in the eastern USA (**Figure 1**).[1] AMD discharge has severe, long lasting impacts on water quality and stream ecology in affected watersheds. The remediation of environmental damage caused by these mines is also extremely costly. Between 2005 and 2012, monitoring and reclamation of over 300 km of streams and rivers in Ohio was done at a cost of over $25 million dollars.[2] In the US, AMD and other toxins from abandoned mines have polluted 180,000 acres of reservoirs and lakes and 12,000 miles of streams and rivers.[3] It has been estimated that cleaning up these polluted waterways will cost US taxpayers between $32-72 billion.[4] Typical AMD treatment systems include a series of passive remediation cells that remove acid or metals from the waste stream.[5] These system primarily target point-sources of AMD including abandoned coalmines and mine ponds. However, historic waste spoil adjacent to these point-sources acts a non-point source of AMD and can continue to contribute to the acid and metal loading of impacted waterways despite up-stream treatment.

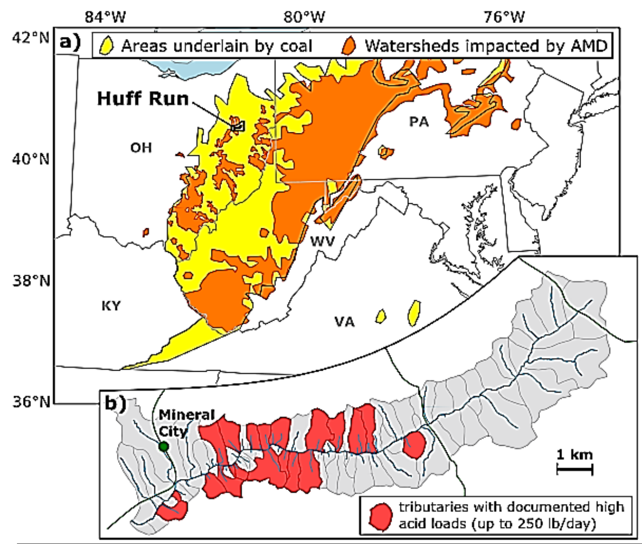

Figure 1. Huff Run site map; based on surveys by the US-EPA and Ohio Department of Natural Resources. Figure courtesy C. Rowan (KSU).

Although there is extensive literature on the reclaimed and remediated coal mine tailings (for review, see [5, 6]), few studies have addressed the fate of trace metals in historic mine tailings, which were abandoned in the early 20th century, and now blend into the surrounding landscape. These topographic highs are known to be long-term sources of slow-leaching AMD adjacent to reclamation projects that can hinder effective remediation.[7, 8] For example, the Linden-Lindentree passive remediation project (**Figure 2**) in the Huff Run Watershed cost over $590,000 to complete[7, 9]. Treated-AMD from the first four stages drains into the far side of a settling pond, which then drains into the Huff Run from a culvert at the bottom right of the image. Despite up-stream remediation, the pond often has high metal loadings and low pH values due to leaching from the untreated mine tailings that make up the hillside on both sides of the pond. Mine tailing-degraded soils are anthropogenic-impacted habitats, which experience a wide-range of problems that hinder the establishment and maintenance of healthy soils.[10] Ultimately, if the reclamation of an AMD-impacted area is going to be successful, the leaching of metals and acid from historic mine-tailings must be addressed. Despite the significance of long-term soil development on mine tailings, these processes are not specifically addressed and monitored during development and construction of AMD reclamation projects. Without an understanding of how critical geochemical processes which occur during soil development limit or promote metal and acid mobility, restoration will continue to be ineffective at providing an ecological benefit to the state.

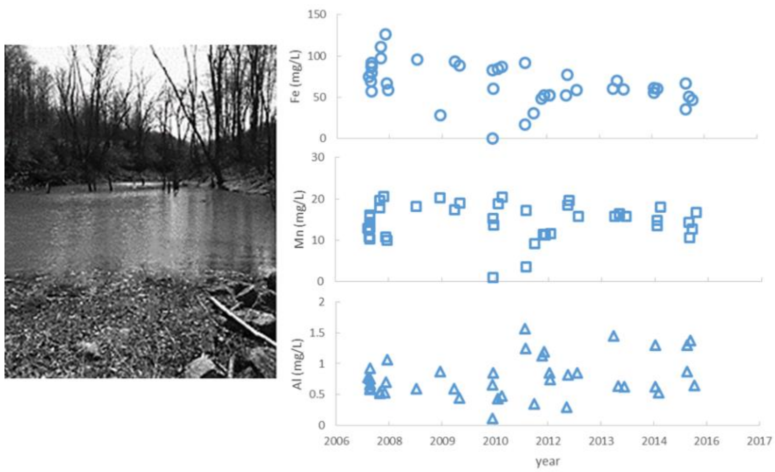

Figure 2. This settling pond, (picture at left), ~10 m in diameter, is the fifth and final stage of the Linden Restoration project, and is buttressed by hills made of mine spoil. Although Fe has decreased slightly over time (top panel), concentrations of Mn and Al remain high in the tributary (middle and bottom panel) after discharge from the settling pond. Data from watersheddata.com.

This project investigated the effects of soil development on historic coal mine tailings and the geochemical processes which control trace metal mobility over time. Specifically, we aimed to determine how do geochemically-driven mineralogical transformations in soils developing on coal mine tailings impact trace metal mobility. The main project objective was to determine mineralogical abundance and trace metal concentration in soils developing on coal mine tailings vertically through a soil profile. The work focused on the Huff Run watershed because of the magnitude of AMD challenges and efforts towards improvement in the region. To achieve the objective, measurements of spatial and temporal changes in mineralogy and trace metal mobility were performed. These measurements allowed for an integrated understanding of the geochemical processes driving these changes, and investigate their role in impacting water quality at the meter- to micron-scale.

Methodology

Field work was focused on the heavily AMD-impacted Huff Run watershed (**Figure 3**). The extent of groundwater contamination in the Huff Run region by AMD is currently unknown. Since 1996, 18 AMD remediation projects have been built in the watershed at a cost of over $4.5 million.[8] Sites within the Huff Run have metal discharges of up to 250 lbs/day, dominated by iron and aluminum.[8] Restoration at sites such as Huff Run target discharge from surface and below ground mines, but typically do not target leaching from historic mine tailings. Surface mine spoil, a mixture of compacted, partly weathered fine-earth material and fragments of shale and other rock fragments account for nearly a third of the surface material in the Huff Run Watershed.[11, 12]

The study area is situated within the unglaciated portion of Ohio. The geology of the area is dominated by Pennsylvanian-aged bedrock with exposed strata of sandstone, shale, coal, limestone and iron ores. The watershed topography is dominated by drainages that have cut deep valleys and left narrow ridge tops.[11] Soils from two locations within the Huff Run (HR) Watershed were examined (**Figure 4**). The first was soils developing on historic coal mine tailings at the HR-

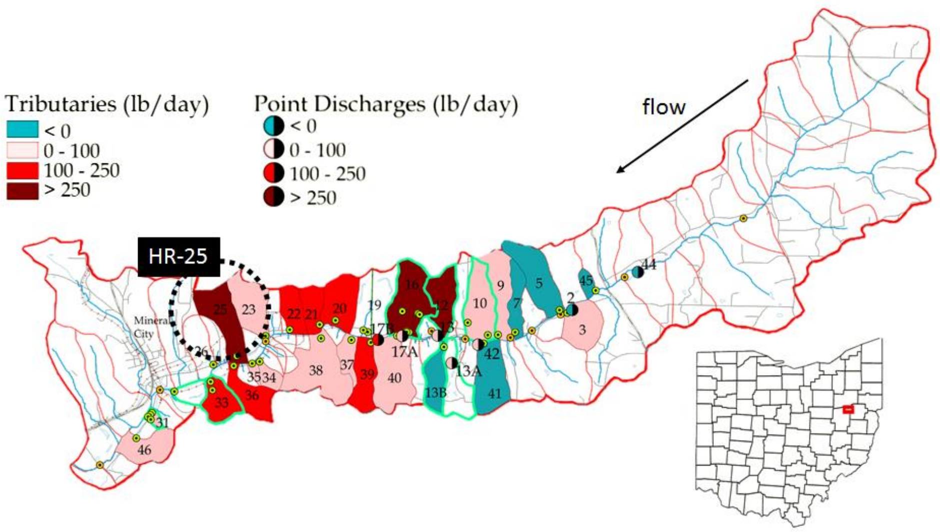

Figure 3. Location of study site (HR-25) within the Huff Run Watershed. Iron discharge from tributaries and point sources are shown. Data from Kinney (2013).

25 site, which has been found to have the worst water quality throughout the watershed.[7] Because of its downstream location, the project has not been a priority until reclamation projects upstream have been completed. The second site examinee soils developing on a shale outcrop northeast of HR-25 within the Huff Run watershed, where AMD is not detected. This provided an important comparison and baseline of trace metal mobility in undisturbed parent material.

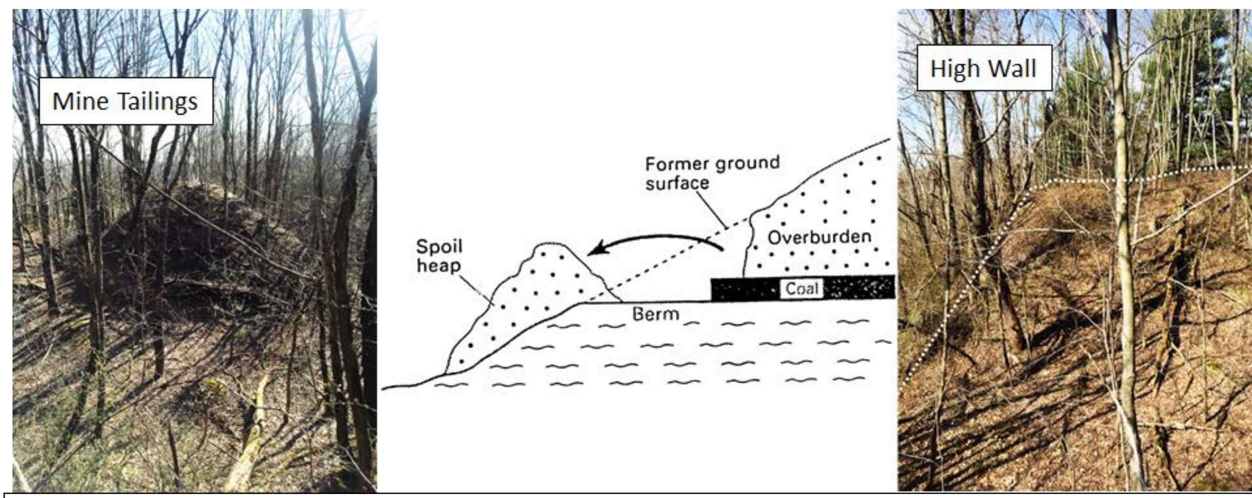

Figure 4. Schematic (center) of the relationship between the highwall and mine tailings (spoil) created during mining operations[13]. The images show a typical conical waste pile (left) and the high wall (right) where the dashed line represents the surface.

Historical records of mine tailing emplacement are limited or non-existent within the watershed. However, the presence of Black Locust trees (*Robinia pseudoacacia*) with diameters of 0.6 m, an indication the trees are ~ 50 years old, suggests the minimum local timescale needed for tailings stabilization. This timescale is consistent with the cessation of active mining within HR-25 in the mid-20th century. Trees cores were collected (**Figure 5**) to provide a lower boundary for how long soils have been developing on these tailings.

Figure 5. Laura Zemanek (MS) collected soil cores (left) which were transported in drinking straws back to the lab to be dried (right). The minimum age of soil development (~50 years) for mine tailings was established by counting tree rings

Soil geochemical analyses

Solid phase characterization of soils was performed on samples collected in 10 cm depth increments from the soil surface to 1.2 m depth in the soil profile. After drying, the soil samples were ground to silt size particles (10-75 micron) with a SPEX-8000M ball mill using tungsten carbide ball bearings and analyzed by: (1) bulk X-ray diffraction (XRD) to determine the dominant mineral phases present; (2) a sequential extraction procedure (discussed below) to determine the concentration of metals associated with the solid-phase as a function of depth; and (3) loss on ignition, as a proxy for organic matter content.

Soil water analysis

A series of lysimeters were installed near where the soil cores are sampled at both field sites to collect pore solutions for trace metal analysis over a 6-month period **(Figure 6)**. This method allows continuous sampling during any period and at several different depths of a soil profile. The installation of the suction probe is easy and the profile is only negligibly disturbed.[14] Lysimeters were installed at 10 cm increments, with one lysimeter per installation well. The wells were spaced at 10 cm intervals on the ground surface to minimize profile disturbances. In order to get good hydraulic contact between the suction cup at the end of the lysimeter and the soil, a slurry of the material from the soil auger was made and put back into the hole before inserting the suction probe.[14] Water was prevented from seeping from the surface down through the shaft as this causes hydraulic short circuits, by sinking the probe completely into the soil and using a collar around the top of the shaft. The installation of the suction probe was followed by a stabilizing phase, including simultaneous water sampling, to precondition the suction cup. Suction was put on the lysimeter using a 2005G2 Vacuum Hang Pump at a pressure of 60 kPa to create a negative pressure inside the soil water sampler. The first sample was rejected in each location. The lysimeters were sampled weekly, and filtered samples will be analyzed for cations (ICP-OES), anions (IC), and total dissolved carbon.

Figure 6. Lysimeter installation photos (May 2015), top left, showing (from left to right) Laura Zemanek (MS), Jonathan Mills (BS), Elizabeth Herndon (KSU assistant professor), and Mikala Coury (BS). At right, soil core samples being bagged and labeled by Laura Zemanek. The bottom left image shows the installed lysimeters from the high wall site; the dashed line represents the cliff edge.

Sequential Extraction

Sequential extraction modified after Tessier (1979) was performed on soil samples from all sampling depth, with triplicates performed at depths of lysimeter installation of Al, Fe, and Mn in soil fractions including exchangeable cations, oxide minerals (reducible), and organic matter/sulfides (oxidizable). Powdered soil (1 ± 0.0523 g) was agitated with four sequential solutions to remove ions from operationally defined phases. After agitation with each solution, the sample was centrifuged at 4,000 rpm for 30 minutes and decanted. 10 mL of Milli-Q water was then added and centrifuged for an additional 30 minutes to remove any remaining solution and ions before adding the next extraction solution. This rinse solution was set aside to use as a dilutant before running on the ICP-OES. The first extraction step targeted exchangeable ions; 8 mL of 2M NaCl solution at room temperature with a pH of 7, was agitated with the soil for 2 hours. The second extraction step targeted ions bound to iron and manganese oxides; a 20 mL solution of 0.3 M $Na_2S_2O_4$ (Sodium Dithionite),0.175 M Na-citrate, and 0.025 M H-citrate was agitated with soil for 6 hours at room temperature. The third and final extraction step targeted ions bound to organic matter;3 mL of 0.02 M HNO_3 and 5 mL 30% H_2O_2, adjusted to pH 2 by adding additional H_2O_2, was heated with the soil samples to a temperature of 85ºC for 2 hours, with occasional agitation. After cooling, 5 mL of 3.2 M NH_4OAc in 20% (v/v) HOAc was added, with

Milli-Q water added to reach 20 mL if needed. The extracted solution along with the rinse solution was analyzed by ICP-OES for Fe, Mn, Al, Cu, Ni, As, and Se.

Principal Findings and Significance

Preliminary XRD analysis of the soil core samples (data not shown) indicated the presence of typical shale weathering products; quartz, feldspars, kaolinite/illite. Crystalline Fe-bearing phases are dominated by goethite. On-going work aims to quantify the distribution of minerals in the soil cores.

Soil pore water chemistry, averaged over the field season, is shown in **Figure 7**. The pH and dissolved oxygen concentrations of pore water from the mine tailings were lower, consistent with a greater degree of coal weathering leading to AMD release. Metal solubility increases near the soil surface, but differs between sites; Fe and Al are more mobile in the highwall; Mn is more mobile in mine tailings. Interestingly, sulfate is lower in the mine tailings pore water, which was not expected as we hypothesized that greater AMD production would also result in increased sulfate concentrations. It is possible that high sulfate concentrations result in gypsum ($CaSO_4 \bullet H_2O$) precipitation which should be resolved in the on-going XRD analyses.

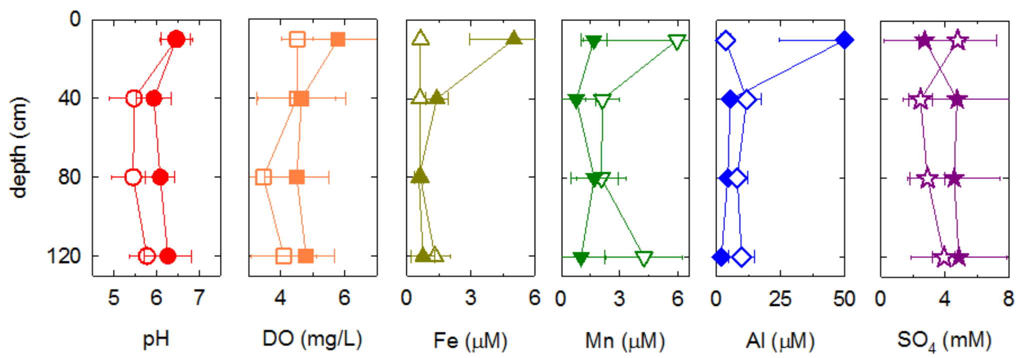

Figure 7. Soil pore water chemistry; average values from May-November 2015 for the highwall (closed symbols) and mine tailings (open symbols) lysimeters.

Loss-on-ignition was used as a proxy for solid phase organic matter, and coupled with DOC measurements of the soil pore water to provide an anlysis of the fate of C in this system (**Figure 8**). The mine tailings contained dark, organic-rich soil at depth. Dark lenses were observed in the cores and are likely dominated by coal-bearing waste materials. DOC was high for the shallow highwall lysimeters, consistent with heavier vegetation cover and root exudates. In pore water, plant-derived organic C may mobilize Al and Fe in the undisturbed highwall soils (discussed below).

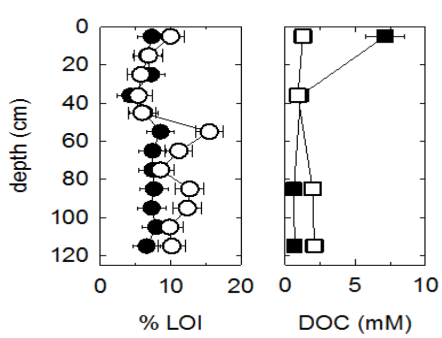

Figure 8. Loss-on-ignition (left) and dissolved organic carbon (right) for the highwall (closed symbols) and mine tailings (open symbols)

A sequential extraction procedure was used to quantify the distribution of Fe, Al, and Mn (**Figure 9**) and other metals (not shown). This can aid in determining what the potential is for these soils to leach metals to surface water. In the exchangeable fraction, Fe was not detected, and the mine tailings are relatively enriched in exchangeable Mn that increases near the soil surface.

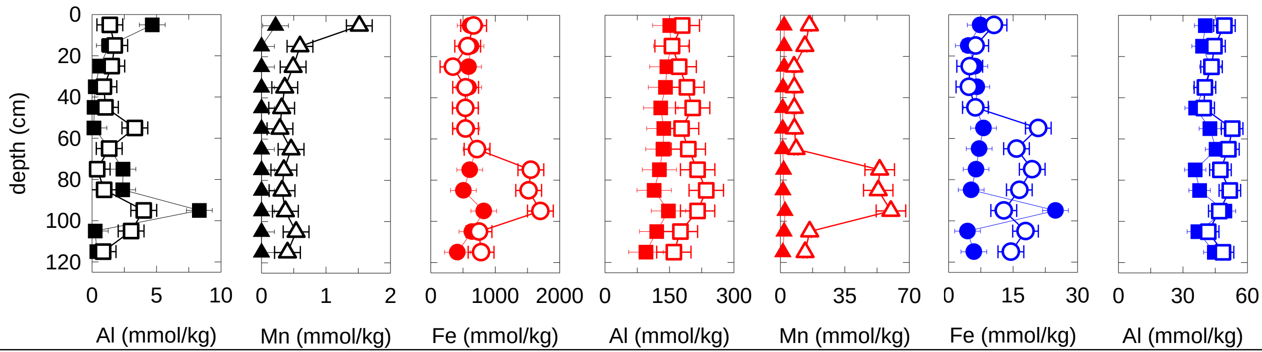

Figure 9. Sequential extraction data for Fe (circles), Al (squares), and Mn (triangles) from the highwall (closed symbols) and mine tailings (open symbols) soil cores. The three fractions are exchangeable (black), reducible (red), and oxidizable (blue).

An organic-rich zone (**Figure 8**) was also enriched in Fe and Mn oxides, potentially derived from pyrite oxidation. Small fractions of Fe and Al (< 3%) were observed in the oxidizable fraction; Fe (either as sulfides or organic-bound) tracks with percent loss-on-ignition. These results suggest that a pool of Fe, Mn, and Al can continue to be mobilized during weathering and impact downgradient water.

Our current conceptual framework for our results is that at depth, the mine tailings contain organic-rich zones where pyrite is weathering to form Fe and Mn-oxides. These Fe and Mn-oxide phases may compete for trace metals (e.g., As) released during weathering. Near the surface, high concentrations of labile, plant-derived DOC in highwall soils complex and mobilize Fe and Al deeper into the soils. Finally, the mine tailings are a potentially larger source of Mn to streams than previously understood.

Impact and Significance

One expected outcome of this project was knowledge to guide AMD reclamation projects regarding how to address metals and acid leaching from soils developing on mine tailings, and thus improve stream quality. This research provided data that are directly transferrable to planning future AMD reclamation projects and evaluating their success. Further, this research is novel in its approach to understanding soil developing on historic coal mine tailings in Appalachian Ohio. Integration of physical and chemical data allowed us to describe how multiple processes interact, developing an important understanding that can be used by practitioners planning reclamation design. This information will allow future reclamation projects to be more effective in the long-term management of AMD to Ohio rivers and streams. Results of this project will benefit reclamation practitioners, regulators, and the scientific community through an organized plan for dissemination of the findings. Data generated from this project has been presented at state and national scientific meetings. We intend to publish our results in the journal *Applied Geochemistry*. Finally, we have continued to meet and collaborate with representatives from the Ohio Department of Natural Resources (ODNR) and from the Huff Run Watershed Restoration Partnership (HRWRP) who work with landowners in the area. The ultimate goal of the Partnership is to return Huff Run to its original, warm-water habitat. We anticipate that our findings can be employed to guide reclamation designs in Appalachian Ohio and beyond.

References Cited

1. Boyer, J.; Sarnoski, B., Progress report—statement of mutual intent strategic plan for the restoration and protection of streams and watersheds polluted by acid mine drainage from abandoned coal mines. *US Environmental Protection Agency* **1995**.
2. Bowman, J.; Johnson, K. *2012 Stream Health Report: An Evaluation of Water Quality, Biology, and Acid Mine Drainage Reclamation in Five Watersheds: Raccoon Creek, Monday Creek, Sunday Creek, Huff Run, and Leading Creek.*; Voinovich School of Leadership and Public Affairs at Ohio University: 2012.
3. Reece, B. A. *Acid Mine Drainage: Perpetual Pollution*; Mineral Policy Center: 1995.
4. Woody, C. A.; Hughes, R. M.; Wagner, E. J.; Quinn, T. P.; Roulson, L. H.; Martin, L. M.; Griswold, K., The mining law of 1872: change is overdue. *Fisheries* **2010,** *35*, (7), 321-331.
5. Johnson, D. B.; Hallberg, K. B., Acid mine drainage remediation options: a review. *Sci. Total Environ.* **2005,** *338*, (1–2), 3-14.
6. Hossner, L. R.; Hons, F. M., Reclamation of Mine Tailings. In *Soil Restoration*, Lal, R.; Stewart, B. A., Eds. Springer New York: 1992; Vol. 17, pp 311-350.
7. Ohio Watershed Data (Huff Run Watershed Restoration Partnership). In Voinovich School of Leadership and Public Affairs at Ohio University: http://www.watersheddata.com/map/map.aspx?WaterShed=HR1.
8. Wise, M. *Huff Run Watershed Plan*; Huff Run Watershed Restoration Partnership, Inc.: Mineral City, Ohio, 2005.
9. Kinney, C., Huff Run Acid Mine Drainage Abatement and Treatment Plan Addendum (Available from watersheddata.com). **2013**.
10. Wong, M. H., Ecological restoration of mine degraded soils, with emphasis on metal contaminated soils. *Chemosphere* **2003,** *50*, (6), 775-780.
11. Waters, D. D.; Roth, L. *Soil survey of Tuscarawas County, Ohio*; United States Department of Agriculture: 1986.
12. Gerber, T. D.; Buzard, R. W. *Soil Survey of Carroll County, Ohio*; United States Department of Agriculture: 1983.
13. Evans, A. M., *An introduction to economic geology and its environmental impact*. John Wiley & Sons: 2009.
14. Grossmann, J.; Udluft, P., The extraction of soil water by the suction-cup method: a review. *Journal of Soil Science* **1991,** *42*, (1), 83-93.

Spatial and Temporal Dynamics of Non-Point Source Pollution

Basic Information

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2. Apostel, A., Herak, P. and Sivandran, G., 2015. Comparing Methods for the Identification of Critical Source Areas in an Effort to Establish Total Maximum Daily Load Requirements, June 14th 2015, Poster Presentation, Gordon Research Seminar- Catchment Science: Interactions of Hydrology, Biology & Geochemistry.
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Progress Report 2015
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Summary

Despite extensive and increasing investment by the USDA since 1987, agricultural non-point source (NPS) pollution remains the leading cause of water quality problems in the United States and a serious concern for policy makers, scientists, and the residents who rely on impacted waters. The Upper Big Walnut Creek (UBWC) watershed in central Ohio provides drinking water for approximately 600,000 residents of Columbus and surrounding communities. The majority of headwater streams in this watershed are impaired by nutrient enrichment, pathogens, and habitat degradation stemming from agricultural management practices. Existing pressures on the watershed may also be exacerbated by local impacts of global climate change.

Global climate change is predicted to increase climate variability, altering the distribution of environmental variables that influence agricultural and hydrological processes (e.g. longer growing seasons, more frequent extreme precipitation events, and increasing periods between events). Under an uncertain climate future, it is critical to understand how all linked processes—biogeochemical, hydrological, and land management—could accelerate and exacerbate the impacts of NPS pollution in the UBWC Watershed.

The overarching goal of this research is to identify the hydrologic and land surface characteristics that influence the spatial and temporal dynamics of NPS pollutants (nitrogen and phosphorus, in particular). To achieve this objective the following activities will be undertaken for the UBWC watershed: (i) Assessment of a suite of existing indexing methods used to identify critical source areas (CSAs); (ii) Application of the Soil and Water Analysis Tool (SWAT) to quantify pollutant load for comparison of SWAT derived CSAs to those identified by existing methods.

By developing relationships that link hydrologic, biogeochemical and land management characteristics to the spatial and temporal dynamics of NPS pollutants, the identification of chronic and acute CSAs will be achieved for the UBWC watershed. This methodology is generalizable and applicable to any agricultural based watershed with NPS pollution issues. The CSAs developed by this research will aid water resource managers by providing a method to prioritize the deployment of conservation measures and monitoring equipment.

Spatial and Temporal Dynamics of Non-Point Source Pollution

1 Statement of Regional Water Problem and Research Benefits

The August 2014 drinking water emergency in the City of Toledo, Ohio affecting over 400,000 residents was largely triggered by non point source (NPS) pollutants, namely phosphorus and nitrogen, originating from agricultural lands. This event has sparked the Great Lake Region to address the issue of nutrient runoff with the Ohio Lake Erie Phosphorus Task Force (2013) recommending a 37% reduction from the 2007-2012 spring averages and a 41% reduction in dissolved reactive phosphorus for the Maumee Basin which is the principal Ohio watershed draining into Lake Erie. With region being poised to invest heavily in order to meet these targets the need for an understanding of the spatial and temporal dynamics of NPS pollutants is critical for the efficient use of these resources.

Pollutant loading into surface waters varies in space and in time. This variability originates from the need for hydrologic transport mechanisms such as surface runoff (Dunne and Black, 1970; Horton, 1940), subsurface and tile drainage (Freeze, 1972) and erosion (Lyon *et al*., 2006) to intersect and mobilize a pollutant sources from the land surface to surface waters (Heathwaite *et al.*, 2000). This variability creates a disproportionate contribution of pollutants to surface waters, driven by differences in source and transport mechanisms across a watershed and is central to the concept of Critical Source Areas (CSAs).

Several studies focusing on CSAs have observed that a relatively small fraction of a watershed can generate a disproportionate amount of pollutant load (Pionke et al., 2000; Gburek *et al.*, 2000; Yang and Weersink, 2004). A study of Oklahoma watersheds, reported that just 5% of the land area yielded 50% of the sediment load and 34% of the P load (White et al., 2009). Similarly in a Vermont river basin, about 74% of the annual P load was estimated to come from just 10% of the land area (Winchell *et al*., 2011). In the Upper Scioto Watershed of Ohio Xie (2014) calculated that 50% of the phosphorus came from 32% of the land. The identification of CSAs in a watershed offers an opportunity to prioritize and tailor conservation practices that will better protect water quality and reduce costs and transform a purely NPS problem into a quasi-point source problem.

Often neglected in these studies is the temporal nature of hydrologic transport mechanisms, primarily driven by seasonal variability in precipitation and land cover. For example, precipitation falling on fields prior to planting in early April would result in very different regions of the watershed contributing pollutant loads to if the same precipitation event where to occur just after fertilizer application or late in the growing season. As the land surface characteristics change through the course of a growing season, so too will the location of the CSAs. From a monitoring and management perspective, an understanding of these temporal dynamics may improve targeted short term mitigation efforts.

On a longer timeframe, global climate change is predicted to increase weather variability, altering the distribution of environmental variables that influence agricultural and hydrological processes (Mendelsohn *et al.,* 1994). Longer growing seasons, more frequent extreme precipitation events, and increasing periods between events are among the numerous ways in which environmental drivers of land management are anticipated to change under an altered climate. Under an uncertain climate future, it is critical to understand how all linked processes—biogeochemical, hydrological, and land management—could accelerate and exacerbate the impacts of NPS pollution and alter the spatial and temporal dynamics of NPS pollution and CSAs. Changes to hydrological flow paths could impact the transport of NPS pollutants, while changes to soil moisture dynamics and soil temperature could affect the biogeochemical cycling of pollutants within the soil column. It is critical to quantify how the fate and transport of pollutants are impacted by the current land use and how local impacts of global climate change manifest and combine with other pressures. A sound understanding of the interactions between NPS pollution and global climate change will allow for better long term policy-making and management of agricultural lands and water resources in central Ohio.

This study used the Upper Big Walnut Creek (UBWC) watershed in central Ohio as a case study. The UBWC watershed is comprised of approximately 60% agricultural cropland and provides drinking water for approximately 600,000 residents of Columbus and surrounding communities. It was identified as a priority impaired watershed by the Ohio Environmental Protection Agency (EPA) in 1998, 2000, and 2003 (King *et al*., 2008). The majority of headwater streams in the watershed are impaired by nutrient enrichment, pathogens, and habitat degradation stemming from agricultural management practices (King *et al*., 2008). In addition, current pressures from agricultural intensification (increases in tile drainage and fertilizer application) and rapid urbanization within the UBWC Watershed may be exacerbated by local impacts of global climate change (Melillo *et al*., 2014).

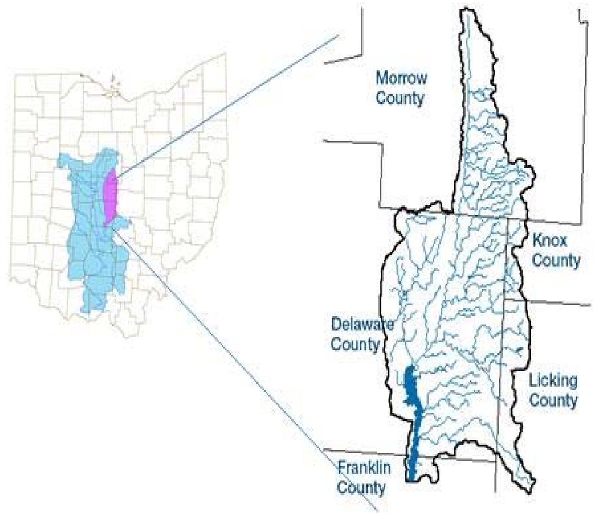

Figure 1: Left - Scioto River Watershed (blue) and Big Walnut Creek Watershed (magenta); Right - Upper Big Walnut Creek

2 Objectives

The overarching objective of this research is to:

Identify the hydrologic and land surface characteristics that influence the spatial and temporal dynamics of NPS pollutants (nitrogen and phosphorus, in particular).

To achieve this objective the following activities will be undertaken:

(i) Assessment of a suite of existing indexing methods (eg topographic Index, phosphorous index) that can be used to identify CSAs in the UBWC watershed and to compare these results with pollutant load quantifications observed by the USDA.

(ii) Application of the Soil and Water Analysis Tool (SWAT) model to explicitly represent the linked interactions between biogeochemical, hydrological, and land management to quantify pollutant load for comparison of SWAT derived CSAs to those identified by the methods listed above.

3 Methods

3.1 Site Description: The Upper Big Walnut Creek (UBWC) Watershed

The UBWC watershed is an 11-digit watershed (HUC 05060001-130) located in central Ohio (latitudes 40°06’00 to 40°32’30”, longitudes 82°56’00” to 82°42’00”, Figure 1). It covers 492 km^2 (190 mi^2) and contains 686 km of perennial and intermittent streams that drain into Hoover Reservoir.

The UBWC watershed is one of the twelve benchmark watersheds in the United States that are being evaluated as part of the Agricultural Research Service’s (ARS) component of the Conservation Effects Assessment Project (CEAP) (Mausbach and Dedrick, 2004). UBWC is unique among ARS watersheds because it involves the combined evaluation of the hydrological, chemical, and ecological responses of channelized and unchannelized headwater streams to conservation practices.

Agricultural cropland comprises the largest land use classification in the watershed (approximately 60%). The primary agricultural crops are corn, soybeans, and wheat. Management practices include conservation tillage, fertilization, and herbicide applications. A substantial portion of the watershed used for agricultural production is systematically tile-drained. In addition to crop production, approximately 15% of the watershed (in the southwestern portion) is transitioning from agriculture to urban land use that is composed of single- and multi-unit dwellings, parks, and golf courses. Additionally, soils in the watershed are clayey, poorly drained.

Current agricultural intensification (increase in tile drainage and fertilizer application) and rapid urbanization within the UBWC Watershed provides a unique opportunity to examine the impact of different land uses, agricultural management and conservation practices on NPS pollution by leveraging a paired watershed study conducted by the USDA ARS (King *et al*., 2008).

3.2 Existing Index Methodologies

In order to assess the utility of our propose model based identification of CSAs an assessment of an the existing methods used for the identification of CSAs will be undertaken for the UBWC watershed. The following methods were examined:

- Topographic index (TI) approach (Beven and Kirkby, 1979), and
- Curve number (CN) model (Arnold *et al.*, 1998),

The results obtained from these methods will be validated using pollutant loads observed by the USDA (data from 8 sub-basins from 2006-present). These methods rely on observed correlation between land surface characteristics and the generation of runoff to implicitly identify CSAs. Their advantage is in their simplicity, but consequently they lack the ability to predict and quantify runoff and nutrient generation processes and pollutant load (Srinivasan *et al.*, 2005; White *et al.*, 2009). Precipitation characteristics, which are projected to alter significantly under a changing climate, are noticeable absent in any of the methods listed above.

3.3 SWAT and Identification of CSAs

SWAT is a process-based, river basin-scale model developed by the USDA Agricultural Research Service (Arnold *et al.*, 1998) to predict the short- and long-term impacts of land management practices on aquatic water, sediment, and nutrient fluxes in large complex watersheds with varying soils, topography, land use, and management conditions. It explicitly represents the key hydrological, biophysical, and biogeochemical processes in both terrestrial and aquatic ecosystems, as well as management practices. A wide array of nonlinear biological and environmental processes are captured across all of the major model components, including crop and other vegetation growth patterns, temporal patterns in precipitation inputs and resulting interception by plant canopies, subsequent estimation of surface runoff, evapotranspiration, other hydrologic components, and estimation of soil erosion and transport of sediment, nutrients, and other pollutants.

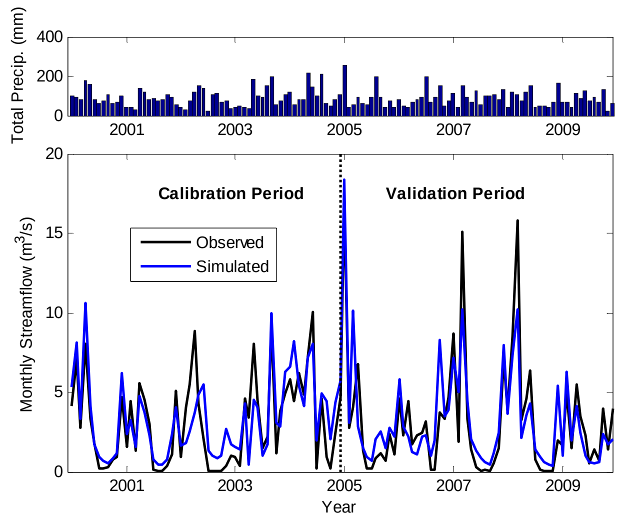

Figure 2: Flow calibrated SWAT Model for UBWC Watershed. Comparison of mean monthly observed streamflow to simulated streamflow at USGS Gauge 03228300 (Sunbury, OH) for the model calibration and validation periods of 2000-

Numerous studies have reported successful applications of SWAT for reproducing observed hydrologic and/or pollutant loads across a wide range of watershed scales and environmental conditions, as well as its applications, in assessing impacts of conservation practices, land use, water management, and other scenarios (Gassman *et al.*, 2007). SWAT has been widely evaluated for simulating streamflow and in-stream water quality constituents, such as water temperature, sediment fluxes, dissolved oxygen, nitrogen, and phosphorus (e.g. Daloglu *et al.*, 2012; Ficklin *et al.*, 2012; Jang *et al.*, 2012; White *et al.*, 2012; Palao *et al.*, 2013), and applied to assess effects of cropland management practices, wetland restoration, and reservoir construction and operations on water quality (e.g. Garg *et al.*, 2012; Michalak *et al.*, 2012; Comin *et al.*, 2013; Zhu *et al.*, 2012).

As a semi-distributed model, SWAT requires a suite of spatially-explicit inputs, primarily including topography, soil properties, land use/cover, weather/climate data, and management practices. For our modeling analyses, a 10m digital elevation models will be used for topography; soil parameters will be retrieved from USDA's SSURGO database; annual land-use maps at a 56-m resolution, with crop-specific land classes, will be obtained from NASA's Crop Data Layer; weather data, such as precipitation and temperature, will be collected and extrapolated from NOAA's weather stations (or national re-analysis data) for historical periods and from downscaled GCM projections for future scenarios; management data, such as fertilization, tillage, planting, harvesting, and other field operations will be derived from survey data obtained directly from operator interviews from the National Resources Inventory (NRI) CEAP Cropland Survey.

This study will used the latest release of SWAT, SWAT2012, which includes a synthetic weather generator and improvements to simulation of nutrient cycling (nitrogen and phosphorus). We will focus on the UBWC Watershed in order to take advantage of the USDA datasets available at the site and the existing modeling efforts the Sivandran Lab has in the watershed.

3.3.1 Chronic and Acute CSAs

It is critical also to make a distinction between chronic or persistent sources of NPS pollutants which may result from the slow release of nutrients through leaching through the soil column into tile drainage, and the more acute or episodic NPS pollutant fluxes which may arise from fields susceptible to the generation of episodic surface flows from high intensity precipitation events. These two sources are of equal importance to the total load accumulating in receiving water bodies, even though at present the use of the Total Maximum Daily Loads (TMDL) thresholds by the EPA tends to neglect these chronic sources. By explicitly making this distinction, operational land management decisions can be made to focus conservation efforts to address local watershed issues.

To identify CSAs, we will utilize a calibrated and validated SWAT model for the UBWC watershed flow (Figure 2). SWAT model simulation output will be analyzed at the hydrological response unit (HRU) level. Monthly aggregated NPS pollutant loads will be synthesized to identify chronic CSAs subsetted for each pollutant, following an approach similar to Niraual *et al.* (2013). HRU-level yields will be ranked in terms of yields from the highest to lowest and then functionally related to the land surface characteristics and management practices at each HRU.

In order to capture the temporally variable component of NPS-pollutants, high NPS pollutant yielding events will be analyzed at the HRU-level to determine the location of acute CSAs. These events will be analyzed to determine what hydrologic or land surface conditions resulted in the episodic events occurring. Focus will be given to antecedent watershed conditions, particularly soil moisture conditions prior to the event, vegetation phenological state, and the precipitation event characteristics (intensity, duration and interstorm period). By combining relationships between acute CSAs and antecedent conditions with relationships linking land surface characteristics and land management practices with chronic CSAs a real time spatial risk index can be created.

4 Research Outcomes

4.1 Critical Source Areas

SWAT simulations of watershed were conducted and the subbasins (depicted in Figure 4-2) were ranked in terms of their load contributions. The top sextile, which aggregates the contribution of the top sixth of subbasins, was responsible for contributing 52% of the total nitrogen and 55% total phosphorous (Figure 4-2).

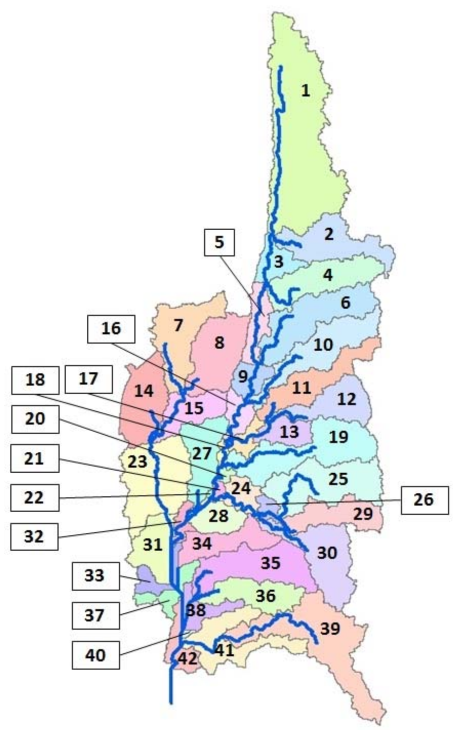

Figure 4-1. Subbasins in UBWC.

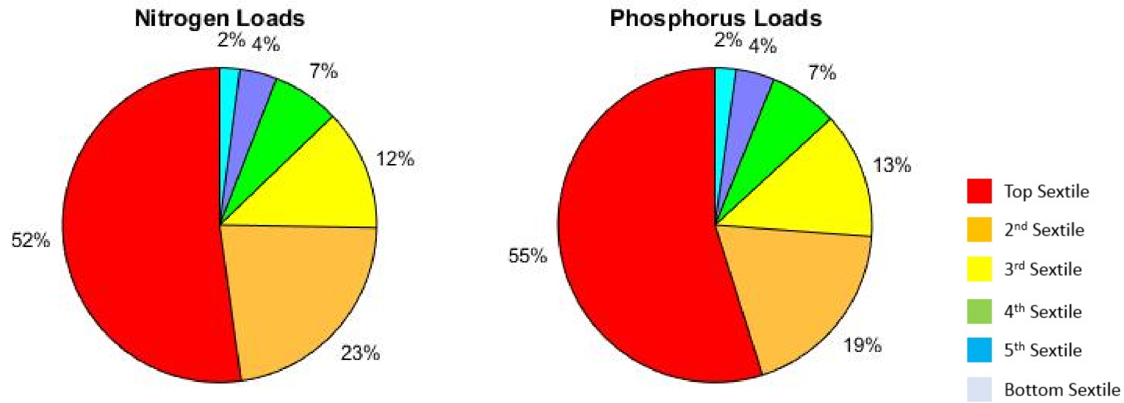

Figure 4-2. Percentage of Load for Each Sextile of Subbasins in UBWC

4.2 Critical Source Times

Rather than focusing exclusively on area, variation in loadings over time were examined to determine if they could possibly be a source of this variation. Over the 60-months of the study, the Top Sextile would be represented by the 10 individual months that had the highest loads (months Feb [50], Dec [24], Feb [14], Mar [3], Feb [26], Mar [15], Jul [55], Oct [58], Apr [52], and Oct [10]). This sextile (Critical Source Times) represents 54% total nitrogen and 58% total phosphorous of the nutrient load (Figure 4-3). Three months, February (50, 14, and 26), March (3 and 15), and October (58 and 10) account for seven of the top ten contributing months.

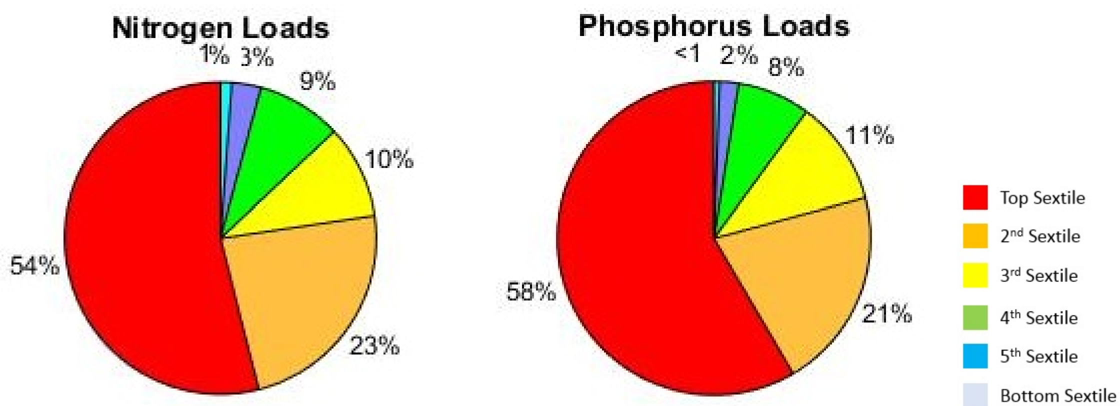

Figure 4-3. Percentage of Load for Each Sextile of Time (Month) in UBWC

4.3 Critical Source Junctures

When examining the spatial-temporal interactions (subbasins and time), the top sextile represents 62% (Nitrogen) and 65% (Phosphorus) of the nutrient load, and an even higher percentage than CSAs or CSTs individually. Therefore, it can be hypothesized that addressing the CSJs will have a greater impact than CSAs independently.

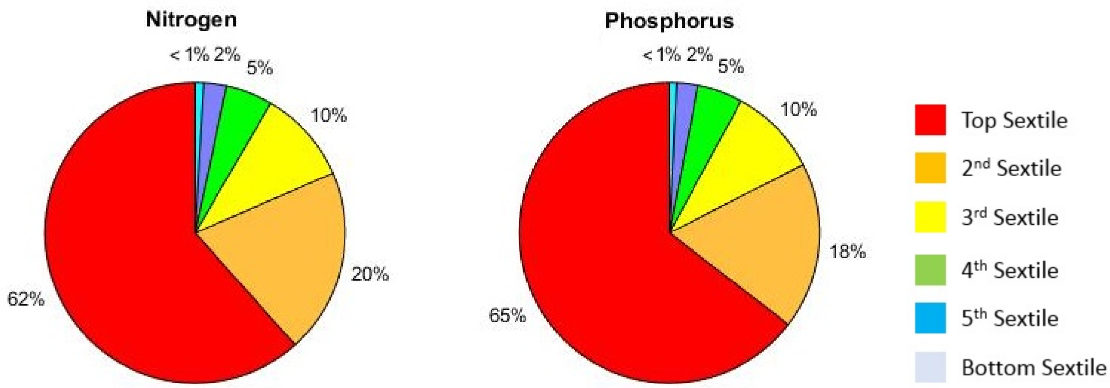

Figure 4-4. Percentage of Load for CSJ sextiles.

4.4 Discussion and Conclusions

Both time (Critical Source Times) and space (Critical Source Areas) can be used to determine contributors for the highest nutrient loads. However, by examining both time and space, a greater percentage of Nitrogen and Phosphorus loadings can be explained. Implementing BMPs that address both spatial and temporal issues could be a more efficient method of addressing nutrient pollution.

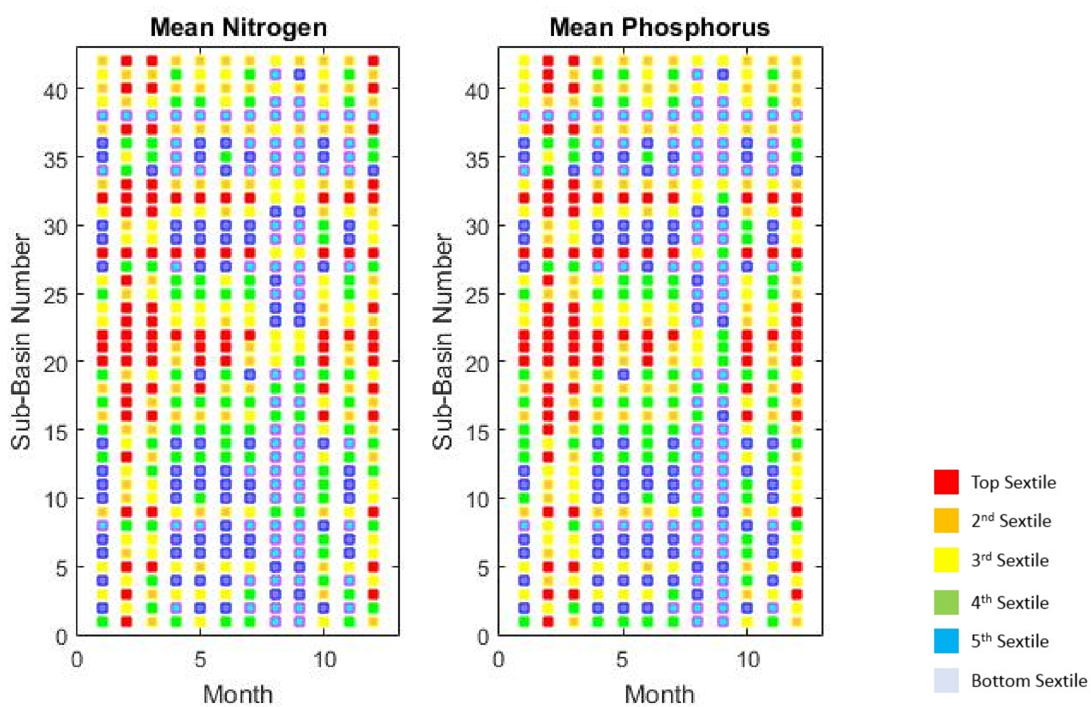

Figure 4-5. Monthly Mean Nitrogen and Phosphorus Load among 42 Subbasins and 12 Months in UBWC. Vertical features indication times of year of interest whereas horizontal features indicate areas within the catchment that consistently contribute to NPS.

Figure 4-5 indicates the interaction of both time and space. Vertical features on these plots indicate consistent spatial response at a given time. February, March, October, November and December all show stronger contributions across all subbasins. Horizontal features indicate a givens subbasins contribution to NPS across time. Subbasins 20,21,22 and 32 all show high levels of contribution regardless of time.

This form of temporal and spatial analysis could prove useful in determining the timing of intervention, the duration of that intervention as well as which subbasins would result in the greatest overall reduction in contaminate load. With mitigation resources limited, the ability to prioritize regions where long term conservation measures would greatly improve the ability to reduce the overall transport of nutrient load into Hoover Reservoir.

5 Reference list

Arnold JG, Allen PM, Bernhardt G, 1993. A comprehensive surface groundwater flow model. *J Hydrol*, *142*(1-4):47-69.

Beven KJ, and Kirkby MJ, 1979. A physically based variable contributing area model of basin hydrology. *Hydrol. Sci. Bull*. 24(1):43-69.

Browning D, 2014. Modeling non-point source pollution in surface water under non-stationary climates and land uses. Masters Thesis. Department of Civil Environmental and Geodetic Engineering. The Ohio State University.

Chen J, Brissette FP, and Leconte R, 2011, Uncertainty of downscaling method in quantifying the impact of climate change on hydrology, *Journal of Hydrology*, *401*(3–4), 190-202.

Comin FA, Sorando R, Darwiche-Criado N, García M, and Masip A, 2013. A protocol to prioritize wetland restoration and creation for water quality improvement in agricultural watersheds. *Ecological Engineering*.

Daloglu I, Cho KH, and Scavia D, 2012. Evaluating causes of trends in long-term dissolved reactive phosphorus loads to Lake Erie. *Environmental Science & Technology,* 46(19), 10660-10666.

Dunne T. and Black RD, 1970. Partial area contributions to storm runoff in a small New England watershed. *Water Resour. Res*. 6:1296-1311.

Ficklin DL, Luo Y, Stewart IT, and Maurer EP, 2012. Development and application of a hydroclimatological stream temperature model within the Soil and Water Assessment Tool. *Water Resources Research*, 48(1).

Freeze RA. 1972. Role of subsurface flow in generating surface runoff: 1. Base flow contributions to channel flow. *Water Resources Research*, 8(3), 609-623.

Fowler HJ, Blenkinsop S, and Tebaldi C, 2007, Linking climate change modelling to impacts studies: recent advances in downscaling techniques for hydrological modelling, *International Journal of Climatology*, *27*(12), 1547-1578.

Garg KK, Bharati L, Gaur A, George B, Acharya S, Jella K, and Narasimhan, B. 2012. Spatial mapping of agricultural water productivity using the SWAT model in Upper Bhima Catchment, India. *Irrigation and Drainage*,61(1), 60-79.

Gassman PW, Reyes MR, Green CH, Arnold JG, (2007). The Soil and Water Assessment Tool: historical development, applications, and future research directions. *T ASABE, 50*(4): 1211-50.

Gburek, WJ, Sharpley AN, and Folmar GJ, 2000. Critical areas of phosphorus export from agricultural watersheds. In A. N. Sharpley (ed.), Agriculture and Phosphorus Management (pp. 83- 104). Boca Raton, FL: Lewis Publishers.

Heathwaite AL, Sharpley AN, and Gburek WJ. 2000. Integrating phosphorus and nitrogen management at catchment scales. *J. Environ. Qual*. 29:158-166.

Horton RE, 1941. An approach toward a physical interpretation of infiltration capacity. *Soil science society of America journal*, 5(C), 399-417.

Jang JH, Jung KW, and Yoon CG. 2012. Modification of SWAT model for simulation of organic matter in Korean watersheds. *Water Science & Technology*, 66(11), 2355-2362.

King KW, Smiley PC, Baker BJ, and Fausey NR, (2008). Validation of paired watersheds for assessing conservation practices in the Upper Big Walnut Creek watershed, *Ohio. journal of soil and water conservation*, *63*(6), 380-395.

Liang X-Z, Pan J, Zhu J, Kunkel KE, Wang XLJ, and Dai A, (2006), Regional climate model downscaling of the U.S. summer climate and future change, *Journal of Geophysical Research: Atmospheres*, *111*(D10), D10108.

Lyon SW, McHale MR, Walter MT, and Steenhuis TS. 2006. The Impact Of Runoff Generation Mechanisms On The Location Of Critical Source Areas. *Journal of the American Water Resources Association*, 42: 793–804. doi: 10.1111/j.1752-1688.2006.tb04493.x

Mailhot A, Duchesne S, Caya D, Talbot G, 2007. Assessment of future change in intensity duration frequency (IDF) curves for Southern Quebec using the Canadian Regional Climate Model (CRCM). *J. Hydrol. 347*, 197-210.

Mausbach MJ, and Dedrick AR, 2004. The length we go measuring environmental benefits of conservation practices. *Journal of Soil and Water Conservation*, 59(5), 96A-103A.

Mearns LO, Gutowski W, Jones R, Leung R, McGinnis S, Nunes A, and Qian Y, 2009, A Regional Climate Change Assessment Program for North America, *Eos, Transactions American Geophysical Union*, *90*(36), 311-311.

Melillo, Jerry M., Terese (T.C.) Richmond, and Gary W. Yohe, Eds., 2014: *Climate Change Impacts in the United States: The Third National Climate Assessment*. U.S. Global Change Research Program, 841 pp. doi:10.7930/J0Z31WJ2.

Mendelsohn R, Nordhaus WD, and Shaw D, (1994). The impact of global warming on agriculture: a Ricardian analysis. *The American Economic Review*, 753-771.

Michalak AM, Anderson EJ., Beletsky D, Boland S, Bosch NS, Bridgeman TB, and Zagorski, MA. 2013. Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proceedings of the National Academy of Sciences*, 110(16), 6448-6452.

Niraula R, Kalin L, Srivastava P, and Anderson CJ. 2013. Identifying critical source areas of nonpoint source pollution with SWAT and GWLF. *Ecological Modelling*, 268, 123-133.

Ohio Lake Erie Phosphorus Task Force 2013 Ohio Lake Erie Phosphorus Task Force II Report-Final Report. Joint publication of the State of Ohio Departments of Agriculture, Natural Resources, Environmental Protection Agency and the Lake Erie Commission.

Olsson J, Berggren K, Oloffsson M, Viklander M, 2009. Applying climate model precipitation scenarios for urban hydrological assessment: A case study in Kalmar City, Sweden. *Atmos. Res.*, *9*, 364-375.

Olsson J, Uvo CB, and Jinno K, 2001, Statistical atmospheric downscaling of short-term extreme rainfall by neural networks, *Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere*, *26*(9), 695-700.

Ouyang W, Skidmore AK, Hao F, and Wang T. 2010. Soil erosion dynamics response to landscape pattern. *Science of the total environment*, 408(6), 1358-1366.

Palao LKM, Dorado MM., Anit KPA, and Lasco RD. 2013. Using the Soil and Water Assessment Tool (SWAT) to Assess Material Transfer in the Layawan Watershed, Mindanao, Philippines and

Its Implications on Payment for Ecosystem Services. *Journal of Sustainable Development*, 6(6), p73

Pionke HB, Gburek WJ, and Sharpley AN. 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. *Ecol. Eng*. 14: 325–335.

Sheffield J, et al. (2013), North American Climate in CMIP5 Experiments. Part II: Evaluation of Historical Simulations of Intra-Seasonal to Decadal Variability, *Journal of Climate*.

Sivertun A, and Prange L, 2003. Nonpoint source critical area analysis in the Gisselo watershed using GIS. Environ. *Modelling Software* 18:887-898.

Srinivasan MS, Gerard-Marchant P, Veith TL, Gburek WJ, and Steenhuis TS. 2005. Watershed scale modeling of critical source areas of runoff generation and phosphorus transport: agricultural hydrology and water quality. *Journal of the American Water Resources Association* 41 : 361–375.

Srinivasan MS, and McDowell RW. 2007. Hydrological approaches to the delineation of critical-source areas of runoff. *New Zealand J. Agri. Res*. 50:249-265.

Taylor KE, Stouffer RJ, and Meehl GA, 2012, An Overview of CMIP5 and the Experiment Design, *Bulletin of the American Meteorological Society*, *93*(4), 485-498.

Teng J, Chiew FHS, Timbal B, Wang Y, Vaze J, Wanga B, 2012. Assessment of an analogue downscaling method for modelling climate change impacts on runoff. *J. Hydrol*., 472-473, 111-125.

USEPA. Better Assessment Science Integrating Point and Nonpoint Sources (BASINS): Basic Information. United States Environmental Protection Agency (USEPA). http://www.epa.gov/waterscience/basins/basinsv3.htm; 2008; [accessed 23 November 2009].

Wang Y and G Sivandran G, 2014. Hourly Precipitation Downscaling With Empirical Statistical Methods: Case Study near the US Eastern Coast, Abstract H41A-0788 accepted to 2014 Fall Meeting, AGU, San Francisco, CA.

Wang Y. 2014. Assessment of the Uncertainty of Statistical Downscaling Methodologies. Masters Thesis. Department of Civil Environmental and Geodetic Engineering. The Ohio State University.

White MJ, Storm DE, Busteed PR, Stoodley SH, Phillips SJ. 2009. Evaluating nonpoint source critical source area contributions at the watershed scale. *J. Environ. Qual* 38: 1654-1663.

White MJ, Harmel RD, Arnold JG, and Williams JR. 2012. SWAT check: A screening tool to assist users in the identification of potential model application problems. *Journal of Environmental Quality*.

Willems P, Vrac M, 2010. Statistical precipitation downscaling for smallscale hydrological impact investigations of climate change. *J. Hydrol. 402,* 193–205.

Winchell MD. Meals W, Folle S, Moore J, Braun D, DeLeo C, Budreski K, and Schiff R. 2011. Identification of Critical Source Areas of Phosphorus within the Vermont Sector of the Missisquoi Bay Basin. Final Report to The Lake Champlain Basin Program. Montpelier, VT: Stone Environmental, Inc. Online: http://www .lcbp.org/ijc.htm [accessed May 18, 2014]

Xie, Yina 2014. Watershed Modeling, Farm Tenancy, and Adoption of Conservation Measures to Facilitate Water Quality Trading in the Upper Scioto Watershed, Ohio. Ph.D. Dissertation. Environmental Sciences Graduate Program. The Ohio State University.

Yang W, and Weersink A. 2004. Cost-effective Targeting of Riparian Buffers. *Canadian Journal of Agricultural Economics/Revue canadienne d'agroeconomie*, 52(1), 17-34.

Zhu Z, Broersma K, and Mazumder A. 2012. Impacts of Land Use, Fertilizer and Manure Application on the Stream Nutrient Loadings in the Salmon River Watershed, South-Central British Columbia, Canada. *Journal of Environmental Protection*, 3(8), 809-822.

Developing Integrated Assessments of Water and Energy Ohio

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Final Report: Developing Integrated Assessments of Water and Energy in Ohio

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1 Summary

Water and energy are intrinsically linked in modern energy systems, where demand for one results in a demand for the other. These demands will depend on the weather (e.g., air-conditioning), the supply of water (e.g., precipitation), and the type of cooling technology used in a power plant. The problem is that our understanding of these linkages is far from complete. The overall objective of this project is to improve our understanding of how electricity demand, and the demand for water by thermoelectric power plants that supply electricity, depend on weather in the short and long term. The research comprises two major parts. The first part comprises of constructing a statistical model that relates regional electricity demand to weather variables (temperature, relative humidity, wind speed) in a parametric form. The model was based and tested on more than ten years of hourly data in two transmission zones in the Pennsylvania-New Jersey-Maryland (PJM) Interconnection. One novel feature of the model is that the statistical technique employed enables empirical estimation of the base temperatures in cooling-and heating-degree hours, which are two traditional metrics used in estimating electricity demands associated with cooling and heating. The result indicates that a piecewise linear function is approximately valid for describing the relationship between electricity demand and temperature, with a relatively flat region over the medium temperatures that define a “comfort zone” between heating and cooling. The result also gives quantitative measures of the effects of past temperatures, relative humidity, and wind speed on electricity demand. The second part comprises of connecting the electricity demand to the water demand of power plants. We have acquired monthly facility-level water demand data from the Energy Information Administration (EIA) database and the Ohio Department of Natural Resources (ODNR). Since the data are of low quality, they are currently being cross-examined with a variety of resources (literature values, google earth, previously corrected datasets, etc.) for quality control and correction, and as yet partially analyzed. Conversations are ongoing with AEP (initiated in the second quarter of this project) to acquire daily facility-level water data.

2 Problem and research objectives

2.1 On the relationship between electricity demand and weather variables

Accurate electricity load forecasts have a number of applications. Forecasts on the timelines of days to several years are important to inform electricity dispatch scheduling, capacity expansion by utility companies, and state-level policy-making (Beccali et al., 2008; Chandramowli and Felder, 2014; Dordonnat et al., 2008). On the timeline of decades, models of electricity load can help understand the potential impacts of, and thus adaptation needs to, climate change (Auffhammer and Aroonruengsawat, 2011; Braun et al., 2014; Ruth and Lin, 2006). One way to characterize how electricity demand depends on temperature is to use the concept of degree days (the counterparts on hourly scale are called degree hours). Heating Degree Days (HDDs) are calculated as the number of degrees that a day is below a reference temperature, and Cooling Degree Days (CDDs) are calculated as the number of degrees that a day is above a reference temperature. Together, these two metrics represent the relationship between energy demand and temperature as a V-shaped distribution about the reference temperature. This HDD/CDD approach and has frequently been used in decadal projections of energy demand and in load forecast studies (Amato et al., 2005; Mirasgedis et al., 2006; Pardo et al., 2002; Ruth and Lin, 2006; Scapin et al., 2015; Shorr et al., 2009; Vu et al., 2014).

One difficulty in using HDDs and CDDs is that the base temperatures are difficult to estimate. Studies have used established reference temperatures (18.3 ^{o}C), estimated the reference temperature by visual examining the data, or determined the reference temperature by maximizing some criterion for model performance (Amato et al., 2005; Mirasgedis et al., 2006; Scapin et al., 2015; Shorr et al., 2009; Vu et al., 2014). Most of these types of studies assumed that the reference temperature for HDDs is the same as the reference temperature for CDDs (Amato et al., 2005; Mirasgedis et al., 2006; Shorr et al., 2009; Vu et al., 2014). It has been noted that geographical variations can exist in base temperature (Brown et al., 2016), and that the uncertainty in base temperature should be considered during the model estimation process (Woods and Fuller, 2014). Also, past studies that used smooth transition regression models, which replace the V-shaped transition from HDDs to CDDs by parameterized logistic or exponential functions, suggested that a “comfort zone” exists at intermediate temperatures where the electricity demand is less sensitive to changes in temperature (Bessec and Fouquau, 2008; Moral-Carcedo and Vicéns-Otero, 2005). This means that different base temperatures for HDDs and CDDs may be needed.

While temperature is generally considered the most important weather determinant of electricity load, lagged temperatures, humidity, wind speed, solar radiation, precipitation, and/or air pressure can have secondary influences. The decadal-scale studies usually only used temperature to capture the major effect of climate change, in addition to socioeconomic factors (e.g. electricity price, population) (Amato et al., 2005; Auffhammer and Aroonruengsawat, 2011; Franco and Sanstad, 2007; Ruth and Lin, 2006). Studies focusing on shorter time scales generally included some of the secondary factors to increase the accuracy of the predicted electricity demand (Beccali et al., 2008; Dordonnat et al., 2008; Mirasgedis et al., 2006; Psiloglou et al., 2009; Scapin et al., 2015; Vu et al., 2014), but such studies tended to focus more on the skill of prediction than understanding of the functional form.

Therefore, one objective of the study is to empirically determine the base temperatures of heating- and cooling-degree hours (HDHs, CDHs) using a segmented regression technique (Muggeo, 2008). This method allows different base temperatures for HDHs and CDHs, and is robust because the uncertainty in base temperatures is considered during the model estimation process (Muggeo, 2008). Compared to the smooth transition regression models, the piecewise linear assumption in the degree days concept is not modified in segmented regression (Muggeo, 2008). The second objective is to determine the effects of temperature, past temperatures, relative humidity, and wind speed on electricity load, and their relative importance. To achieve these, a model was first developed using only temperature, and then the model was extended to include past temperatures, relative humidity, and wind speed as additional predictors of electricity load.

2.2 The water demand of thermoelectric power plants

Thermoelectric power plants traditionally have significant water demand. In 2010, the water withdrawal by thermoelectric power plants accounted for 45% of the national total water withdrawal, though only a small portion of the withdrawal is consumptive (e.g. lost to evapotranspiration) (Maupin et al., 2014). Water consumption by the thermoelectric sector is small compared to agriculture, but is still larger than all other industrial consumptions combined, and is expected to grow by 40~60% by the 2030s (The Great Lakes Commission, 2011). This high water demand means that the thermoelectric sector is vulnerable to water scarcity caused by simultaneous demand from multiple end-use categories (e.g. irrigation, public supply, the aquatic ecosystem), and by weather variability and climate change. Instances where the development of new power plants was hampered by lack of cooling water are increasing (Scott and Huang, 2007). In a few extreme cases, power plants have curtailed electricity production or shut down due to high water temperature or low streamflow (see Förster & Lilliestam 2010).

The water intensity of thermoelectric power plants is generally characterized by water withdrawal and consumption factors, which, respectively, are the amount of water withdrawal and consumption per unit net electricity generation. Annual water withdrawal and consumption factors for power plants in the US have been synthesized or estimated in a number of past studies, and are found to depend the power generation technology, cooling system, and the existence of additional features such as carbon capture and storage (Diehl and Harris, 2014; Macknick et al., 2011; Strzepek et al., 2012). Within each of the categories, the water withdrawal and consumption factors are still highly variable, and unexplained discrepancy exists between the factors calculated from actual water use data and the factors estimated by literature (Averyt et al., 2013; Macknick et al., 2011). Our inadequate understanding of the water withdrawal and consumption factors means that current national estimation of the water use by thermoelectric power plants is highly uncertain.

Therefore, the objective of the study is to improve our understanding of the water use by thermoelectric power plants by examining the influence of weather variables. A statistical model will be constructed that relates weather variables (e.g. temperature, atmospheric pressure, humidity) to the cooling water withdrawal and consumption factors, while controlling for the known time-invariant factors such as power plant configuration and cooling technology.

3 The response of hourly electricity load to meteorological variables in the PJM Interconnection

3.1 Methodology

The temperature-only and the extended electricity demand models were estimated using the time series routine (segmented.Arima) in the R-package "segmented" (version 0.5.1.1) (Muggeo, 2008) under the R 3.2.0 environment. The basic formulas are:

$$E_{t,j} = a_j + \sum_{m=1}^{11} b_{j,m} M_m + \sum_{w=1}^{w=6} c_{j,w} W_w + \boldsymbol{X_{t,j}} \boldsymbol{\beta_j} + \zeta_{t,j}$$

$$\theta_p(B)\Theta_P(B^S)\zeta_{t,j} = w_q(B)W_Q(B^S)e_{t,j}$$

where $E_{t,j}$ is electricity load on day t and hour j; a_j is the intercept, interpreted as the base load on any Sunday in December; M_m, m=1,2,...,11, are the monthly dummies, equal to 1 for January through November, respectively, and 0 otherwise; $b_{j,m}$ are coefficients that describe the fixed monthly variations in base load; W_w, w=1,2,...,6, are the weekday dummies, equal to 1 for Monday through Saturday, respectively, and 0 otherwise; $c_{j,w}$ are coefficients that describe the fixed weekly cycles; $\boldsymbol{X_{t,j}}$ is the vector of meteorological variables (temperature, past temperatures, relative humidity, wind speed, interactions), and together with $\boldsymbol{\beta_j}$ describe the weather sensitive part of the load; $\zeta_{t,j}$ is the residual, and is assumed to follow a seasonal autoregressive moving average (S-ARMA) process, where $\theta_p(B)$, $\Theta_P(B^S)$, $w_q(B)$, and $W_Q(B^S)$ are polynomials of the backward shift operator B, p and q are the autoregressive and moving-average orders, P and Q are the seasonal autoregressive and moving-average orders, S is the seasonal cycle (here weekly), and $e_{t,j}$ is a white noise process with mean = 0 and some standard deviation.

For the temperature-only model, the weather-sensitive part is:

$$\begin{aligned} \boldsymbol{X}_{t,j}\boldsymbol{\beta}_j &= \beta_{j,1}TP_{t,j} + \beta_{j,2}\left(TP_{t,j} - TP_{0,h}\right) * I\left(TP_{t,j} > TP_{0,h}\right) \\ &+ \beta_{j,3}\left(TP_{t,j} - TP_{0,c}\right) * I\left(TP_{t,j} > TP_{0,c}\right) \end{aligned}$$

where $TP_{0,h}$ and $TP_{0,c}$ are the estimated breakpoints that correspond to base temperatures in the definition of HDDs and CDDs; $\beta_{j,1}$, $\beta_{j,2}$, and $\beta_{j,3}$ are regression coefficients.

For the extended model, the weather-sensitive part is:

$$\begin{aligned} \boldsymbol{X}_{t,j}\boldsymbol{\beta}_j &= \beta_{j,1}TP_{t,j'} + \beta_{j,2}\left(TP_{t,j'} - TP_{0,h}\right) * I\left(TP_{t,j'} > TP_{0,h}\right) \\ &+ \beta_{j,3}\left(TP_{t,j'} - TP_{0,c}\right) * I\left(TP_{t,j'} > TP_{0,c}\right) + \sum_{\Delta=1}^{3} g_{j,\Delta} TP_{t-\Delta,j'} \\ &+ \sum_{m=1}^{11}\sum_{\Delta=1}^{3} h_{j,m,\Delta}\left(M_m \times TP_{t-\Delta,j'}\right) + k_j RH_{t,j} + l_j\left(TP_{t,j'} \times RH_{t,j}\right) + n_j WD_{t,j} \end{aligned}$$

where the temperature part is the same as above, $TP_{t-\Delta,j'}$ is the temperature on day t-Δ and hour j', $RH_{t,j}$ is relative humidity on day t and hour j, $WD_{t,j}$ is wind speed, and $g_{j,\Delta}$, $h_{j,m,\Delta}$, k_j, l_j, and n_j are regression coefficients. This formula was determined after testing various types of interactions among the weather variables and monthly dummies, and was found to be a good trade-off between complexity and performance.

The two transmission zones investigated in the PJM Interconnection is displayed in Figure 1. The two transmission zones were selected because they sample the diversity in location and area of the transmission zones in the PJM Interconnection, and have relatively long records. The AEP zone has an area of about $1.34*10^5$ km^2 and a population of about $7.63*10^6$ in 2010; the PS zone about $3.35*10^3$ km^2 and a population of about $4.36*10^6$ (U.S. Cenus Bureau, 2010).

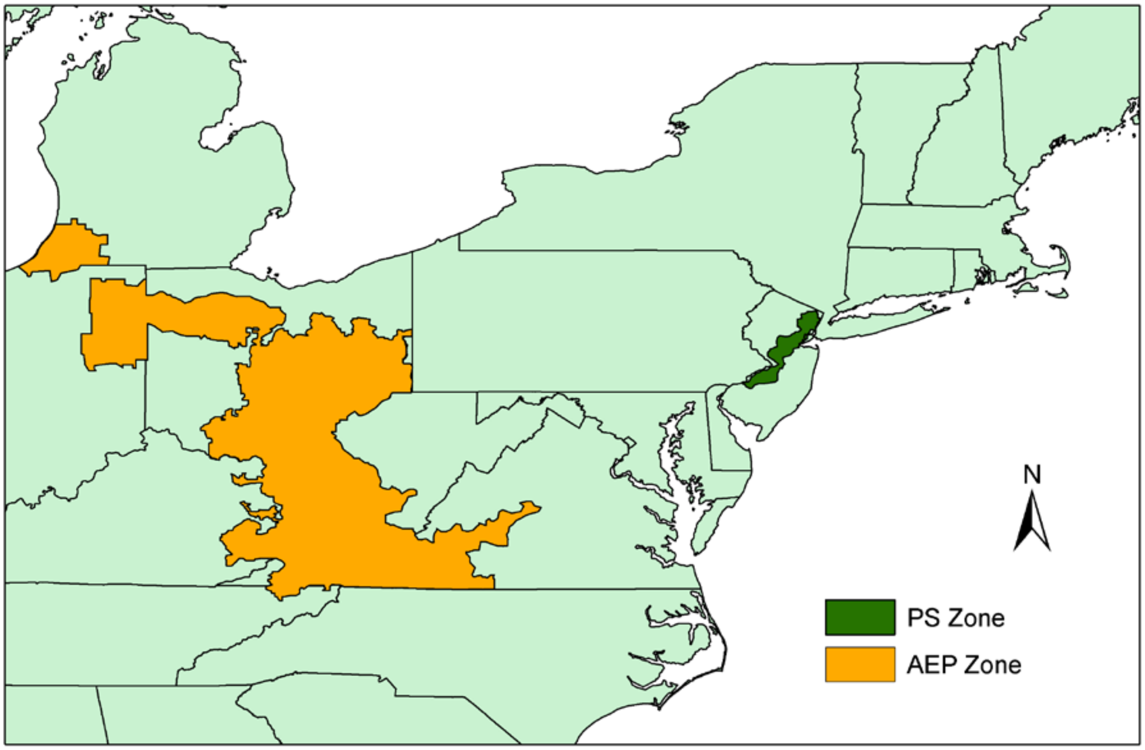


Figure 1. The transmission zones used in this study. Map adapted from the information on the PJM website (PJM, 2016). PS - Public Service Electric and Gas Company. AEP - American Electric Power Co., Inc.

3.2 Principal findings

Figure 2 shows the result of fitting the temperature-only model in hour 14 in the two transmission zones; the piecewise relationship is shown for a typical Sunday in December, a typical workday (Wednesday) in December, and a typical workday in July. The electricity response to temperature is characterized reasonably well by piecewise relationship. In the AEP zone, some high-load days on the heating arm were missed, which might be due to the fact that AEP zone spans a larger and more heterogeneous area than the PS zone. Electricity was used as the primary heating fuel in the southern part of the AEP zone, compared to natural gas in its northern part and the PS zone (EIA, 2015). This means that electricity demand could be more sensitive to temperature at the southern part of the AEP zone than the northern part, making the resulting data a mixture and therefore less easily captured by a single linear relationship

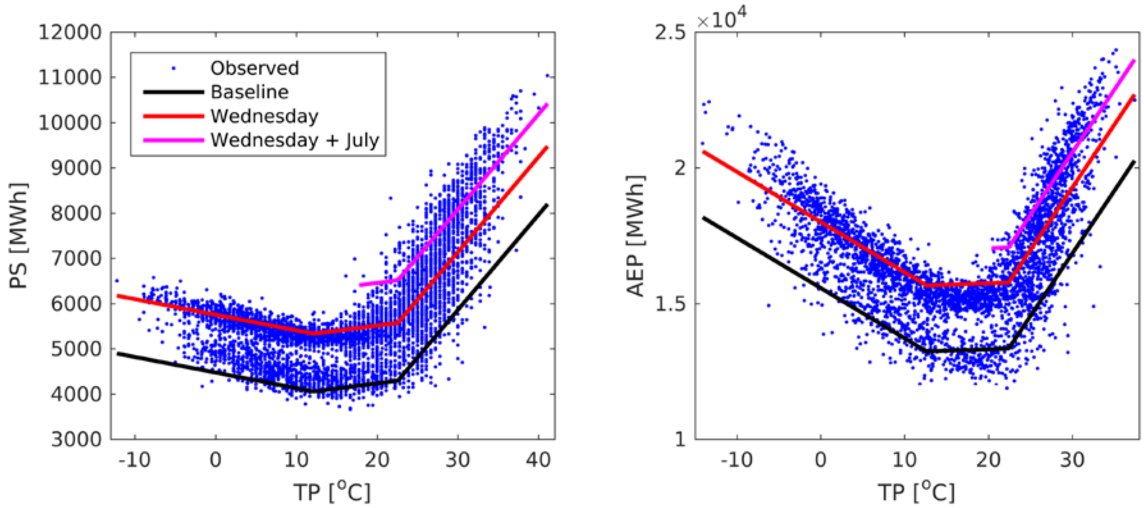


Figure 2. Observed relationship between temperature and electricity load at hour 14, and the fitted piecewise linear relationship for the baseline (i.e. December Sunday), Wednesday (December), and July Wednesday. The July relationship is only shown for July temperatures.

Figure 3 displays the estimated breakpoints ($TP_{0,h}, TP_{0,c}$) and slopes of the electricity response ($\beta_{j,1}, \beta_{j,1} + \beta_{j,2}, \beta_{j,1} + \beta_{j,2} + \beta_{j,3}$) of the temperature-only model and the extended model in the two transmission zones. In both transmission zones, the breakpoints were higher in working hours than in the night hours. In the PS zone, the diurnal range of the lower breakpoints was 10.2 ~ 15.8 °C, and of the higher breakpoints 17.8 ~ 23.2 °C. In the AEP zone, the diurnal ranges were 9.3 ~ 13.8 °C and 17.0 ~ 23.7 °C. Transition between the day and night hours occurred gradually. The higher breakpoints, corresponding to the base temperature for CDH, were significantly separated from the lower breakpoints, as indicated by the confidence intervals. This indicates the existence of a comfort zone. The slopes in the comfort zone (the middle line in the upper and lower left panels) were mostly statistically indistinguishable from 0 in the AEP zone. In the PS zone, the slope was positive, though much smaller than the slope of the cooling arm. These indicate that electricity load was relatively insensitive to temperature in the comfort zone.

Figure 3 also shows that the slopes of the heating arm were more negative in the extended model than in the temperature-only model; this was caused by the particular form of interaction between temperature and relative humidity ($TP_{t,j'} \times RH_{t,j}$). A more appropriate form of the effects of relative humidity has been identified and will be the subject of a follow-up study.

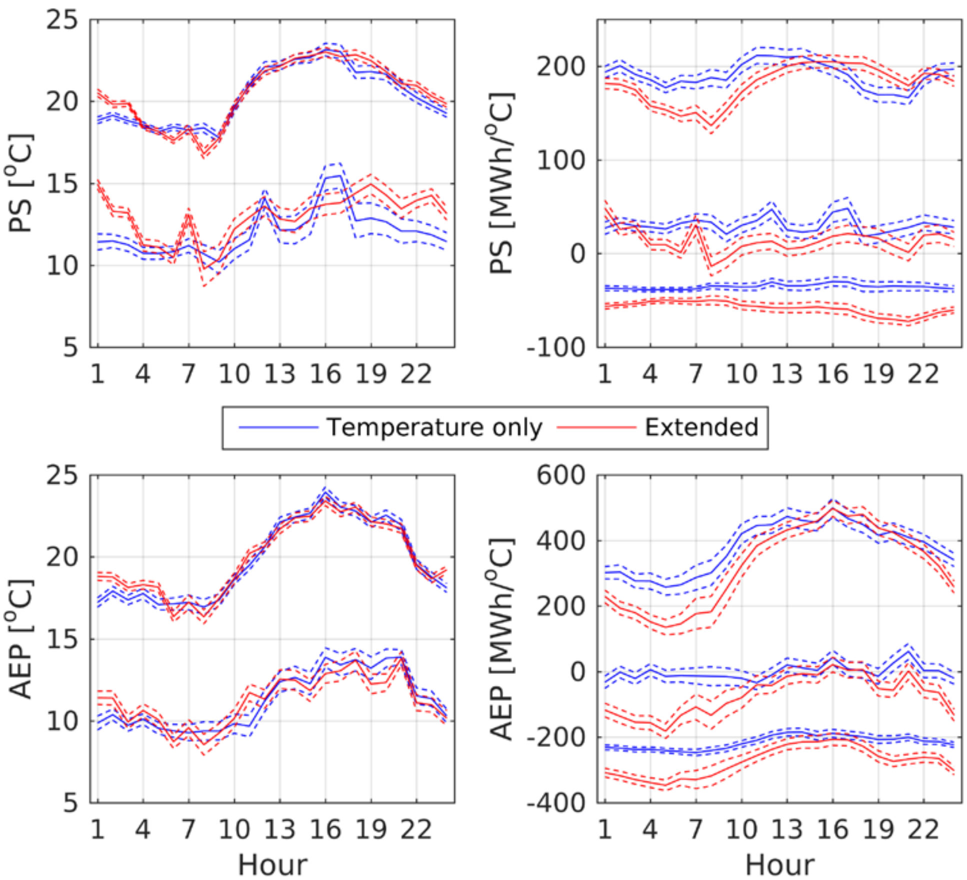


Figure 3. The estimated breakpoints and electricity response slopes in the PS and AEP zones by the temperature-only model and the extended model. Solid lines are the point estimations. Dashed lines show the 95% confidence intervals.

The effect of past temperatures on the same day is illustrated in Figure 4 for the extended model. The relationship between varying lags (*j-j'*) up to 7 hours and mean absolute percentage errors of the extended model is displayed. As can be seen, in the PS zone, electricity load was generally best predicted by temperature on the same hour, though some small lags appeared to exist between hours 3 and 6. In the AEP zone, the smallest errors were obtained at non-zero lags between hours 23 and 8, with the clearest effect being between hours 24 and 5. This lagged effect may be attributed to building insulation effects, where indoors temperature do not adjust immediately to outdoor temperature.

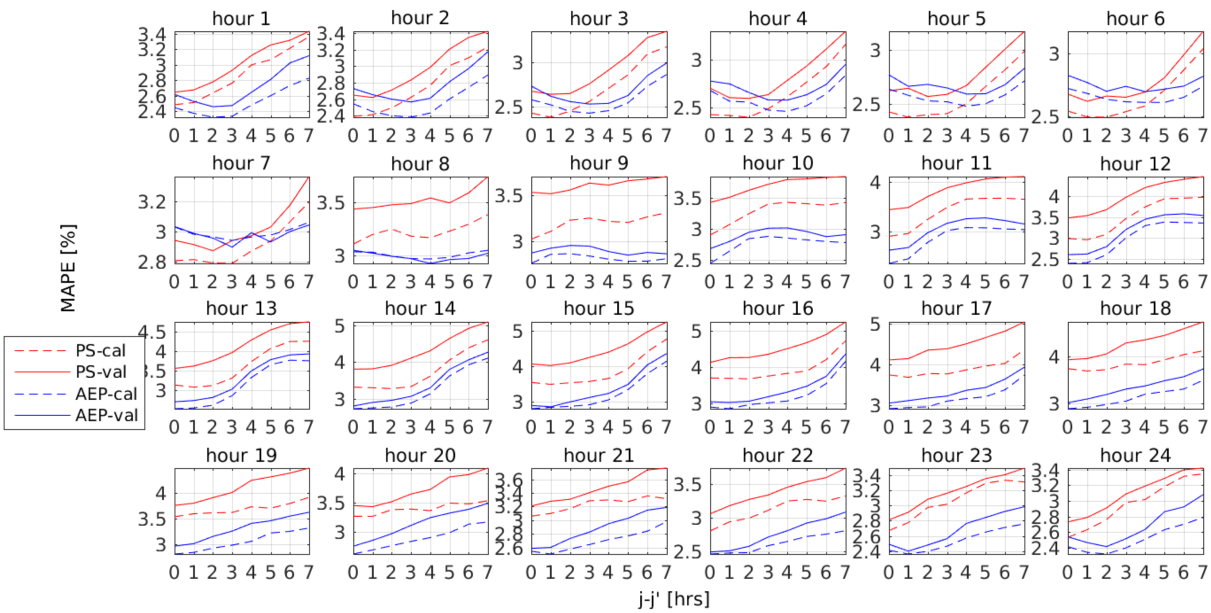


Figure 4. The effect of spacing between the hour of electricity observation *j* and the hour of temperature observation *j'* (*j*, *j'* = 1, 2, …, 24) on model MAPE. The MAPE was calculated from all the residuals in the calibration (cal) and validation (val) periods in the PS zone and the AEP zone.

The effect of past days' temperatures on electricity load are shown in Figure 5 for the selected hours 1, 10, 15, and 20. It shows that the past 1-day temperature has the strongest effect on electricity load, with clear positive coefficients in the summer months and negative coefficients in the winter months. The effects of past 2-3 days' temperatures were smaller, probably generally 0 in the winter months but sometimes positive in the summer months, provided that the uncertainty in the coefficients was similar to the uncertainty in the slopes. Overall, the slopes suggest that past days' temperatures amplified the relationship between present-day temperature and electricity load: higher past days' temperatures in winter resulted in lower electricity load, and in summer higher electricity load. This is consistent with the past observed cumulative effect in urban environments in summer (Li et al., 2014). In winter, the effect of past days' temperatures might be because some building managers only start heat supply after temperature was below a threshold for a certain amount of time. In past forecast studies at regional level, past days' cooling- and heating-degrees had also been found to positively correlate with electricity load (Dordonnat et al., 2008; Mirasgedis et al., 2006).

Judging by the number of standard deviations between the data points and x-axis, the summer cumulative effect was stronger in the PS zone than in the AEP zone. This might be because the PS zone area is more urbanized. The winter effect was stronger in the AEP zone, which might be related to the greater use of electricity for heating in the southern part of the AEP zone (EIA, 2015).

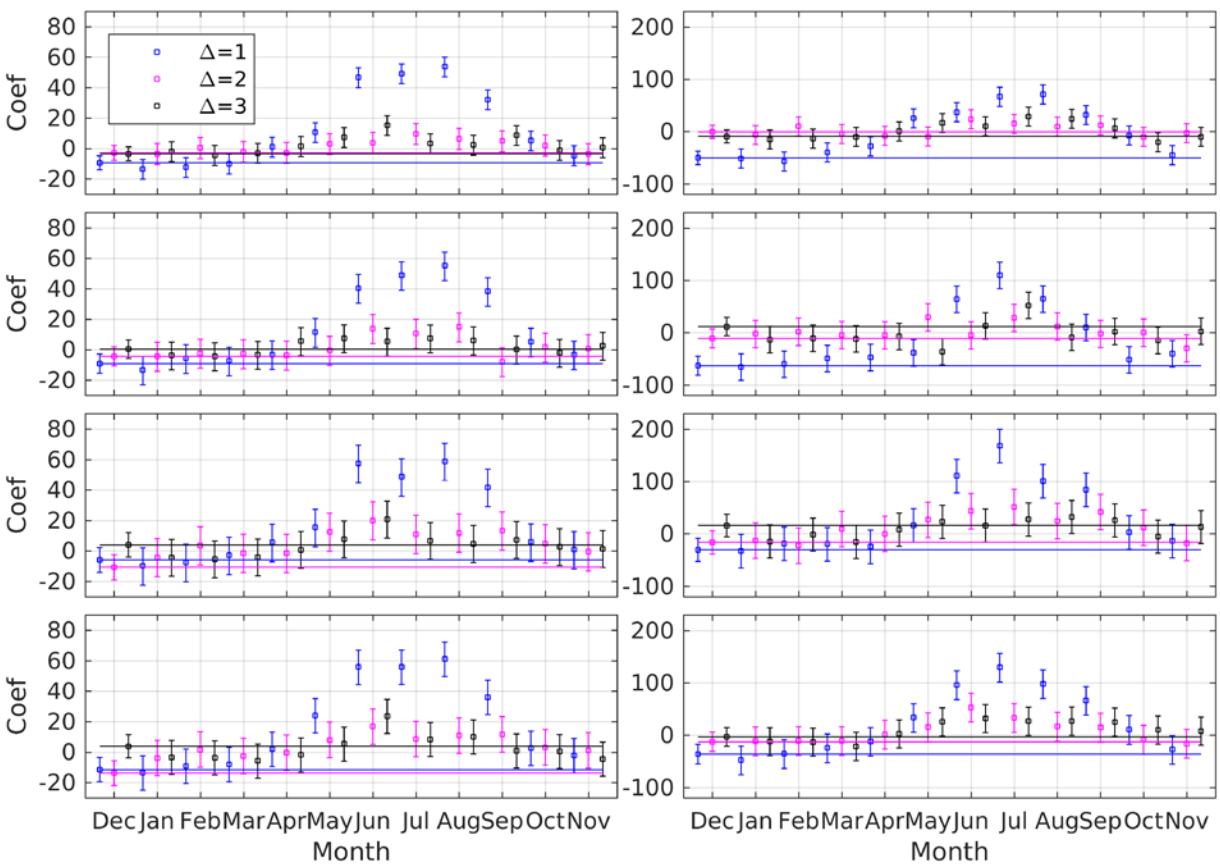


Figure 5. The slopes of electricity response to past days' temperatures ($TP_{t-\Delta,j'}, \Delta = 1, 2, 3$) at hours (top to bottom) 1, 10, 15, and 20 in the PS (left) and AEP (right) zones. Whiskers are ±2*standard

deviations of the regression coefficients. Horizontal lines are reference lines extended from the December coefficients, relative to which other months' coefficients are plotted.

The effect of relative humidity in the extended model ($k_j RH_{t,j} + l_j(TP_{t,j'} \times RH_{t,j})$) cannot be interpreted based on each coefficient, because of the existence of interaction term. The estimated k_js were sometimes negative, but the estimated l_js were always positive. Figure 6 illustrates the relationships between this combined term and electricity load, and between this combined term and temperature. Due to its high collinearity with temperature, this term had a similar relationship with electricity load to temperature. This high collinearity was also identified as the cause of overestimation of the slope of the heating arm in Figure 3. In the high-temperature region, the term was positively correlated with electricity load. The results suggest that while the effect of relative humidity is temperature-dependent, the functional form needs further development. It is hypothesized that relative humidity may only show impact on electricity demand above a temperature threshold. Since the regression model (Muggeo, 2008) used in this study cannot estimate such a relationship, the hypothesized new model is expected to be tested in a follow-up study that also incorporates socioeconomic factors into the statistical model.

The effects of wind speed in the extended model is displayed in Table 1. In the PS zone, wind speed was positively correlated with the load in the night, but negatively correlated in some afternoon hours. In the AEP zone, wind speed was mostly positively correlated with load. The sign of the relationship would have depended on whether the cooling effect of wind was more dominant in winter (causing higher electricity load) or in summer (causing lower electricity load). Compared to other terms, the effects of wind speed is minor and can probably be ignored in long-term prediction.

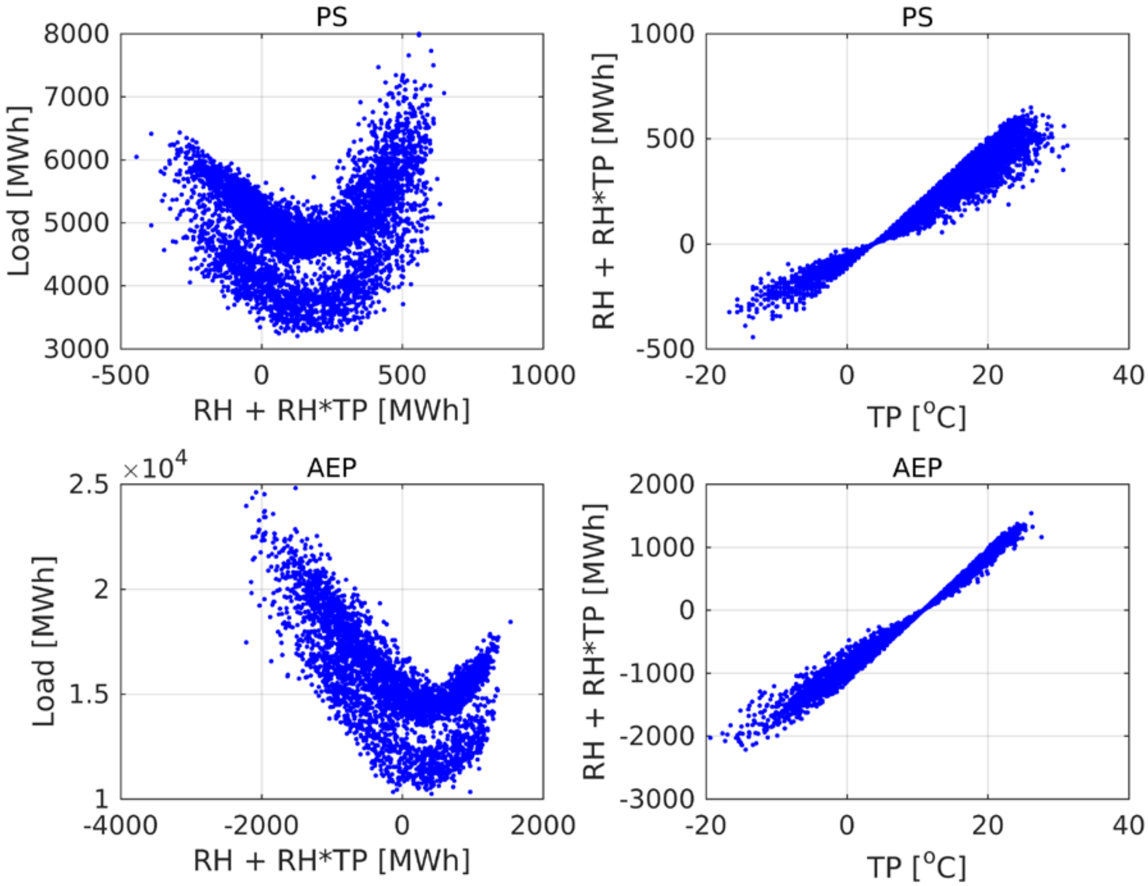


Figure 6. The relationship between $k_j RH_{t,j} + n_j(TP_{t,j}, \times RH_{t,j})$ and electricity load or temperature in the PS and AEP zones at hour 8.

Table 1. The regression coefficients of wind speed in the PS and AEP zones.

Hour	PS		AEP		Hour	PS		AEP	
	n_j	*t* value	n_j	*t* value		n_j	*t* value	n_j	*t* value
1	5.12	1.29	14.3	4.769	**13**	-4.78	1.83	25.65	6.349
2	3.05	1.16	14.57	4.661	**14**	-3.91	1.88	19.26	6.979
3	4.24	1.11	7.803	4.494	**15**	-4.86	2.08	20.77	7.428

4	3.60	1.07	7.692	4.69	**16**	-3.99	2.20	20.45	7.073
5	3.77	1.08	7.885	4.964	**17**	-5.93	2.36	16.69	7.208
6	4.22	1.14	-8.878	5.68	**18**	-6.88	2.48	16.59	7.303
7	3.56	1.40	-13.16	7.539	**19**	-3.17	2.33	22.73	7.39
8	4.69	1.83	-11.65	8.556	**20**	-0.77	2.16	27.96	7.186
9	5.01	1.82	2.936	8.282	**21**	-1.54	1.95	29.58	6.816
10	1.51	1.78	6.568	6.885	**22**	-0.29	1.78	25.37	6.463
11	-0.03	1.73	22.5	6.428	**23**	1.54	1.59	18.07	5.577
12	-2.83	1.74	29.94	6.19	**24**	3.15	1.42	16.87	5.024

4 The water use of thermoelectric power plants

4.1 Methodology

A version of corrected EIA Form 923 water use data for the year 2008 (Rogers et al., 2013) have been examined for the distribution of thermoelectric power plants and cooling technologies in the US. Water use in this dataset is representative of ~30% to ~50% of the US thermoelectric power plants, due to missing data in the original EIA Form 923. This dataset is on the annual level and therefore cannot adequately reflect the influence of weather variables. The original EIA Form 923 contains numerous errors in the locations of power plants, cooling systems, and water use values. Those are expected to be corrected following the approach of past studies (Averyt et al., 2013; Diehl et al., 2013).

4.2 Principal findings

Figure 7 and Figure 8 show some spatial features of the water withdrawal and consumption factors according to the preliminary dataset (Rogers et al., 2013). Overall, the coal-fired power plants tend to be located in the northeastern part of the country, while the more efficient natural gas power plants tend to be located in the southwestern part. More of the natural gas power plants tend to be cooled by recirculating technologies than coal-fired power plants. Along the coastlines, some once-through cooled power plants exist that use ocean water. It can also be seen that for the once-through power plants, the consumption factor is much lower than withdrawal factor, while for the recirculating power plants, the two factors are similar.

Figure 9 and Figure 10 show that a weak negative relationship exists between the water use factors of power plants and their capacity factors. This might be because the very infrequently used power plants are equipped with less efficient technologies to reduce the cost.

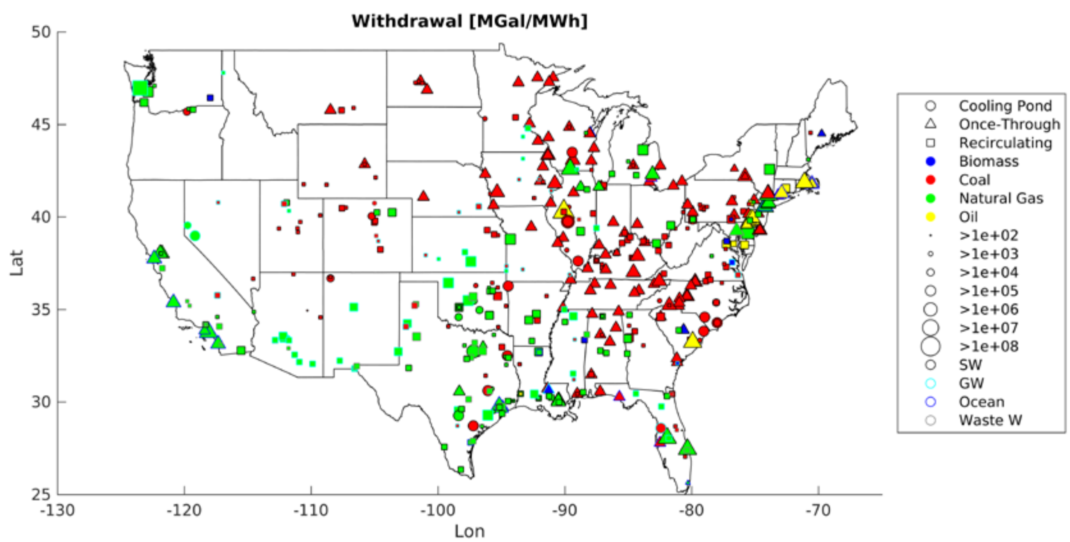


Figure 7. The spatial distribution of water withdrawal factors of the power plants (by size of the labels), as well as the cooling technologies, fuel type, and source of water (SW- surface water; GW – groundwater; Ocean – ocean; Waste W – wastewater).

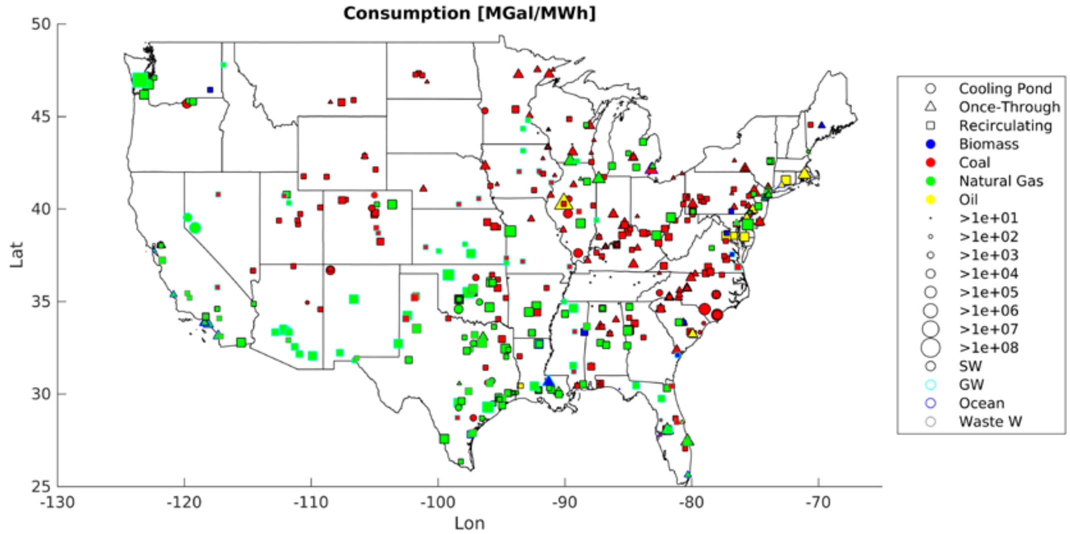


Figure 8. The spatial distribution of water consumption factors of the power plants (by size of the labels), as well as the cooling technologies, fuel type, and source of water (SW- surface water; GW – groundwater; Ocean – ocean; Waste W – wastewater).

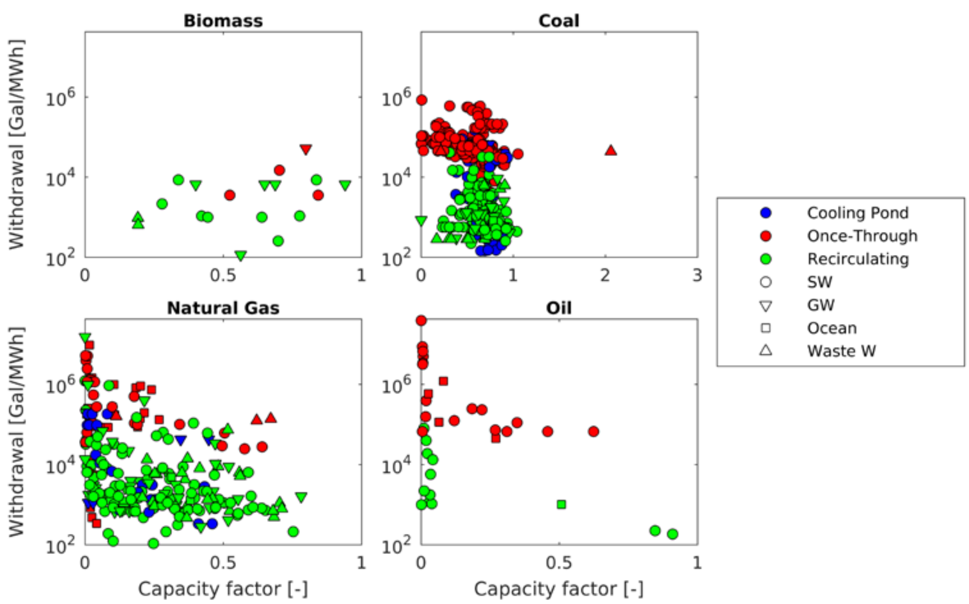


Figure 9. The relationship between water withdrawal factors and capacity factors of the power plants.

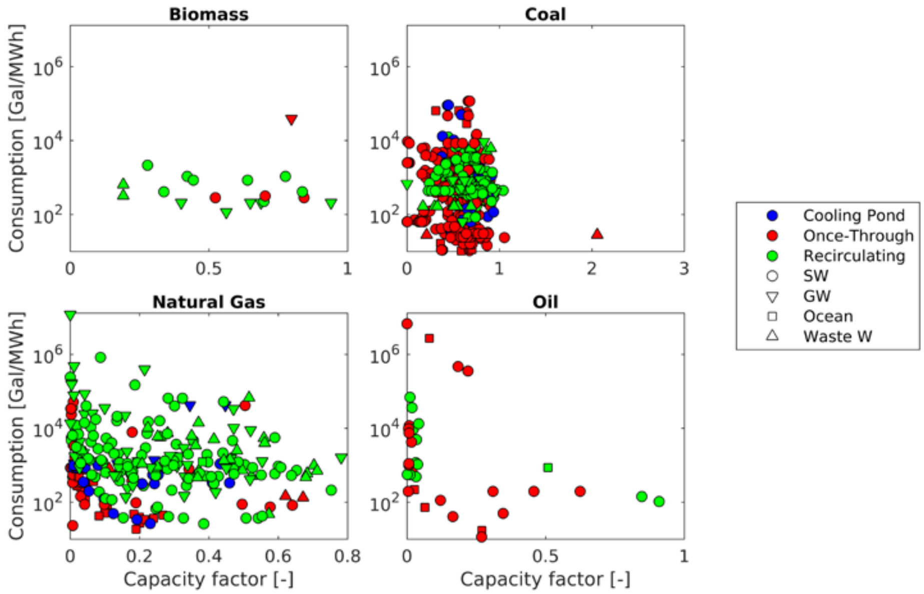


Figure 10. The relationship between water consumption factors and capacity factors of the power plants.

5 Significance

The study estimated the relationship between electricity load and temperature without prior assumptions about base temperatures or the slopes of electricity response. The empirically estimated base temperatures can serve as a reference for other future studies that wish to use CDDs and HDDs for load forecasting. The existence of a comfort zone indicates that the traditional assumption that the base temperatures are equal for CDDs and HDDs is inaccurate.

For other meteorological variables, the finding on the effect of past temperatures corroborates the findings of past studies, and the finding on wind speed shows it is relatively unimportant at the study regions. The finding on the effect of relative humidity is interesting because it suggests that effect of relative humidity is temperature-dependent, but cannot be modeled simply using an interaction term. While relative humidity is generally understood to induce higher electricity demand on hot days, the actual form of its relationship with electricity demand has not be well-studied. The results suggests a potential direction to be explored in future studies.

The study on water use of power plants is still in progress. Temporal-spatial statistical model of water withdrawal and consumptions factors are expected to be constructed based on multiple years of EIA Form 923 data. Monthly data is expected to be used. This will contribute to current literature by illustrating the seasonal and inter-annual evolution of the water use factors.

Baseline measurements of methane emissions from rivers and lake waters in the proposed site of the OSU hydrofracking research station

Basic Information

Title:	Baseline measurements of methane emissions from rivers and lake waters in the proposed site of the OSU hydrofracking research station
Project Number:	2015OH480O
Start Date:	6/1/2015
End Date:	5/30/2017
Funding Source:	Other
Congressional District:	3rd
Research Category:	Climate and Hydrologic Processes
Focus Category:	Surface Water, Climatological Processes, Wetlands
Descriptors:	Membrane separations; water treatment; biomimetic
Principal Investigators:	Gil Bohrer

Publications

There are no publications.

Progress Report 2013-2014

Contract Information

Title	Baseline measurements of methane emissions from rivers and lake waters in the proposed site of the OSU hydrofracking research station
Project Number	6300
Start Date	6/1/2015
End Date	5/30/2017 (including a 12 Month no-cost extension)
Lead Institute	The Ohio State University
Principal Investigators	Gil Bohrer

Abstract
The goal of this project is to provide baseline measurements of methane emissions from the site of future fracking operation in Noble County, Ohio. We leverage on the Ohio State University NETL grant that provides the site, access and opportunity to conduct measurements before and during all stages of the fracking and production processes. We will combine eddy covariance and chamber measurements of the methane flux. Deployment of the observation setup months before drilling operations start will allow establishment of a baseline for the natural emissions of methane in and around the drill site. Originally, the NETL project was planned to be conducted in the OSU Eastern Extension Station in Noble County, and frack in agricultural land. However, the planned activity for the NETL project and the fracking site was changed and the potential new locations are all farther from OSU campus and in forested land. This project leverages on an NSF grant to provide base-line measurements for a future fracking site, which was awarded to PI Gil Bohrer. Specifically the funds from Ohio WRC were requested to supplement travel (to the farther site) and materials (taller tower is needed in forested landscape) that where not accounted for in the NSF grant, which was proposed for the original fracking site.

Methodology

1) A 30 m tall flux tower will be located downwind the planned well pad locations. The tower is relatively tall to allow a wide footprint area.
2) Chamber measurements of methane emissions from Piedmont lake and wetlands at its coast.
3) A 2-D multi-patch footprint model will be used to determine the relative contribution of each component of the landscape (including natural processes and the constructed drill pad) to the methane flux signal at each half hour of observation.
4) An automated neural network model (ANN) will be used, driven by the meteorological observations, footprint calculation, eddy-flux measurements and chamber flux measurements to resolve the natural methane emissions.

Major Activity
Unfortunately, the NETL project has not managed to secure a study site yet and activities in the USEEL have not started. Therefore it was impossible for us to start our fieldwork. The difficulty of finding a site stems mostly from dropping gas prices which postponed most of the new fracking activity and made potential land owners and commercial fracking partners reluctant to

add any limitations to their proposed sites. Negotiations with the Muskingum Watershed Conservancy District have failed an efforts to secure a site for USEEL are now focused on a location in West Virginia. We are optimistic that we could start the field work in an approved site by early next spring. A 1-year no-cost extension for the project was requested and approved.

Findings

None to date

Significance

The project will provide baseline measurements of methane emissions from natural and agricultural aquatic ecosystems around the proposed locations of a hydrofracking site. These observations will allow developing an empirical model for the natural methane emissions from the water system at the site and will allow determining whether these emissions increase due to diffused methane release into the ground water after the drilling operations started.

Spatial Demand Estimation: Moving Towards Real-Time Distribution System Network Modeling

Basic Information

Title:	Spatial Demand Estimation: Moving Towards Real-Time Distribution System Network Modeling
Project Number:	2015OH481O
Start Date:	3/1/2015
End Date:	2/28/2016
Funding Source:	Other
Congressional District:	1st
Research Category:	Engineering
Focus Category:	Water Supply, Management and Planning, Models
Descriptors:	Membrane separations; water treatment; biomimetic
Principal Investigators:	Dominic L Boccelli

Publications

1. Oliveira, P. J., Rana, S. M. M., Qin, T., Woo, H., Chen, J. and Boccelli, D. L.(2016). “Case Study: Evaluation of a Composite Demand-Hydraulic Modeling Framework.” 2016 Water Distribution System Analysis Symposium, Cartagena, Colombia.
2. Chen, J. and Boccelli, D. L. (2015). “A Real-Time Demand-Hydraulic Model of Water Distribution Systems.” Proceedings of the World Water and Environmental Resources Congress, ASCE, Austin, TX.
3. Qin, T. and Boccelli, D. L. (2015). “Grouping Water Demand Nodes by Similarity Among Flow Paths in Water Distribution Systems.” Proceedings of the World Water and Environmental Resources Congress, ASCE, Austin, TX.

Spatial Demand Estimation: Moving Towards Real-Time Distribution System Network Modeling

1 Problem and Research Objectives

Water utilities must ensure that our water infrastructure is sustainable, robust and resilient to both long- and short-term forcing factors. While long-term factors (e.g., climate change, population shifts) are likely to be addressed through changes in infrastructure design, short-term factors (e.g., intrusion events, main breaks) can be addressed through real-time monitoring and decision support. Unfortunately, existing "real-time" decision support tools that, for example, are intended to assist with pump scheduling, are limited in practice as they require real-time estimates of the current and/or future states of the system (e.g., flow rates, water quality concentrations, etc), which are only observed at limited locations throughout the network. To compensate for the lack of observed data, estimates of the user demands – the driving factors for the underlying hydraulic and water quality dynamics – are required to simulate the system-wide states through a network model. Recent software developments have begun to integrate observed data with network models (e.g., IWLive [Innovyze]; Polaris [CitiLogics]), but only include simplistic demand estimation approaches and limited (if any) forecasting capabilities. Thus, there remains a critical need for real-time demand estimation and forecasting to fill the gap between data-model integration and the development of real-time decision support tools. The objective of this project, which is the next step in progressing towards our long-term goals, is to develop a composite demand-hydraulic model – one that couples a demand model with a network hydraulic solver – capable of being updated in real-time using observed hydraulic information.

2 Methodology

Our central hypothesis is that the observed hydraulic data commonly collected via utility SCADA systems can be used to estimate the expected values and uncertainty of a structured demand model that characterizes the temporal and spatial patterns of consumptive demands. Our rationale for developing a composite demand-hydraulic model that can be updated in real-time is to provide the framework for forecasting temporally and spatially correlated demands and the associated network hydraulics. In turn, these capabilities will provide for the development of real-time decision making tools associated with, for example, optimal pump scheduling to minimize energy costs. We will test our central hypothesis and associated objectives by pursuing the following: 1) the development of a composite demand-hydraulic model that will integrate a vectorized times series model with a network hydraulic solver; 2) the implementation of an expectation-maximization algorithm to estimate the demands and model parameters using limited observed hydraulic information; and 3) develop a clustering approach, based on water quality information, to reduce the parameterization of the demand estimation problem.

2.1 *Composite Demand-Hydraulic Model.* The proposed composite demand-hydraulic model will be formulated as a Dynamic Bayesian Network (DBN) – a generic framework for representing complex conditional probabilistic models (Ghahramani, 1998; Koller and Friedman, 2009) – by integrating a time series demand model with a distribution system network hydraulic solver for estimating the hydraulic states (e.g., flow rate, pressure, tank levels, etc) of a distribution system. Figure 1 illustrates the linkage of the variables (boxed) and sub-models, and the conditional relationships of the DBN as the hydraulic states of the network are conditioned upon the distribution of demands, which are themselves conditioned on the demand model parameter estimates. The following further describes the demand and hydraulic sub-models with the estimation algorithm presented in the next section.

$$\boxed{\text{Model parameters}} \xrightarrow[\text{(Time Series)}]{\text{Demand sub-model}} \boxed{\text{Demands}} \xrightarrow[\text{(Solver)}]{\text{Hydraulic sub-model}} \boxed{\text{Hydraulic variables}}$$

Figure 1: Variables (boxed) and sub-models of the proposed demand-hydraulic model.

Demand sub-model. The demand sub-model is proposed as a vectorized Seasonal Auto-Regressive Integrated Moving-Average (ARIMA) time series model (Box and Jenkins, 1976; Wei, 2006), which is capable of quantifying individual, or aggregated, user demands that are both temporally and spatially correlated. Ideally, the number of AR and MA parameters would be determined via standard model identification procedures (Box and Jenkins, 1976). However, in the case of the proposed model, the demand sub-model cannot be identified with regular methods because direct observations of individual, or aggregated, user demands are generally not available. Therefore, we will initially assume that the ARIMA model structure of the demand sub-model will have the same form as the total system demand.

As an example, Chen and Boccelli (2013) showed that a double seasonal autoregressive (AR(2)) model was sufficient to forecast the total system demand from a partner utility. Thus, assuming there are N individual, or aggregated, water users, the demands at time t can be denoted as a N-dimensional vector $\boldsymbol{q}_t = \left[q_{t(1)}, q_{t(2)}, ..., q_{t(N)}\right]^T$, and the resulting double seasonal AR(2) demand sub-model formally expressed as:

$$\nabla_{s_1}\nabla_{s_2}\left(\boldsymbol{q}_t - \boldsymbol{\phi}_1 \cdot \boldsymbol{q}_{t-1} - \boldsymbol{\phi}_2 \cdot \boldsymbol{q}_{t-2}\right) = \boldsymbol{a}_t \tag{1}$$

where ∇_s is the differencing operator defined as $\nabla_s \boldsymbol{q}_t = \boldsymbol{q}_t - \boldsymbol{q}_{t-s}$; s_1 and s_2 represent the lengths of weekly and daily demand periods ($s_1 = 168$ and $s_2 = 24$ for an hourly demand model); $\boldsymbol{\phi_1}$ and $\boldsymbol{\phi_2}$ are the vectors of the autoregressive parameters (e.g., $\boldsymbol{\phi}_1 = \left[\phi_{1(1)}, \phi_{1(2)}, ..., \phi_{1(N)}\right]$); and $\boldsymbol{a}_t$ is a multi-dimensional white noise process with mean 0 and covariance matrix $\boldsymbol{\Sigma}$ that accommodates the spatial correlation. While the time series model may vary for different systems, the proposed approach is generalizable for different vectorized seasonal ARIMA models.

Hydraulic sub-model. The hydraulic sub-model will be based on EPANET (Rossman, 2000) – a common distribution system network hydraulic and water quality solver. Assuming there are K online monitors in the distribution system (e.g., flow rate, pressure, etc), the observed SCADA data can be denoted by $\boldsymbol{Y}_t = \left[Y_{t(1)}, Y_{t(2)}, ..., Y_{t(K)}\right]^T$, and described as

$$\boldsymbol{Y}_t = \boldsymbol{h}(\boldsymbol{q}_t) + \boldsymbol{e}_t \tag{2}$$

where $\boldsymbol{h}(\cdot)$ represents the hydraulic solver that translates the demands into estimates of the observable hydraulic variables, and $\boldsymbol{e}_t$ is a vector of random measurement errors assuming independent Gaussian distributions. The inclusion of the sensor measurement error facilitates the calculation of conditional likelihoods of having observed the hydraulic measurements, which facilitate the estimation of the demands and demand model parameters.

2.2 *Demand and Parameter Estimation.* In the composite demand-hydraulic model, the intermediate variables of water demands are latent (i.e., no direct observations are available). Therefore, the parameters of the demand sub-model can not be estimated via regular maximum likelihood methods for time series analysis. However, the Expectation-Maximization (EM) algorithm (Dempster et al., 1977) was designed to compute maximum likelihood estimates (MLE) for models with incomplete observations. Specifically, an EM-Markov chain Monte Carlo (EM-MCMC) algorithm (Pasula et al., 1999) is proposed for the parameter estimation. Figure 2 illustrates the EM algorithm, which includes two steps – an E-step and an M-step – executed iteratively to determine the MLE of the demand model parameters.

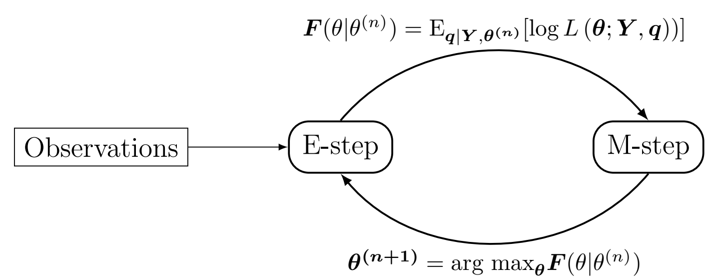


Figure 2: The EM algorithm. $\boldsymbol{\theta}$, $\boldsymbol{q}$, and $\boldsymbol{Y}$ denote the time series parameters, latent variables, and observable variables, respectively. $\boldsymbol{F}(\theta|\theta^{(n)})$ denotes the expectation of the log-likelihood function conditioned on best estimates in the nth iteration.

The Expectation (E-) step generates the distribution of possible demands conditioned upon: 1) the observed hydraulic data, and 2) the current parameter estimates associated with the vectorized seasonal ARIMA model. Unfortunately, the resulting conditional demand distribution is not analytically tractable. Thus, the population of demand vectors will generated by an MCMC algorithm (Brooks et al., 2011; Gilks et al., 1995; Metropolis et al., 1953) – a sampling method for calculating statistics of complex, multi-dimensional probability distributions suitable for generating the conditional probability distributions associated with BNs (Koller and Friedman, 2009; Robert, 2007). We will implement the MCMC algorithm using the Differential Evolution Markov Chain (DE-MC) (Laloy and Vrugt, 2012) approach, which has been developed to improve the performance and convergence speed of the MCMC algorithm. The Maximization (M-) step then utilizes the distribution of demand vectors generated in the E-step to update the parameters of the demand sub-model. To update the parameters of the demand sub-model, a standard minimum sum of square (MSE) algorithm for ARIMA models (Box and Jenkins, 1976) will be used to generate the MLEs. The new MLEs replace the current demand model parameters to start a new EM cycle until the difference between MLEs generated by two consecutive cycles is less than some threshold. The resulting estimates of the EM algorithm have been shown to converge to the parameter estimates of a MLE algorithm (Dempster et al., 1977; Koller and Friedman, 2009).

2.3 *Spatial Aggregation.* While the proposed composite demand-hydraulic model is capable of generating temporally and spatially correlated demands, estimating the model parameters for a realistic network model, in which the number of consumer nodes (N) can range from 10^4 to 10^5, is not realistic. First, the computational burden would be too significant to perform the estimation in real-time. Second, the typical amount of observational data, K, is likely insufficient to accurately estimate the parameters at such fine spatial scales. Thus, an approach to group, or cluster, nodes that are assumed to behave similarly is required. To develop the clusters, we will utilize the approach of (Qin and Boccelli, 2015) that utilizes a backtracking algorithm (Shang et al., 2002) to determine the average hydraulic travel paths of every node within the network from the last 24-hours of a sufficiently long simulation. A correlation matrix generated based on the path information from each node, and a k-nearest-neighbor (*knn*) clustering algorithm (Larose, 2005) is used to identify nodes with similar path histories. The advantages of this algorithm are two-fold as the clustering approach will identify locations that are: 1) common to the larger, upstream flows, and 2) have similar residence time and hydraulic paths that should have longer-term benefits associated with water quality modeling. Figure 3 shows a portion of ten clusters from a test network using the hydraulic path correlation matrix and *knn* algorithm, as well as simulated chlorine signals from three randomly selected locations within each cluster. These results demonstrate the similarity in water quality signals resulting from similar hydraulic paths. While this approach must be performed on the network model prior to demand estimation, recent studies (van Thienen and Vries, 2013; Yang and Boccelli, 2013) have shown that the random nature of demands can impact the underlying residence times, but do no not significantly alter the hydraulic paths. Thus, determining the grouping of nodes using hydraulic path information should provide an adequate trade-off between spatial aggregation and model performance.

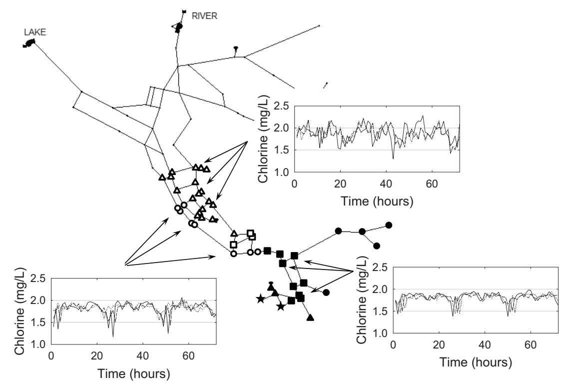


Figure 3: Water quality signals from locations within clusters identified using hydraulic path similarity [symbols] that demonstrate similar dynamics due to similar hydraulic paths.

3 Principal Findings and Significance

The following two sub-sections present the findings associated with the development and implementation of the composite demand-hydraulic model, and the analysis of the clustering algorithm for identifying nodes with similar water quality information.

3.1 Composite Demand-Hydraulic Model

3.1.1 *Case Study.* The "Net1" network included in the EPANET software package, shown in Figure 4(a), was used to demonstrate the capabilities of the composite demand-hydraulic parameter estimation algorithm. The network includes nine junctions (of which eight represent consumer nodes), twelve pipes, one reservoir, one pump, and one storage tank. In the original model, the eight customers belong to a single demand group with temporal changes in demands represented by a demand multiplier pattern that repeats every 24 hours. The

Table 1: Variables recorded within the virtual SCADA database

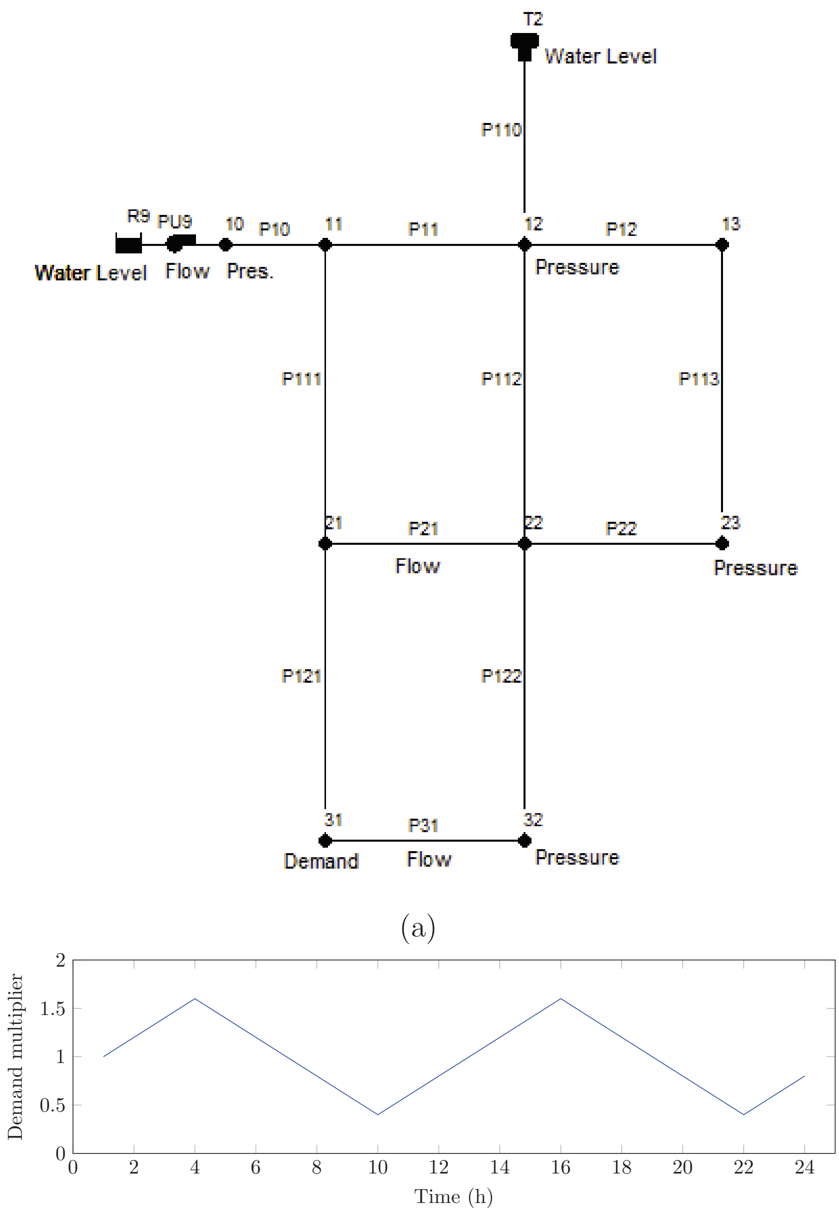
ID	Location	Variable	Unit
1	PU9	On/Off	
2	R9	Water level	Feet
3	T2	Water level	Feet
4	PU9	Flow rate	GPM*
5	P21	Flow rate	GPM
6	P31	Flow rate	GPM
7	10	Pressure	PSI†
8	12	Pressure	PSI
9	23	Pressure	PSI
10	32	Pressure	PSI
11	31	Demand	GPM

*GPM = Gallons Per Minute; †PSI = Pounds Per squared Inch

average total demand for the network is 1100 GPM and the max total demand is 1760 GPM. The selection of a small network was to facilitate a detailed analysis of the performance of the proposed estimation algorithm, which includes a correlation analysis of the water demands that would prove more difficult with a larger, realistic sized network model. For large-scale networks the methodology is still applicable, but the preliminary clustering and/or grouping of the nodes may be required to improve the computational performance.

To create a virtual SCADA database for the network, the observations of hydraulic states for a total duration of 504 hours (three weeks) were simulated in this study. During the generation process, random variations were added to the water demands that drive an extended period simulation (EPS) routine. Once the hydraulic states were computed, further random errors were added to the results to generate the actual measurements. To mimic the limited online data coverage of the real systems, only an incomplete set of hydraulic variables were written to the SCADA database. Table 1 lists the hydraulic variables monitored by the virtual SCADA system. Similar to a real SCADA system, the water levels of Reservoir R9 and Tank T2 were monitored in real-time. Three variables related to Pump station PU9 were also recorded: the on/off status, the flow rate, and the pressure on the discharge side. In addition, there were assumed to be three pressure transducers and three flow meters in the network producing hourly readings. Among the online flow meters, two were positioned along the Pipes 12 and 23, one was positioned to monitor the real-time water demand at Junction 31. The real-time water demands for the other seven consumers are unknown and will have to be estimated during the parameter estimation process. The 3-week simulated SCADA data were pre-computed and stored in a PostgreSQL 9.4 database.

A parameter estimation program was developed to implement the EM-MCMC algorithm described previously. The program was written in C/C++ and includes an EM core function along with miscellaneous routines for input/output processing. In the E-step, the demands are estimated based on the log-likelihoods of observing the measured flow rates, pressure and tank levels, and observing the estimated demands given the current parameter estimates for



(b)

Figure 4: (a) Map of the network "Net1"; the names of the nodes and links are shown close to their respective points and lines, and the text of "water level", "flow", "pressure", and "demand" represent the hydraulic variables recorded by the virtual SCADA system; and (b) the original demand pattern of Net1 with 24 hourly demand multipliers shown in the chart.

Table 2: Algorithmic parameters for the EM-MCMC algorithm

Name	Notation	Value
Start time of the estimation window	t_0	337
Estimation window size	W	168
MCMC chain size	M	10,000
Burn-in size	M_0	2,000
Standard deviation of the proposal density	σ_1	1
EM convergence threshold	ϵ_0	10

the time series demand model. In the M-step, the parameters of the time series demand model are estimated using the estimated demand information from the E-step. The algorithmic parameters used in this study are listed in Table 2. The estimation window was selected as the third week from the three-week simulated SCADA data. The "burn-in" size and total chain size of the MCMC were empirically selected to ensure the removal of initial boundary effects and proper chain mixing. The standard deviation of the proposal density is chosen to maintain an acceptance rate of 20% to 50% in the MCMC chains. The EM convergence threshold is set as 10, or an average shift of 0.05 for every parameter.

The parameter estimation program ran on a PC with a 2.5GHz Intel Sandy-Bridge CPU and 6 GB of memory. The program retrieved and processed the hydraulic measurements for the last week of the virtual SCADA database. In the test, 15 E-M cycles were required for the parameters to converge under the preset threshold, resulting in a total running time of around 80 minutes to estimate the complete set of demands during the third week.

3.1.2 *Results and Discussion.* The effectiveness of the proposed EM-MCMC algorithm relies on two factors. First, during the E-step, the MCMC sampler must produce a well-mixed chain of samples that represent the overall distribution of the water demands. Second, the EM iterations must converge to a point estimate of the parameters. If both criteria are met, the algorithm will produce the final time series model parameter estimates and demand estimates. Using the parameter and demand estimates, spatial and temporal correlations of the multivariate water demands can be analyzed for the network being studied.

3.1.2.1 MCMC Sampling. To illustrate the performance of the MCMC sampler, Figure 5 shows the Box-Whisker plots of individual water demands versus the size of the MCMC chain (including the burn-in samples) for the first time step in the first E-M iteration. The data are shown in seven subplots, each of which represents one dimension, or a single consumer node. For a subplot, the X-axis represents the number of sample points generated, and the Y-axis represents the value of water demands. For example, in the first subplot the fifth box from the left represents the empirical distribution of the first 5,000 samples comprising the MCMC chain. The upper edge, middle line, and lower edge in a box represent 25%, 50% (i.e., median), and 75% percentiles. The upper and lower whiskers represent the approximated extents for the population. Values are drawn as outliers if they are larger than $p_1 + 1.5 \times (p_3 - p_1)$ or smaller than $p_1 - 1.5 \times (p_3 - p_1)$, in which p_1 and p_3 are the 25th and 75th percentiles, respectively. The outliers are marked individually in the plot as "jittered" dots.

All of the subplots in Figure 5 demonstrate similarities in fluctuations of the demand percentiles for the first two to three thousand samples. The visuals do not change as significantly after these initial periods, indicating that the statistics of the sampled water demands are stabilized. The analysis on the other time steps and in subsequent E-M iterations results in similar conclusions. Based upon the convergence information from these plots, the length of the MCMC chain used in this study was set at 10,000 with the first 2,000 samples discarded as the "burn-in" period and the last 8,000 samples utilized for computing the demand estimates.

3.1.2.2 EM Convergence. To evaluate the performance of the E-M algorithm, the parameters produced by successive E-M iterations are investigated. The changes to parameters are quantified using $\epsilon^{(r)} = \left|\Theta^{(r+1)} - \Theta^{(r)}\right|$, or the Euclidean distance between the parameters. In each E-M iteration, the hydraulic and demand likelihoods are calculated for evaluating the performance of the parameters. Figure 6 shows the changes in parameters and likelihoods during the test run of the parameter estimation algorithm. The X-axis is the number of E-M iterations, the left Y-axis is $\epsilon^{(r)}$ in base-10 logarithmic scale, and the right Y-axis is log-likelihood for the parameter estimate. The crisscross symbol denotes the parameter changes $\epsilon^{(r)}$. The circles, diamonds, and squares denote hydraulic likelihoods, demand likelihoods, and total likelihoods, respectively. Based on the information in Figure 6, the parameter estimates changed significantly during the first four iterations. Thereafter, the iterations yielded gradually smaller changes to the parameters. ϵ dropped below the pre-set threshold of 10 after the 15th iteration when the algorithm stopped. The refinement of the parameters is accompanied by the increasing likelihoods. The trend lines of the likelihoods show that major improvements happen in the first three iterations, which is mainly driven by the improvements associated with the hydraulic likelihoods. The demand likelihood showed a temporary decrease in the second iteration before increasing again. All three measures of likelihood continued to increase after the fourth iteration, but the gains in likelihoods became less significant as the parameter estimates converged to the final values. The overall analysis showed that the EM algorithm was effective in converging to a set of the parameters estimates for the composite model.

3.1.2.3 Demand Estimates. The EM iterations not only produced the parameter estimates but also estimates of the consumer demand and the hydraulic states of the system. In this study, the estimated demands and the "real" demands can be compared, because the simulated "real" water demands, though not exposed to the EM algorithm, have been recorded separately during SCADA data generation. Table 3 shows the estimation errors for the seven customers using measures of R^2 and AARE. Overall the estimated demand show reasonable match to the "real" demands, with AARE values ranging from 7.1% to 10.4%. Figure 7 shows the scatter plot of the estimated versus the "real" demands for best and worst performing consumer nodes for the one-week time span. The demands for Junction 11 and 32 are marked as plus signs and crosses, respectively. From the figure, the estimated demands generally matched the real demands, but the estimates for the high-demand hours are not as accurate as those for low-demand hours. Noticeably, the high demands at Junction 11 (greater than 200 GPM) are mostly underestimated. A possible explanation is that Junction

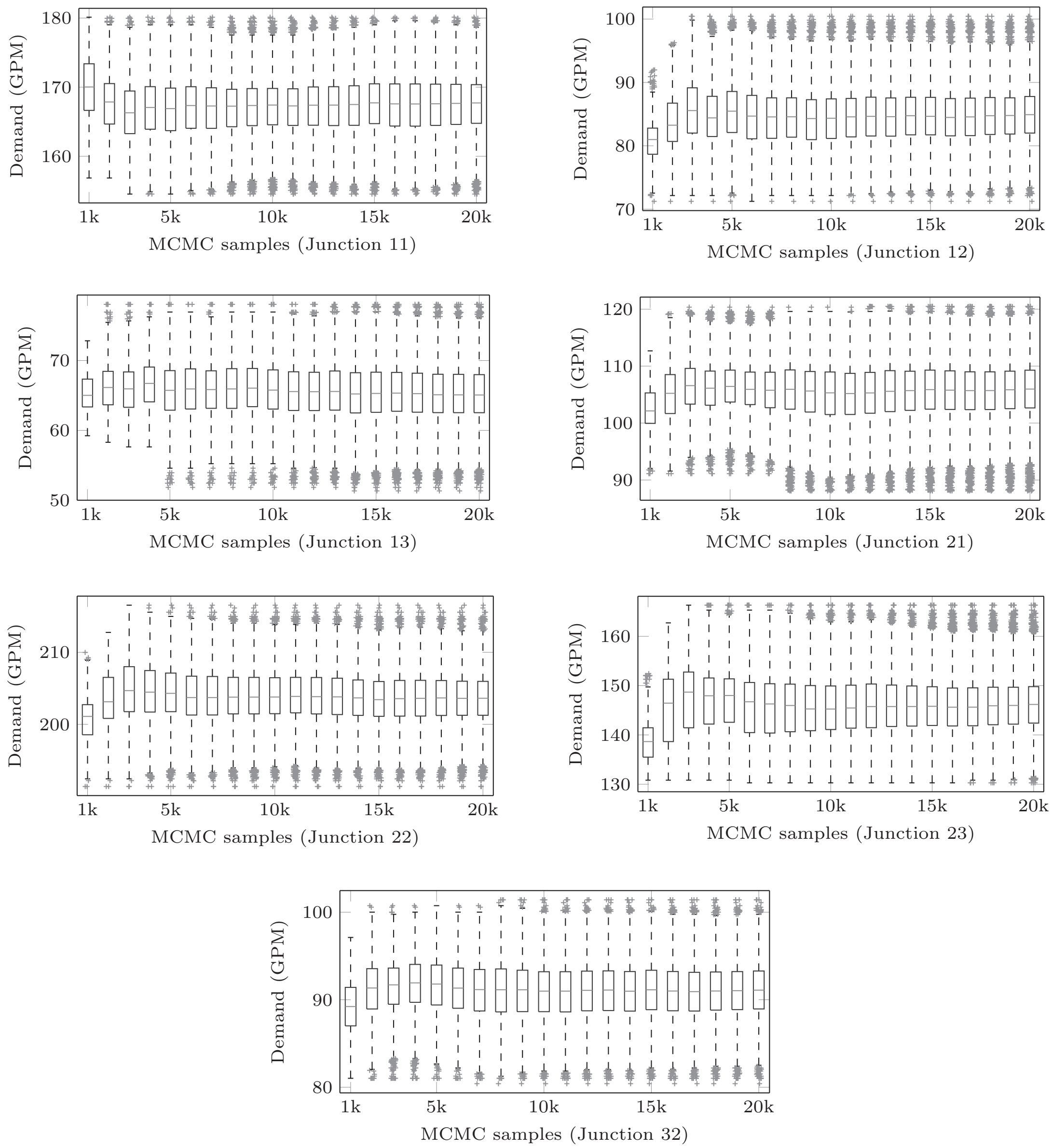


Figure 5: Distribution of the samples generated by the MCMC algorithm versus the chain size for all seven of the consumer nodes; the MCMC samples include those in the burn-in period.

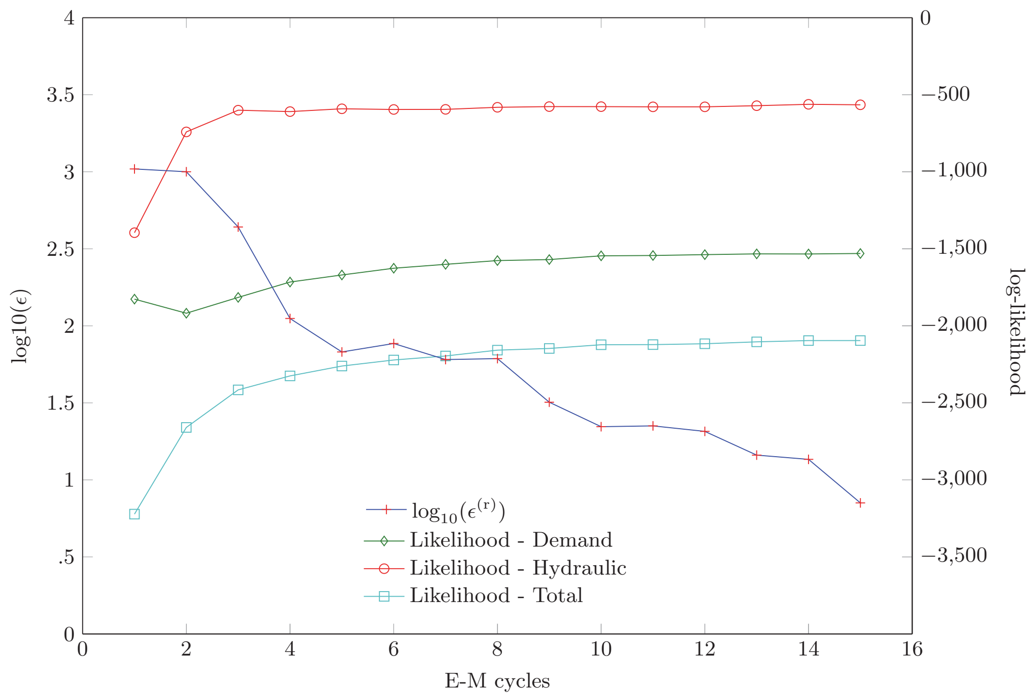


Figure 6: Convergence of the parameters during EM cycles

11 does not have pressure nor flow rate sensors, and the changes in high demands at Junction 11 will only yields limited impacts to the sensor readings located elsewhere. Therefore, using measurements from these sensors, the E-M algorithm could not provide accurate demand estimates in some hours. Installing more sensors around the area is expected to increase the accuracy of the demand estimates.

3.1.2.4 Temporal Correlations of Demand Estimates. The EM algorithm produced a time series of demands at each consumer node that can be analyzed with respect to the temporal correlation expected to be included in the data. Considering a single customer, the temporal correlations of the time series are studied by plotting the autocorrelation (AC) function. A

Table 3: Errors associated with the estimated water demands.

Customer	R^2	AARE*
Junc. 11	0.88	10.4%
Junc. 12	0.92	8.4%
Junc. 13	0.93	7.1%
Junc. 21	0.91	7.3%
Junc. 22	0.91	8.0%
Junc. 23	0.93	8.1%
Junc. 32	0.94	7.1%

*AARE = Average Absolute Relative Error

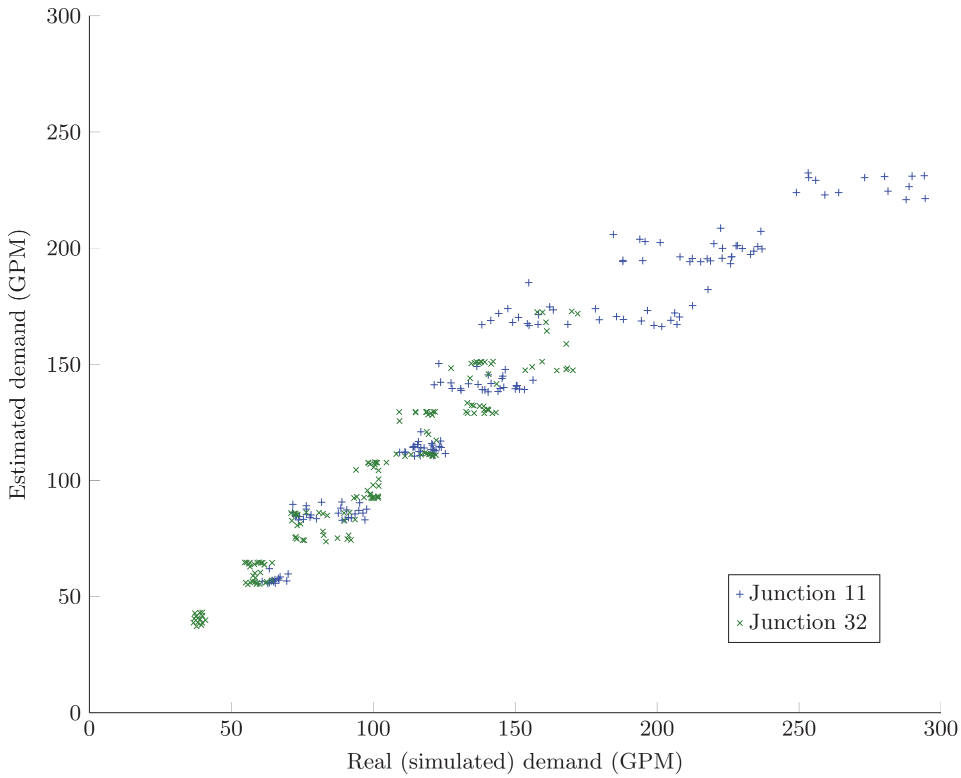


Figure 7: Comparison of estimated demands and "real" demands

lag-L autocorrelation coefficient is defined as

$$AC_L = \frac{\sum_j \left(x_j^{(i)} - \bar{x}^{(i)}\right)\left(x_{j+L}^{(i)} - \bar{x}^{(i)}\right)}{\sum_j \left(x_j^{(i)} - \bar{x}^{(i)}\right)^2} \tag{3}$$

in which $x_j^{(i)}$ is the demand for the i-th customers in time step j, $\bar{x}^{(i)}$ is the mean demand for the i-th customer. Figure 8 are the time series and autocorrelation plots of the demand estimates at Junction 11. The time series plot on the top shows that the estimates follow a general diurnal pattern with random deviations. The autocorrelation (AC) plot at the bottom shows that the series has both strong short-term (1- to 4-hour) correlations and strong periodic (24-hour, 48-hour, etc.) correlations. The auto-correlation gradually decays with longer lag-times. The analysis on the other customers show similar results. The results are also consistent with our previous study on uni-variate aggregated water demands (Chen and Boccelli, 2016) in which the same two types of correlations are identified. For the multi-variate time series introduced in this study, the temporal correlations presents important information that can be exploited for predicting water demands in future studies.

3.1.2.5 Spatial Correlations of Demand Estimates. The demand estimates generated by the EM algorithm were also employed to study the spatial correlations. The z-transformation is performed on the vectors of demand estimates to normalize the data, and the results are illustrated in Figure 9 using 1-d and 2-d histograms.

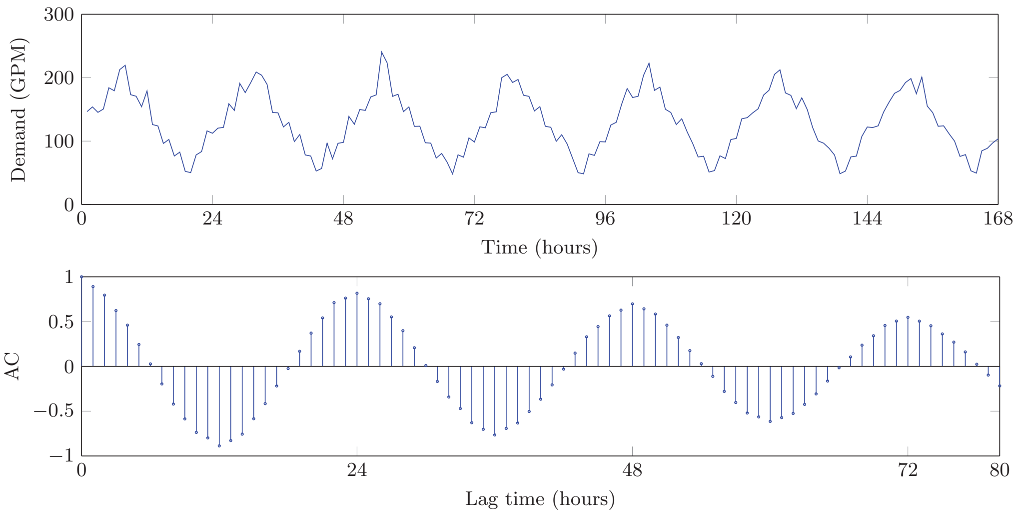


Figure 8: Time series plot and autocorrelation plot of demand estimates at Junction 32

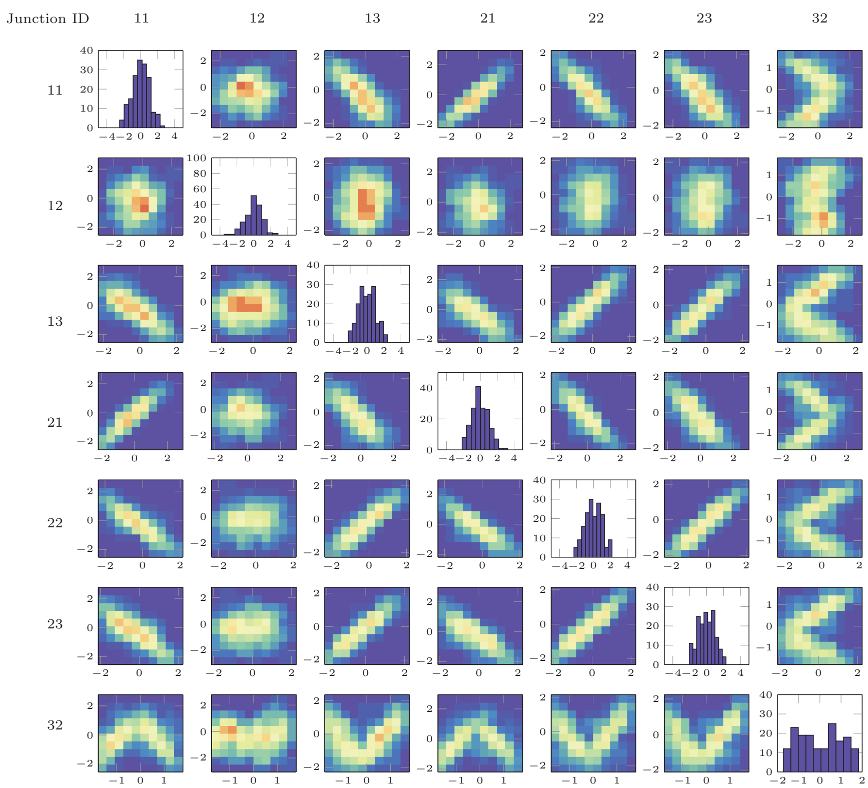


Figure 9: Spatial correlations of demand estimates

In Figure 9, the seven diagonal plots are (1-D) histograms of water demands for the seven customers. The X-axes are hour-in-a-day mean-adjusted and normalized water demands and the Y-axes are frequencies. From the plots, most demand estimates show centered distribution except for Junction 32. The off-diagonal subplots in Figure 9 present the correlations between different customers. Each plot is a 2-D histogram in which X- and Y-axes are demands for the pair of customers, and the colors represent frequencies of data points falling in the 2-D bins. The plots show that the 1st and 4th customers are positively correlated; the 3rd, 5th, and 6th customers are also mutually positively correlated. However, the customers across the two groups are negatively correlated with each other. From the network map, the 1-4/3-5-6 grouping is consistent with the network adjacency, which seems to suggest that customers can be roughly grouped in terms of their topological proximity. Moreover, the 2nd customer is relatively less correlated with any other customers. For the 7th customer, the estimates around the two peaks in the distribution show opposite types of correlations. The estimates in the lower part are positively correlated with the 1-4 group while the estimates in the higher part positively correlated to the 3-5-6 group. This result suggests that the characteristics of spatial correlations for Junction 32 vary with different hours in a week. The structure shown in Figure 9 reveals how the estimates of water demands are correlated under the given layout of customers and SCADA sensors.

3.2 Spatial Aggregation Based on Water Quality Characteristics

3.2.1 *Case Studies.* The proposed clustering algorithm was applied to two network examples, one small example network and one large real-world network.

Figure 10 shows the small network for Case Study 1, which is EPANET Example Network 3, that includes two sources (Lake and River). This network consists of 92 nodes, 3 tanks, 2 reservoirs, and 2 pumps that operate periodically. The three tanks float on the system, meaning that the flows into and out of that tanks are dependent on the source flow rates (inflows) and total system demand (outflows). For Case Study 1, a simulation lasting for 72 hours was applied to the system, and the last 24 hours concentration output collected to calculate the impact coefficients. The source species selected for each node is a conservative input, of which the concentration was set as 100 mg/L.

Figure 11 shows the real-world network model to be used as Case Study 2 that includes 12000 individual nodes (each represents, on average, eight service connections), three sources and two tanks. Two of the sources are the main feeds to the northern and southern portions of the system; the third source feeds a small portion of the network located in the east-northeast section of the network. For Case Study 2, a simulation lasting for 360 hours was applied to the system with the last 24 hours of output concentrations collected to calculate the impact coefficients. The source species selected for each node was assumed to be a conservative input with a concentration of 100 mg/L.

3.2.2 *Results and Discussion.* In this section, the results associated with the proposed clustering algorithm will be presented with an emphasis on the ability of the algorithm to cluster similar nodes and generate distinctly different clusters.

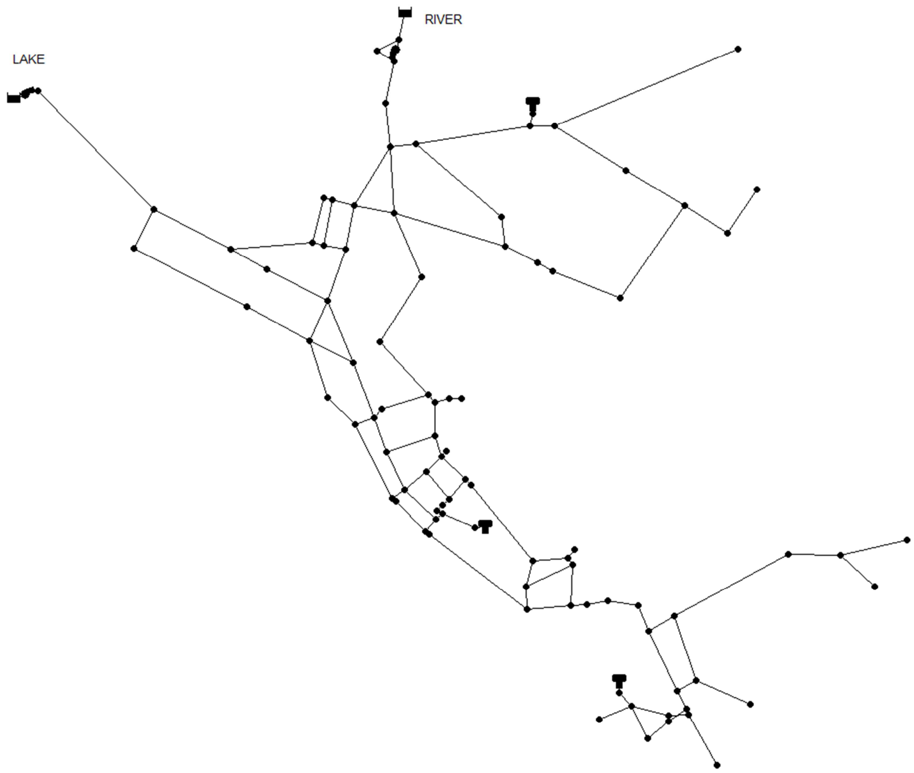


Figure 10: Small test network for the evaluation of the proposed clustering algorithm

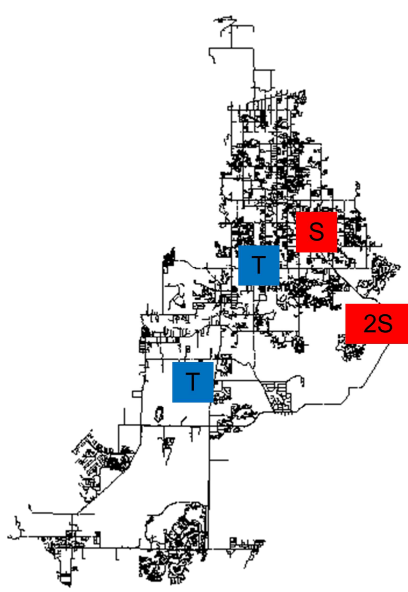


Figure 11: Real-world network model include the sources (S) and tanks (T)

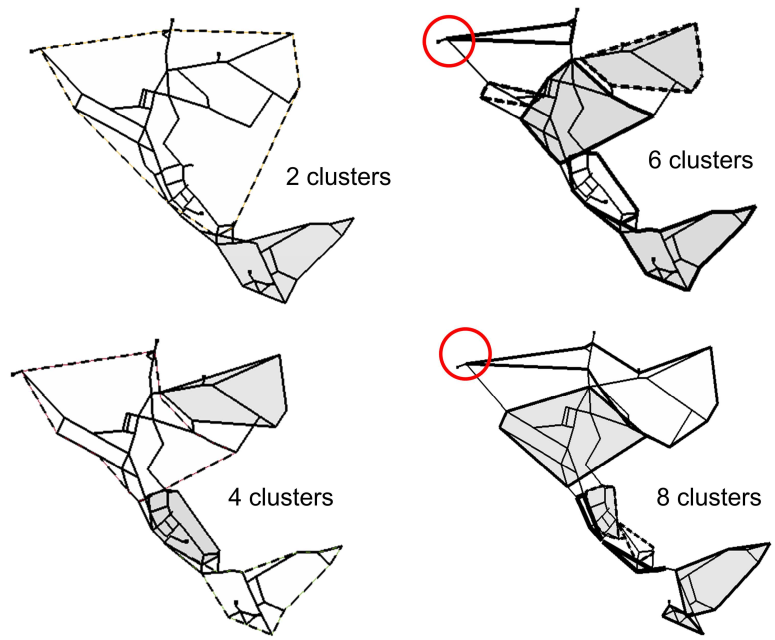


Figure 12: Results associated with clustering the small network; the node in the circle represents a zero-demand node that resulted in an outlier from the other clusters.

3.2.2.1 Case Study 1. Figure 12 illustrates the results when the network was divided into 2, 4, 6 and 8 clusters. By simple observation, the resulting clusters were spatially grouped into non-overlapping clusters by the similarity in the hydraulic flow paths. One outlier occurred (within the clusters of 6 and 8) in the northwest portion of the network (Node 10), which was a zero demand node connected to a pump that only operated intermittently resulting in periods of no flow passing that location. As a result, the resulting flow path for this node was significantly different than the nodes immediately downstream of that location and could be omitted as not meaningful.

To assess the similarity among nodes within a given cluster, water quality simulations were performed to allow the signals at the individual nodes to be evaluated. The hydraulic and water quality simulations were performed with a 504-hour duration and a water quality species entering the system at the two sources modeled as a first-order decay process (decay rate of -0.2s^{-1}). Figure 13 shows a plot of the source concentrations, which were applied to the sources. The reason for injecting the source water quality species was to assess the clustering algorithm as nodes within clusters should have similar concentration patterns. To evaluate the clustering algorithm, the water quality concentrations from each node in the network were collected at hourly intervals from hours 408 to 504. If the hydraulic paths among nodes within each cluster were truly similar, the expectation was that the concentration time series among the nodes within in each cluster would also be similar.

Using the results associated with separating the network into four clusters as an example, Figure 14 shows the concentration time series for each node within each cluster, the number of nodes within each cluster, and the variance of the means and standard deviations from each node in the cluster. By visual inspection, the nodes within each cluster are generally

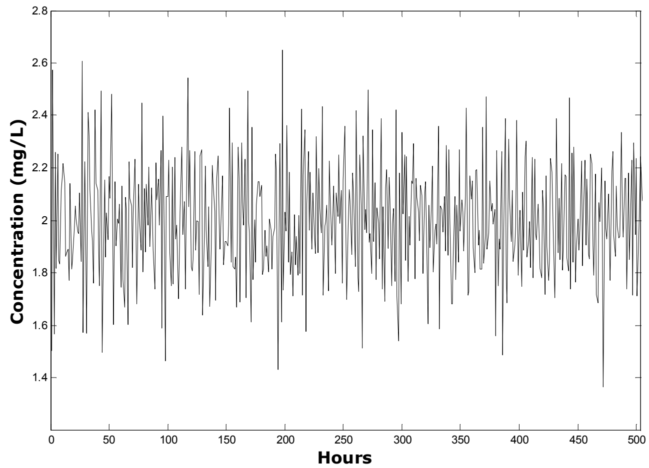


Figure 13: Influent concentrations applied to the two sources of the small test network.

observed to be similar to each other with common patterns in the time series dynamics. To better assess the similarities in water quality signals within each cluster, statistical measures, such as the mean and standard deviation, of the concentration signals were calculated for each node and used to compare similarities within and across clusters.

For assessing the similarities within a cluster, the expectation was that as the number of clusters increased the similarity of the water quality signals within each cluster would become more similar. Thus, increasing the number of clusters should also reduce the variability in the intra-cluster means and standard deviations. To test this hypothesis, the mean and standard deviation of the water quality signals from each node with every cluster was calculated; these data were used to estimate the variability of the means and standard deviations from each node within every cluster. Figure 15 summarizes these results illustrating the variance in the intra-cluster means and standard deviations, respectively, as box-and-whisker plots for clusters ranging from 2 to 15. The number of samples associated with each box-and-whisker plot is simply the number of clusters. The results from the intra-cluster analysis show that both intra-cluster variability of the means and standard deviations (Figure 15) of the water quality signals generally decreased as the number of clusters increased. However, an outlier appeared in both the intra-cluster variability of the means and standard deviations as the number of clusters were greater than 10. This outlier was related to one cluster, and the nodes in this cluster are located in the southern corner of network closest to the southern tank (when the number of clusters between 11 to 13 were analyzed, there were 6 nodes in this outlier cluster; when the number of clusters was greater than 13, there were 7 nodes in this outlier cluster; these nodes are circled in Figure 16). Figure 16 shows the concentration curves related to this outlier cluster when the total number of clusters to be analyzed was 12. The bold curve represents node 243, which is the source of the "outlier" in the clustering results. The reason why this cluster behaves as an outlier is that concentration of the node 243 (around 1.3 mg/L; Figure 16) is obviously lower than the concentrations of the other nodes within the cluster (around 1.8 mg/L). As this node is located at the end of the network and the demand is relatively low (around 6 GPM compared with around 20 to 50 GPM of the other dead-end nodes), the hydraulic residence time was considerably longer relative to

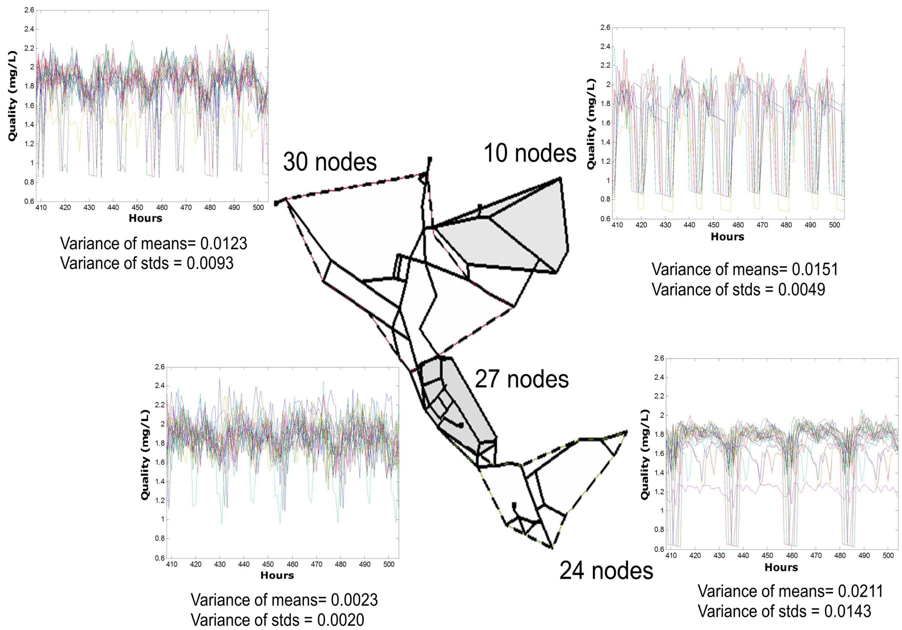


Figure 14: Plots of concentration versus time for the all nodes with the four clusters developed for the small test network.

the other nodes within the cluster. Thus, even though the path to get to all of these nodes was essentially the same, the longer residence time associated with Node 243 resulted in a significantly lower concentration leading to the larger variances in the means and standard deviations of the water quality signals within this cluster that caused the apparent outlier in Figure 15. Overall, the general trends in the intra-cluster analysis support the notion that as the number of clusters increased, the similarity between the nodes within each cluster became increasingly similar.

In addition to assessing the statistics within each cluster, the performance of the clustering algorithm can also be assessed by evaluating the statistics across the clusters. Thus, another approach for demonstrating the performance of the clustering algorithm was to evaluate the inter-cluster variability of the means of the concentration for nodes within the clusters. For the inter-cluster variability of the means, if the clustering algorithm truly separated the nodes, then the variability of the cluster means of concentrations should become greater as the number of clusters increased. Thus, for each cluster the overall mean of the concentrations from all of the nodes was calculated with the variability of those means representative of the differences between the clusters. Figure 17 presents the inter-cluster variability of the means and illustrates that as the number of clusters increased the variability in the cluster means became larger. These results suggest that the clusters were becoming more distinct with large variations in the inter-cluster variance as the number of clusters becomes very large. For the results with the large number of clusters, there are more clusters with only one node, which affects the inter-cluster variance.

3.2.2.2 Case Study 2. For the large network, the clustering results for two clusters are shown in Figure 18. While this network was successfully clustered into two groups, there appear to be overlapping area in the image. These regions are actually distinct and are an artifact of

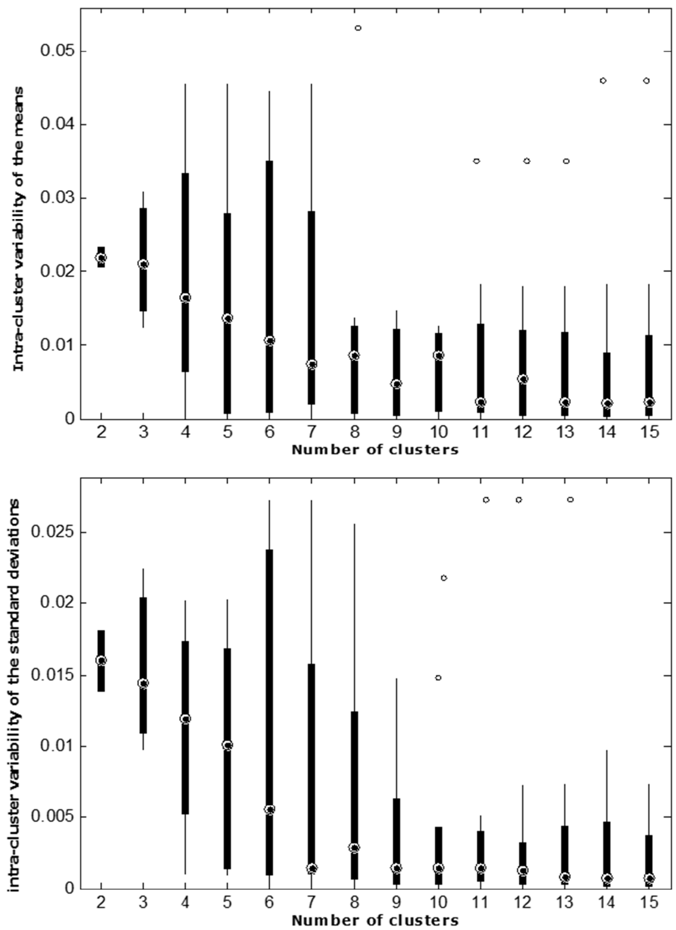


Figure 15: Box-and-whisker plots of the intra-cluster variability of the means and standard deviations for the small test network.

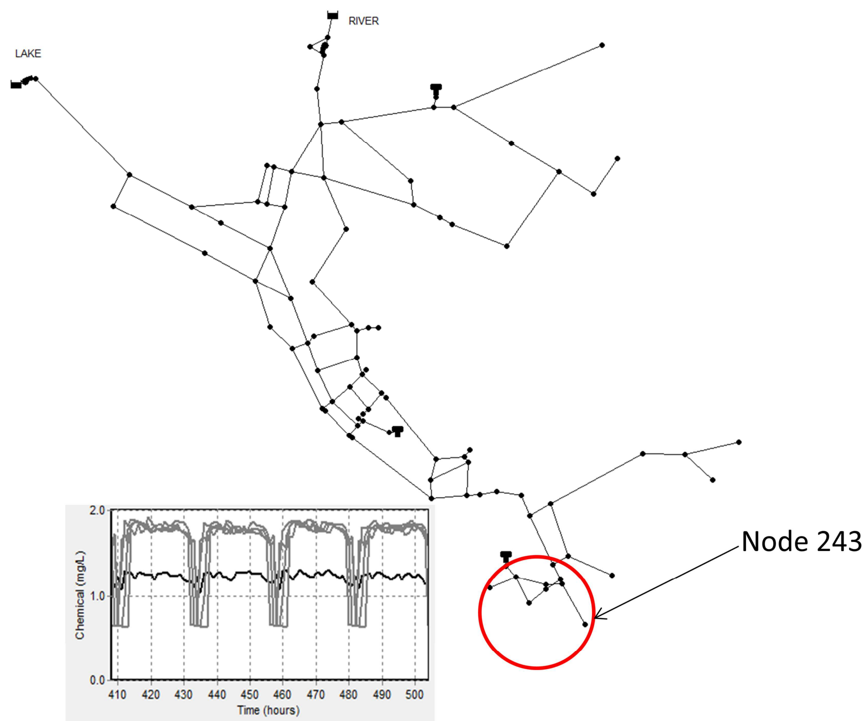


Figure 16: Location and concentration plots of the outlier cluster when the small test network is separated into 12 clusters.

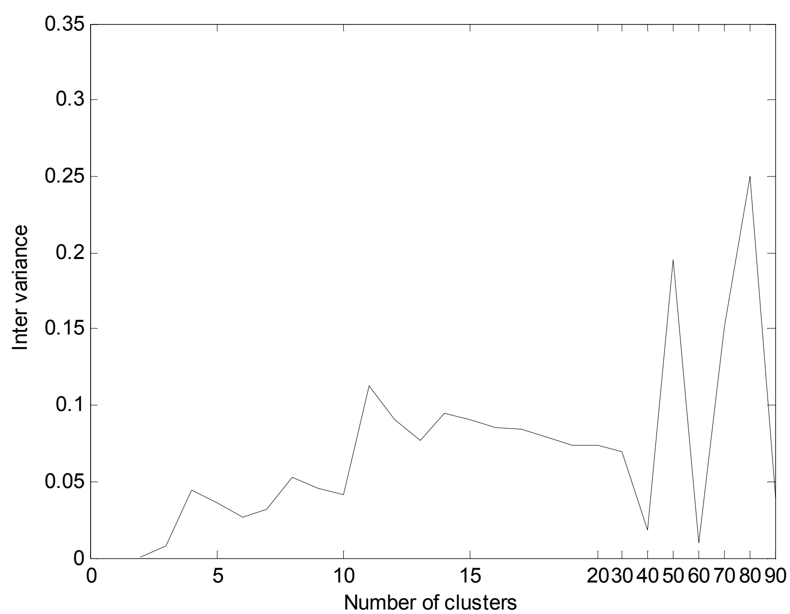


Figure 17: Inter-cluster variance of the mean concentration for an increasing number of clusters.

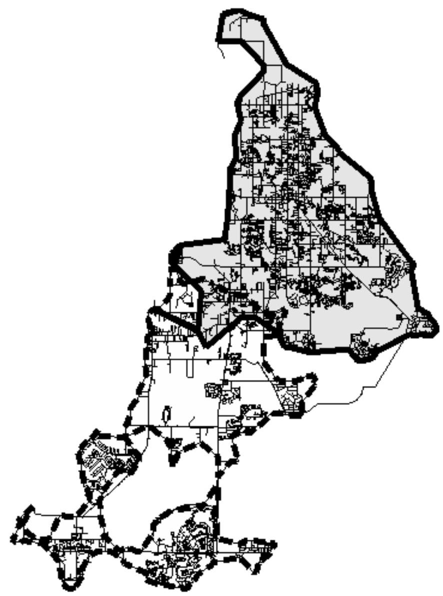


Figure 18: Separation of the realistic network into two clusters.

the current approach used to visualize the clusters.

To assess the similarity of hydraulic paths among nodes from the clustering algorithm, a water quality simulation was performed with a source concentration of 400 mg/L. The three water treatment plants were selected as sources for chemical injections. The hourly water quality concentrations were collected from hours 264 to 360. The variances in the intra-cluster means and standard deviations for clusters of 2, 4, 6, 8, 10, 20, 30, 40, and 50 are shown in Figures 19 and 20, respectively. From the results, the clusters of the large network lead to more similar clusters based on the similarity of hydraulic paths when clusters move towards finer ones, especially when number of clusters step into range exceeding 10 clusters.

Similar to inter-cluster variability analysis in Case Study 1, the same method was applied to the clustering results from this case study. Figure 21 presents the inter-cluster variability of the means, and the variability of the mean concentrations that demonstrate that both values increased as the number of clusters increased. These results suggest that the clusters continue to become more distinct as the number of clusters increased.

3.3 Significance

The results of this study are significant for two primary reasons. First, the development of the composite demand-hydraulic model was shown capable of estimating the demands and parameters of a time series model using limited hydraulic information. This result is the first approach to link an actual demand model to a network hydraulic model that will allow not only for demand estimation but demand forecasting to be performed. The latter of which will allow real-time decision making possible. Second, the proposed clustering algorithm was shown capable of grouping nodes based on similarities in water quality. This ability to group nodes will provide opportunities to reduce the scale of network demand estimation

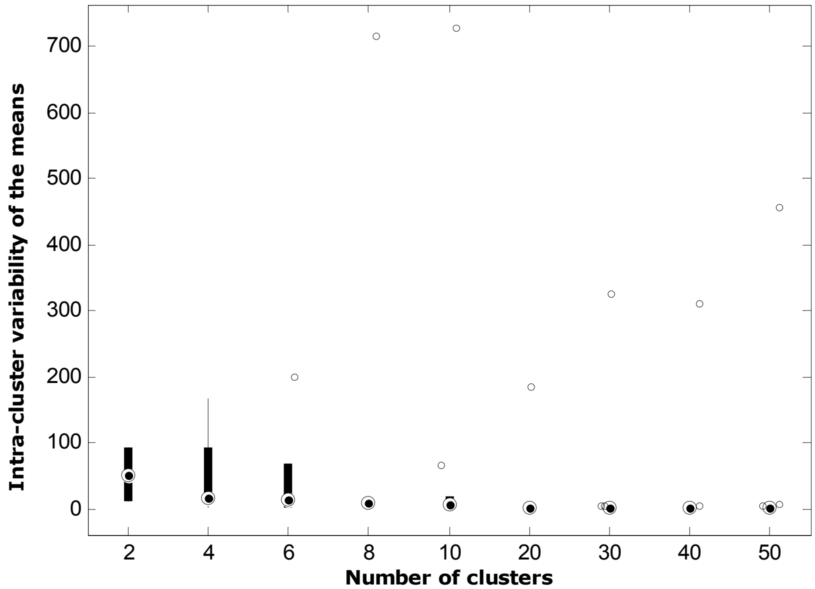


Figure 19: Box-and-whisker plots of the intra-cluster variability of the means.

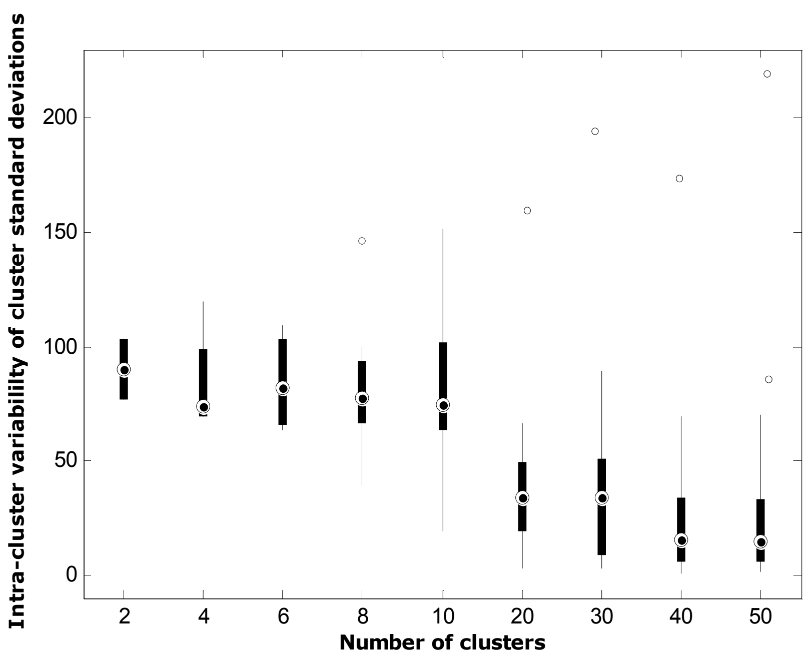


Figure 20: Box-and-whisker plots of the intra-cluster variability of the standard deviations.

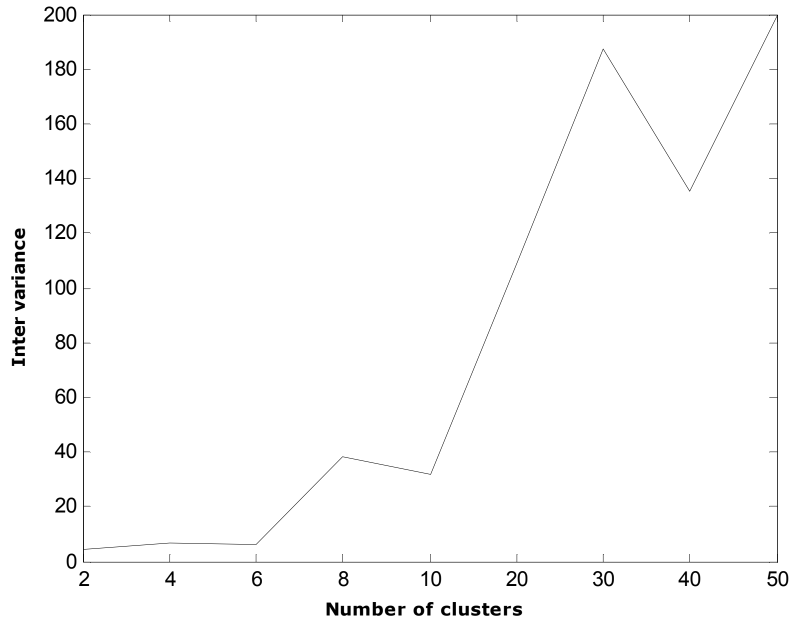


Figure 21: Inter-cluster variance of the mean concentration for an increasing numbers of clusters.

problems. That is, the clustering algorithm provides the capability to effectively reduce the scale of the demand estimation problem for realistic networks (e.g., Figure 18) to the scale of smaller network (e.g., Figure 4). Additionally, the clustering approach presented allows the grouping of nodes with similar water quality characteristics that can also help to reduce the problem scale of other applications such as locating sensors for contaminant warning systems or identifying regulatory sampling locations.

4 References

Box, G. E. P. and G. M. Jenkins (1976). *Time Series Analysis: Forecasting and Control*. Holden-Day series in time series analysis and digital processing. Oakland, Calif.: Holden-Day, Inc.

Brooks, S., A. Gelman, G. L. Jones, and X.-L. Meng (Eds.) (2011). *Handbook of Markov chain Monte Carlo*. Chapman & Hall/CRC.

Chen, J. and D. L. Boccelli (2013). Demand forecasting for water distribution systems. In *Proceedings, Computing and Control in the Water Industry*, Perugia, Italy.

Chen, J. and D. L. Boccelli (2016). Forecasting hourly water demands with seasonal autoregressive models for real-time application. *Water Resources Research*. in revision.

Dempster, A., N. Laird, and D. Rubin (1977). Maximum likelihood from incomplete data via the EM algorithm. *Journal of the Royal Statistical Society. Series B (Methodological)*, 1–38.

Ghahramani, Z. (1998). Learning dynamic Bayesian networks. *Adaptive Processing of Sequences and Data Structures*, 168–197.

Gilks, W., S. Richardson, and D. Spiegelhalter (1995). *Markov Chain Monte Carlo in Practice: Interdisciplinary Statistics*, Volume 2. Chapman & Hall/CRC.

Koller, D. and N. Friedman (2009). *Probabilistic Graphical Models: Principles and Techniques*. The MIT Press.

Laloy, E. and J. Vrugt (2012). High-dimensional posterior exploration of hydrologic models using multiple-try dream (zs) and high-performance computing. *Water Resources Research 48*(1), W01526.

Larose, D. T. (2005). *Discovering Knowledge in Data: An Introduction to Data Mining*. Hoboken, NJ: Wiley-Interscience.

Metropolis, N., A. Rosenbluth, M. Rosenbluth, A. Teller, and E. Teller (1953). Equation of state calculations by fast computing machines. *The Journal of Chemical Physics 21*, 1087.

Pasula, H., S. Russell, M. Ostland, and Y. Ritov (1999). Tracking many objects with many sensors. In *International Joint Conference on Artificial Intelligence*, Volume 16, pp. 1160–1171.

Qin, T. and D. L. Boccelli (2015). Grouping water demand nodes by similarity among flow paths in water distribution systems. In *Proceedings of the World Water and Environmental Resources Congress*, Austin, TX. ASCE (accepted).

Robert, C. (2007). *The Bayesian Choice: From Decision-Theoretic Foundations to Computational Implementation*. Springer Science+Business Media, LLC.

Rossman, L. A. (2000). *EPANET2 User's Manual*. Cincinnati, OH: Risk Reduction Engineering Laboratory, U. S. Environmental Protection Agency.

Shang, F., J. G. Uber, and M. M. Polycarpou (2002). Particle backtracking algorithm for water distribution system analysis. *Journal of Environmental Engineering 128*(5), 441–450.

van Thienen, P. and D. Vries (2013). Probabilistic backtracing of drinking water contamination events in a stochastic world. In *Proceedings, Computing and Control in the Water Industry*, Perugia, Italy.

Wei, W. W. S. (2006). *Time Series Analysis: Univariate and Multivariate Methods* (2^{nd} ed.). Pearson Education, Inc.

Yang, X. and D. L. Boccelli (2013). A simulation study to evaluate temporal aggregation and variability of stochastic water demands on distribution system hydraulics and transport. *Journal of Water Resources Planning and Management*. doi:10.1061/(ASCE)WR.1943-5452.0000359.

Information Transfer Program Introduction

The Ohio WRC conducted a number of activities designed to transfer water related information to a wide range audience throughout Ohio, including state, federal, county, and municipal agencies, as well as to the academic community of researchers and students. In addition, many of our efforts target non-professional audiences including children, and to private citizens. The Ohio WRC conducted information transfer by (1) promoting center activities, researchers, and research projects via newsletters, the Ohio WRC website, email correspondence, brochures, booths at conferences, personal meetings with water professionals and agencies representatives; (2) organizing, sponsoring, and participating in workshops, seminars, guest lectures and conferences; (3) serving and volunteering in various water organizations and their advisory boards such as the Water Management Association of Ohio, Ohio Water Resources Council and Friends of Lower Olentangy Watershed NGO; and (4) sponsoring two information transfer projects – Dr. Hoornbeek's and Dr. Bohrerova's. Specific activities included:

1)Promoting Ohio WRC research, results of projects, and investigators a)Preparation of Ohio WRC website content (wrc.osu.edu), website updates of events and news, and general maintenance of website. We had over 3,500 website hits, the majority of which came from new visitors. b)Preparing one page summaries of research projects, including the importance of the research topic for the State, relevant outcomes and results, and investigator background. These summaries were distributed to our Advisory Board members and other stakeholders. c)Publishing research project summaries in the Ohio Water Table, a quarterly newsletter published by the Water Management Association of Ohio (WMAO). During the reporting period, the highlighted researchers and projects were: Dr. Buffam's project #2013OH297B, Dr. Jaeger and Sullivan's project 2014OH327B, Dr. Mouser's project funded by other sources and Dr. Sharma's project 2014OH312B. This newsletter is distributed to about 575 people and organizations in Ohio in the water resources field from private sector (33%), universities (8%), nonprofit/citizens (17%) and federal, state and local government agencies (42%) d)Preparing and publishing an Ohio WRC brochure, and banners highlighting the annual activities and projects of the Ohio WRC. These are distributed at various events and presented at the Ohio WMAO conference. e)Responding to questions from public regarding water resources issues in the State of Ohio. f)Maintaining and updating statewide database of investigators in Ohio universities with research interests related to water. Currently, the database contains around 250 researchers from 15 different Ohio Universities. g)Meeting with the Ohio WRC Advisory Board Members – once a year - discussing Center direction, requests for proposals, current research and results dissemination, and draft of strategic plan of the Ohio Water Resources Center. h)Meeting with Ohio Congress and Senate members' office staff to discuss Ohio WRC activities, research results, and their impact for the State.

2)Organizing and sponsoring information transfer events a)Co-organized quarterly Ohio WRC-WMAO luncheon seminars, which includes assisting with luncheon administration and securing speakers. This past year the four luncheons were attended by approximately 140 water professionals from government, academia, NGOs and industry. The speakers and topics in this reporting period were: Elizabeth Toman (OSU): "Unpaved rural roads and stream water quality"; Aurea L. Rivera, (Imagineering Results Analysis Corporation): "The Legion of Bloom: Algae, Remote Sensing and Lake Erie"; Alan Hamlet (University of Notre Dame): "Developing a Comprehensive Hydroclimatic Database for the Midwest and Great Lakes Region: 1915-2100"; Theodore "Ted" Lozier and John Watkins (Muskingum Conservancy District): "Water Supply and the Oil & Gas Industry's Impact within the Muskingum Watershed Conservancy District". b)Sponsored 44th Annual Water Management Association of Ohio (WMAO) conference titled: "MOVING THE NEEDLE: Policies, Programs, and people that Drive Change". In 2016 around 250 professionals attended the conference, including academic researchers, students, representatives of State and Federal Agencies, industry and NGO's. The conference is attracting increasing amount of academic researchers, including Ohio WRC researchers, based on our promotion of the conference. We also helped with selecting the student candidate for WMAO award, talked to students during the "Careers in Water Resources" session, and set up a booth at the conference to discuss Center activities. c)Guest lecture for the “Seminar on Sustainability” at the Ohio State University, talking about water and sustainability with a focus on urban water infrastructure and treatment. The seminar is attended by approximately 25 engineering undergraduate students each semester. d)Organizing and leading a 25 minute, hands-on workshop for 5th grade students on principles of buoyancy in the 2015 Central Ohio Childrens Water Festival

3)Serving in multiple water organizations a)Serving on Water Management Association of Ohio (WMAO) board as a Director of Research and Data Management. In this role, we focus on promoting water resources research in the State, and attend bimonthly meetings. b)Member of WMAO student awards committee – evaluating student proposals and deciding the best candidate for the award. c)National Institute of Water Resources Regional Representatives of the Great Lakes Region. d)Participating in quarterly meetings of the Ohio Water Resources Council meetings, forum for collaboration and coordination among state agencies – helping with strategic planning of the Council for FY2015 – 2018 e)Meeting with representatives of Ohio Sea Grant and Ohio EPA, drinking and groundwater division – discussing plans how academic researchers can help State agencies during emergency events f)Serving on Friend of Lower Olentangy Watershed (FLOW) NGO Science committee, helping organize events, write outreach and education proposals. g)Part of OSU Discovery Themes effort – institution wide strategic planning efforts for the University in teaching, research and engagement.

4)Information Transfer Projects Dr. Hoornbeek’s project (2015OH445B) summarizes the current policy used in Ohio for nutrient management and compares Ohio’s approach to two other regions that have nutrient problems. The project summary is part of this report and will be a useful tool for policy makers in Ohio and other organizations dealing with nutrient issues.

Dr. Bohrerova’s project (conducted by Ohio WRC but funded by other funds, 2015OH482O) titled “Adopt Your Waterway” focuses on citizen volunteer lead monitoring of streams in urbanized areas around Columbus, OH for water chemistry and macroinvertebrates. The goal is to educate public about stream health and support water stewards in the area.

Policy Tools for Reducing Nutrient Loads and Combating Harmful Algal Blooms (HABs) in Lake Erie: An Inventory and Assessment

Basic Information

Title:	Policy Tools for Reducing Nutrient Loads and Combating Harmful Algal Blooms (HABs) in Lake Erie: An Inventory and Assessment
Project Number:	2015OH445B
Start Date:	3/1/2015
End Date:	2/28/2016
Funding Source:	104B
Congressional District:	OH-013
Research Category:	Social Sciences
Focus Category:	Law, Institutions, and Policy, Nutrients, Management and Planning
Descriptors:	None
Principal Investigators:	John A Hoornbeek

Publications

1. Hoornbeek, J., Filla, J., Venkata, A., Kalla, S., and Chiyaka, E. (2015). “A Glimpse of Some Preliminary Findings – Reducing Nutrient Flows to Lake Erie: Current Control Efforts and Ideas from Elsewhere.” Presented at the Water Management Association of Ohio Conference. November 17, 2015. Worthington, Ohio.
2. Hoornbeek, J., Filla, J., Venkata, A., Kalla, S., and Chiyaka, E. (2016). “Policy ‘Tools’ for Water Pollution Control: Addressing Nutrient Enrichment and Harmful Algal Blooms in Lake Erie”. Prepared for Presentation at the Panel on Environmental Policy Instruments and Institutions Midwest Political Science Association. April 8, 2016. Chicago, Illinois.
3. Hoornbeek, J., Filla, J., Venkata, A., Kalla, S., and Chiyaka, E. (2016). “Addressing Harmful Algal Blooms in Lake Erie: Policy Instruments for Reducing Nutrient Loads to Lake Erie from Cities and Rural Areas.” Presented at the Water Resilient Cities Conference held at Cleveland State University. Cleveland, Ohio, April 22, 2016.

MAY 6, 2016

FINAL PROGRESS REPORT: POLICY TOOLS FOR REDUCING HARMFUL ALGAL BLOOMS

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330-672-7148

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Cover Photo: Image of Lake Erie via NASA's MODIS Terra and Aqua Instruments

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Problem and Research Objectives

Nutrient Enrichment and Harmful Algal Blooms in Lake Erie: A Description of the Problem

In recent years, Lake Erie has been experiencing symptoms of eutrophication resulting from excess flows of nutrients from agricultural and urban sources. One of the consequences of this nutrient enrichment is the occurrence of large Harmful Algal Blooms (HABs) in Lake Erie – particularly in its western basin. A HAB is any large increased density of algae that is capable of producing toxins (Ohio Sea Grant, 2011). These blooms have received significant attention in recent years, with a (formerly) record-breaking bloom occurring in Lake Erie in 2011. That algal bloom extended from the western basin of Lake Erie near Toledo to the central Lake Erie Basin past the City of Cleveland. In 2014, toxins from a HAB near Toledo were detected in the city's public water system, which uses Lake Erie to supply water for several hundred thousand people in the Toledo area. This contamination resulted in a ban on the use of water from the city's public water system. Yet another record-breaking HAB spread across Lake Erie in 2015 (Associated Press, 2015). In addition to HABs, excess nutrients have also resulted in anoxic zones within the lake, and nuisance levels of Cladophora[1] (Great Lakes Commission (GLC), 2015).

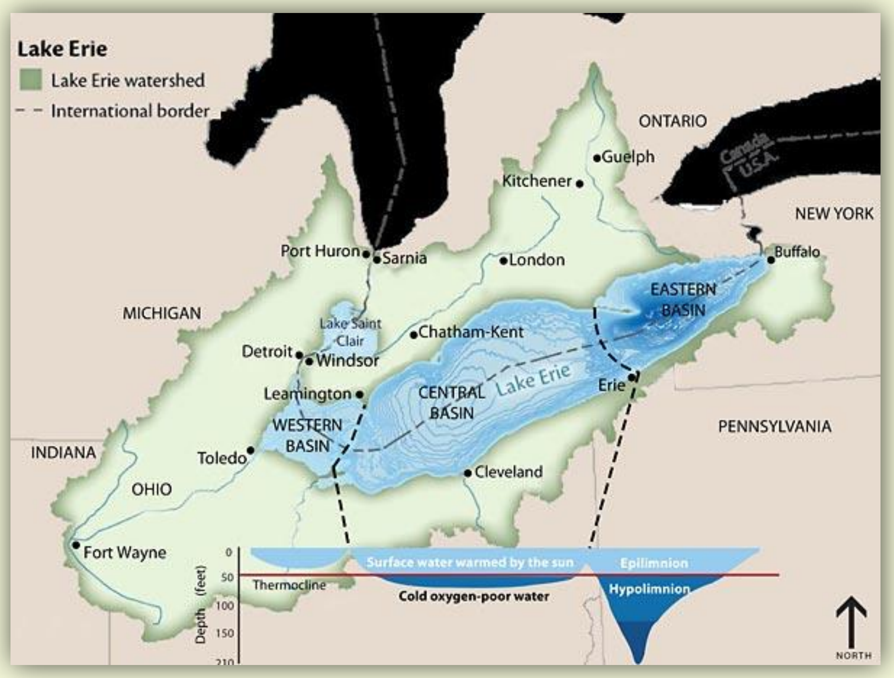
Excess nutrients and HABs result from excess loadings and elevated concentrations of two key nutrients, phosphorus and nitrogen. Both of these nutrients result from non-point sources, such as agricultural operations and urban storm-water, as well as point-sources of nutrient pollution which flow through discrete pipes or conveyances such as wastewater treatment plant outfalls, Combined Sewer Overflows (CSOs) outfalls, and other sources (OSG 2011, IJC 2014, Ohio Phosphorus Task Force 2013, and the GLC, 2015). Existing data and studies suggest that agricultural operations account for the largest share of these excess nutrients, and that phosphorus is a key nutrient of concern for Lake Erie (Phosphorus Task Force, 2013; IJC, 2014; Lucas County Board of Commissioners, 2015). It is also important to recognize that studies have suggested that dissolved phosphorus, as opposed to total phosphorus, may be of particular concern in the creation of HABs in Lake Erie (Phosphorus Task Force, 2013).

HABs can yield a range of negative impacts for human health, the environment, and the economy. These negative impacts include threats to public and ecological health, including toxic effects on human neurological systems, anoxic conditions, and other undesirable effects (Zingone and Enevoldsen, 2000). They also include economic impacts, as HABs may inhibit recreational and other uses of Lake Erie which can reduce the significant economic benefits associated with Lake Erie waters (Austin et al, 2007).

There have been policy responses to the issues of HABs occurring in Lake Erie at the state level in Ohio, and at the federal and international levels. At the state level in Ohio, these responses have included the repurposing of funding from agencies such as Ohio's Environmental Protection Agency (OEPA) and Department of Natural Resources (ODNR) to provide monies to reduce nutrient pollution through the

[1] Cladophora are true algae (unlike the cyanobacteria that create HABs) that can also create large algal blooms. These blooms can be a nuisance and also cause environmental problems. However, they do not produce toxins associated with HABs (OSU Sea Grant, 2011)

Ohio Clean Lakes Initiative and other programs (Clean Lakes Initiative, 2014). The state has also developed new legislation and regulations to reduce the use of agricultural fertilizer during winter months and to further investigate nutrient discharges from certain POTWs. In June of 2015, Ohio also signed an agreement with Michigan and the Canadian Province of Ontario to reduce phosphorus loads to Lake Erie by 40% by 2025.



- The Lake Erie Basin (USEPA, 2016)

The federal government has also been active in attempting to reduce nutrient flows to Lake Erie and the other Great Lakes through the Great Lakes Restoration Initiative (GLRI), and other programs funded and/or administered by federal agencies. It has also been involved in ongoing international processes to coordinate nutrient reduction efforts between the US and Canada via the Great Lakes Water Quality Agreement (GLWQA), an agreement that was last updated in 2012. In February of 2016, the U.S. and Canadian Governments made the 40% phosphorus reduction target an official goal of both nations (USEPA, 2016a). Over the next few years, the two countries plan to use agreed upon mechanisms in Annex 4 of the GLWQA to develop loading allocations to meet this targeted level of phosphorus reduction and to instigate Domestic Action Plans in an effort to achieve the needed reductions. These domestic action plans are expected to define steps to be taken to reduce nutrient loads consistent with the loading allocations made through the Annex 4 process.

To our knowledge, there has not yet been any significant and comprehensive effort to take stock of the nutrient reduction strategies and tools currently in place in Ohio's Lake Erie basin[2]. In addition, we have

[2] As we proceeded with this research, we did uncover a Great Lakes Commission (2012) study that reviewed nutrient reduction programs in place in Great Lakes states and provinces, but it did not deal with efforts in the Ohio Lake Erie basin specifically. In

not seen systematic efforts to compare nutrient management programs and efforts across water basin programs in the United States (US). The inventory and assessment developed through this project takes initial steps to address both of these gaps in our current knowledge base.

Research Objectives

This project seeks to inform current nutrient reduction efforts in Ohio and elsewhere. We present an inventory of policy tools being used in the Ohio portion of the Lake Erie basin to reduce nutrient flows to the lake. We also share the results of an effort to identify and review nutrient reduction efforts being carried out by other water basin management programs in the United States (US). In addition, we use our inventory and the information gained from our review of other American water basin programs to offer ideas for policymakers and public administrators to consider regarding additional policy tools they may want to use in addressing excess nutrient enrichment problems in the Lake Erie basin.

Specifically, we have pursued the following research objectives through our work:

- Create an inventory of current nutrient reduction policies being utilized in the Lake Erie water basin in northern Ohio as a result of state and/or federal programmatic efforts, along with key elements of the strategies used to implement them;
- Identify nutrient reduction policies and implementation strategies used by other place-based water quality management programs elsewhere in the country, and collect information relevant to their effectiveness;
- Determine nutrient reduction strategies that appear promising for reducing nutrient loads to Lake Erie from Ohio, based on their success or perceived success in other areas of the country and the potential for them to usefully supplement current policies and strategies being implemented in Ohio;
- Develop lessons learned and recommendations for nutrient policies and strategies to implement in northern Ohio.

- Harmful Algal Bloom (NOAA, 2009)

addition, it did not specifically compare nutrient reduction efforts for Lake Erie with those in place elsewhere in the country. As such, we used it to inform our project efforts.

Methodology

Inventorying Current Policy Efforts to Reduce Nutrient Flows in the Ohio Lake Erie Basin

To develop an inventory of policy efforts focused on nutrient control in the Lake Erie basin of Ohio, we sought to identify and document use of nutrient control efforts in the State of Ohio generally, and in the Lake Erie basin in particular. To do so, we used Hood's policy tools framework (1983) as a guide and sought to identify exercises of government regulatory authorities, expenditures of funds and resources, key nodal communications such as guidance and information provided by governing organizations, and organizational resources and capacities.[3]

We searched for data and information in these areas through multiple searches on Ohio government agency websites and interviews with officials who are knowledgeable regarding nutrient reductions efforts in the state of Ohio. Our efforts took place over a period of approximately one year in duration and included searches of website material posted by the following federal and state agencies: US Environmental Protection Agency (USEPA); US Department of Interior (USDOI); US Department of Commerce – National Oceanographic and Atmospheric Administration (NOAA); US Department of Agriculture (USDA); Ohio Environmental Protection Agency (OEPA); Ohio Department of Natural Resources (ODNR); Ohio Department of Agriculture (ODOA); Ohio Development Services Agency (ODSA); Ohio Public Works Commission (OPWC), and; the Ohio Lake Erie Commission (OLEC).

We also sought out and interviewed multiple state and federal officials who are knowledgeable regarding nutrient control initiatives the Ohio Lake Erie basin. Our interviews with these officials were intended to: 1) identify nutrient reduction efforts we had missed during our web searches, and; 2) clarify our understandings of the written materials we had collected. The interviews conducted included discussions with staff, or former staff, of OEPA, ODNR, ODA, the Lake Erie Commission, ODSA, and USEPA.

- Long Island Sound Study Logo (LISS, 2016)

Investigating Other American Water Basin Programs

To draw lessons from other water basin programs about ways to address nutrient enrichment problems, we sought to identify water basin management programs around the country. We held discussions with USEPA officials and conducted independent research efforts to identify water-basin programs throughout the country. Through these efforts, we identified a total of 32 water basin programs to investigate. Twenty-eight of these water basin programs were part of the USEPA's National Estuary Program (NEP) and four additional programs were place-

[3] In 1983, Christopher Hood's "The Tools of Government" proposed that government can be viewed as a set of resources that define the policy tools that can be used to "detect" what is going on in society and to "effect" societal conditions in ways that are consistent with policy goals. He defined four major resources: 1) "authority"; 2) "treasure"; 3) "nodality", and; 4) "organization". To improve readability, we interpret these categories as "regulatory interventions", "expenditures of funds and resources", "government strategies, plans, and communications", and "organizational resources and capacities".

based programs set up independent of the NEP. We then subjected these 32 basin programs to a three-phase screening review in an effort to identify programs that were likely to yield potentially useful lessons and insights for the Lake Erie watershed. A list of these screened programs is provided in Appendix 1.

During the first phase of the project, we reviewed websites for each of the programs involved -- along with other publicly available information -- to gain a broad understanding of the work they do. More specifically, we assessed: 1) whether or not nutrients were of concern in the water basin; 2) the likely and/or predominant sources of nutrient flows; 3) stakeholders in the process and the number of jurisdictions involved, and; 4) evidence of potentially innovative and/or effective policy or management approaches to nutrient control.

Chesapeake Bay Program
A Watershed Partnership

- Chesapeake Bay Program Logo

In the second stage of the screening process, we sought to identify programs that seemed to have potential to reveal insights for addressing nutrient concerns in Lake Erie. At this stage, we sought to identify promising programs based on the following criteria: 1) did they address phosphorus and/or nitrogen? 2) were there notable agricultural contributions to nutrient flows in the basin? 3) was there evidence of coordinated implementation across jurisdictions? and 4) was there evidence of potentially innovative and/or effective policy or management practices being undertaken? Eight programs that addressed nutrients and scored relatively highly across the three other areas were selected for further investigation.

We then conducted more in-depth reviews of these eight programs, including discussions with program officials where appropriate, to identify one or two programs that we would investigate in greater detail. During the course of these more detailed investigations, we also asked those we were interviewing whether there were other programs or nutrient reduction efforts that they were aware of that would be likely to yield useful insights for the Lake Erie effort. Based on these further investigations, we chose three programs that were making significant efforts to measure the effectiveness of their nutrient control efforts as a whole and appeared likely to yield useful insights for the Lake Erie Basin. They were the Chesapeake Bay Program (CBP), the Long Island Sound Study (LISS) Program, and the Tampa Bay Estuary Program (TBEP).

We then proceeded to investigate these programs and the policy instruments they used in greater detail. These investigations involved additional interviews with program staff(s) and deeper research into secondary information sources. Throughout the process of investigating these water basin programs, we inventoried nutrient control policies and management strategies with particular focus on approaches that we did not see being implemented in the Ohio Lake Erie basin.

Limitations

Like most research efforts, this project is characterized by limitations that affect both the data and information compiled and the conclusions reached. First, time and resource limitations -- as well as our reliance on publicly available documents and interviews -- mean that we cannot guarantee that we have identified *all* current nutrient reduction efforts in the Ohio Lake Erie basin. However, we did spend a good bit of time collecting data and information, so we believe that we were able to identify most – if not all – significant public sector nutrient reduction efforts being undertaken in the Ohio Lake Erie basin area.[4]

- Tampa Bay Estuary Program Logo (TBEP, 2016)

Second, while other water basin management programs we investigated all focused on nutrients, they were different than the Ohio Lake Erie basin efforts in other respects. The watersheds differed in size, so policy tools which vary in effectiveness or utility based on size might not transfer well to water basins with differing size characteristics. The watershed basin programs we investigated also differed in the nature and extent of cross-jurisdictional work that is required to coordinate nutrient reduction efforts

Third, while we set out with the hope of identifying information on the measured effectiveness of policy tools used in other watersheds, we did not find this kind of information available. We did, however, find watershed programs that were making substantial efforts to measure their *overall* progress against defined nutrient loading criteria and nutrient-related ambient water quality goals, so we chose to focus attention on those efforts in order to enable relevant learning to inform the potential development of similar efforts in the Lake Erie basin.

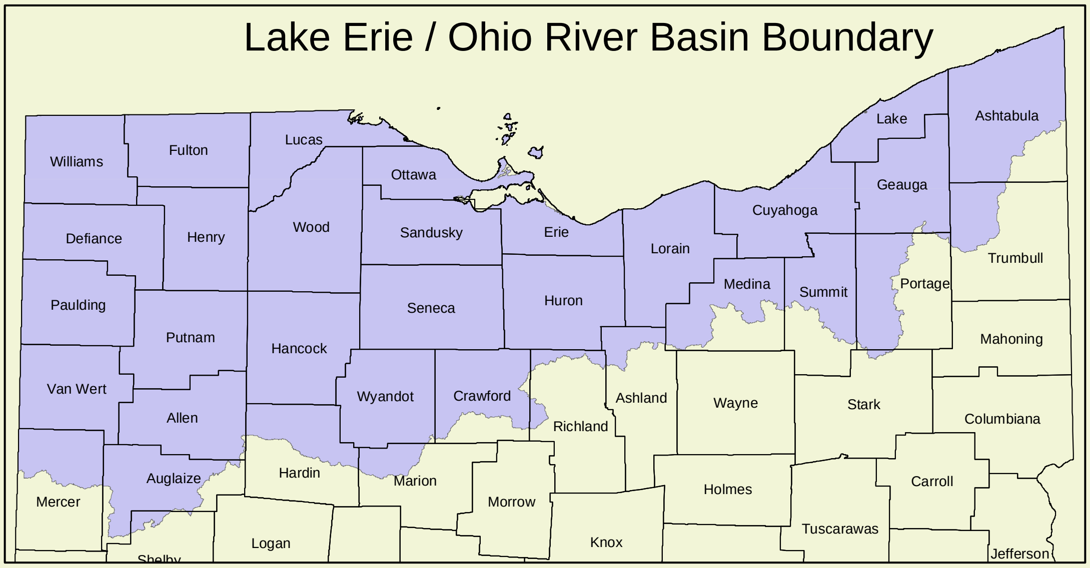
In spite of these limitations, this work has resulted in a rather complete and current compilation of information available on nutrient reduction efforts in the Ohio Lake Erie basin. It also presents an assessment of policy tools that are a part of nutrient reduction strategies which (collectively) are yielding at least some level of progress in their pursuit of water quality improvement goals in other water basins. As a result, the information presented here can enlighten policymakers and natural resource administrators on policy tools that are being used to reduce nutrient flows in other large American water basins. It can inform their discussions about addressing nutrient enrichment and HABs in Lake Erie.

[4] However, we should point out that, due to limitations on the availability of geographically available information, we were limited to statewide information on some nutrient reduction programs in the Lake Erie basin. This was particularly true for agricultural programs, which appear to be subject to limits on data availability due to statutory provisions in the federal "Farm Bill" law.

Principal Findings

Nutrient Reduction Policies in the Ohio Lake Erie Basin

Our investigations of nutrient reduction policy efforts in the Ohio Lake Erie basin identified multiple policies targeting nutrient load reductions to Lake Erie. We summarize key findings below.



- Counties within the Lake Erie Basin (OEPA, 2007).

Regulatory Interventions to Reduce Nutrient Flows from Point Sources

We identified multiple instances in which federal and state regulatory authorities are used to achieve nutrient loading reductions in Ohio. Federal authorities exist under the Clean Water Act (CWA) which require point source dischargers of pollutants to waters of the US to obtain regulatory discharge permits. However, these federal requirements are administered by state agencies in Ohio (and elsewhere as well). To understand regulatory controls for nutrients in Ohio, we investigated National Pollutant Discharge Elimination System (NPDES) point source permits issued pursuant to the federal CWA, as well as state requirements which apply to releases of nutrients to Ohio waters.

We investigated three kinds of NPDES regulated point source discharges: 1) traditional NPDES permits for facilities discharging wastewaters from sewage treatment facilities and industrial/commercial processes; 2) permits for addressing storm-water discharges from separated and combined sewer systems, and; 3) potential releases from Concentrated Animal Feeding Operations (CAFOs), which are treated as point sources under the CWA.[5]

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[5] For all three of these types of point source permits, we investigated current permits in Ohio and the Lake Erie basin, using information available through state agency sources and – in most cases – these sources are available through the websites of the agencies involved. Where needed, we sought clarifications regarding the written information provided from agency staff persons who are knowledgeable regarding the information being investigated.

Our review of NPDES wastewater discharge permits found that:

- OEPA has issued a total of 1,148 NPDES permits for wastewater discharges in the Lake Erie basin.
- Of these permits, 102 are considered major permits which USEPA and OEPA define as those governing discharges of one million gallons of a day (MGD) of wastewater flow or which contain pollutants of particular concern to the water bodies to which they flow (USEPA, 2016b). The remaining 1,046 are considered minor permits.
- Out of the 102 OEPA major permits in the Lake Erie Watershed, 83 permits (81%) have effluent limits on at least one nutrient (Nitrogen and/or Phosphorus). A total of 79 of these permits have effluent limits on total phosphorus.
- There are 19 (19%) major permits that do not appear to have any nutrient limits at all.
- The majority of major permits also have monitoring requirements for nutrients, only 14 out of 102 (14%) had no monitoring requirements at all.
- Minor permits appear to be less likely to have nutrient limits and monitoring requirements than major facilities.

These and other findings are summarized in Table 1 below:

Table 1: Overview of NPDES permits in the Ohio Lake Erie Basin[6]

	Major Permits	Minor Permits	Total
#'s of Permits	102	1046	1148
Nutrient Limits (Either P or N)	83	601	684
No Limits	19[7]	445	464
Nutrient monitoring (either P or N)	87	756	843
No monitoring	15	290	305

In interpreting the figures in Table 1, however, one should be aware that not all permits necessarily need effluent limits – or perhaps even effluent monitoring requirements -- for nutrients. This is because some wastewaters discharged by NPDES permittees do not come from organic sources that are likely to include nutrients such as phosphorus and nitrogen.

One type of NPDES permittee that is likely to discharge wastewaters containing nutrients is Publicly Owned Treatment Works (POTWs). POTWs collect wastewaters from residences, businesses, and storm

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[6] We compiled data on effluent limits and monitoring requirements relating to phosphorus and nitrogen by identifying and reviewing hundreds of NPDES permits issued by OEPA in the Lake Erie basin. We retrieved and reviewed the permits from the OEPA website, http://www.epa.ohio.gov/dsw/permits/npdes_info.aspx, between Summer 2015 and Spring 2016. For nitrogen, we looked for effluent limits and monitoring requirements on ammonia and nitrates, both of which contain nitrogen. For phosphorus, we looked for and found effluent limits and monitoring requirements on total phosphorus in a number of the permits we reviewed. Additional information on methods used and NPDES related findings is available upon request.

[7] Among these 19 major permits without nutrient limits, 11 permits are for various industrial facilities, 7 are for Power Plant facilities and 1 permit is for an Oil Refinery.

water sources, and their discharges often contain organic materials. For this reason, we took a closer look at permit limits and monitoring requirements contained in NPDES permits issued to POTWs. We found:

- All major Publically Owned Treatment Works (POTWs) had some form of nutrient limits (with 55/56 having at least phosphorus limits), while only one POTW had only nitrogen limits.
- There are 141 minor POTWs that have no nutrient limits whatsoever.

Because phosphorus is of particular concern in the creation of HABs in Lake Erie, we also looked specifically at the phosphorus limits we found in major POTW permits. In particular, we assessed the monthly average total phosphorus concentration limits written into the permits of all of the major POTWs in the basin. We found that:

- Among major POTWs in the Lake Erie basin, all concentration limits are at or below 1.0 mg/L[8], as was suggested by the International Joint Commission (IJC) for major POTWs discharging to the Great Lakes about 35 years ago.
- Ten of the fifty-six permits issued to major POTWs in the Lake Erie basin have more stringent limits than 1.0 mg/L average monthly concentration, with the lowest average monthly concentration limit being .60 mg/L.

A summary of these and other findings is provided in Table 2 below.

Table 2: Overview of NPDES Permits for POTWs in the Ohio Lake Erie Basin

	Major Permits	Minor Permits	Total
#'s of Permits	56	187	243
Nutrient Limits (Either P or N)	56	141	197
No Limits	0	46	46
Nutrient monitoring (either P or N)	56	183	239
No monitoring	0	4	4

We identified controls on Combined Sewer Overflows (CSOs), which can discharge nutrients from combined sewers after major storms or rainfall events. Among NPDES permits issued by OEPA, we found that:

- There are currently 77 communities in Ohio that have approximately 1,144 permitted CSOs among them (OEPA, 2015a).[9]

[8] In 1980, the IJC's Phosphorus Management Workgroup recommended that Wastewater Treatment Plants (WWTP's) in the Great Lakes should be designed and operated so that the total phosphorus concentrations in their effluents would not exceed a maximum of 1.0 (mg/L) (IJC,1980). However, it appears as though the GLWQA itself suggested a more ambitious 0.5 mg/L goal for major POTWs in the Lake Erie water basin (GLWQA, 2012), to the extent deemed necessary by the regulatory officials involved.

- Of the 77 communities in Ohio, 45 (58%) are within the Lake Erie watershed, and may therefore discharge nutrients to the Lake Erie basin when they overflow during or after major storm events.
- Of the 45 NPDES permits for the POTWs in those Lake Erie basin communities, we found that none of the permits had CSO nutrient discharge limits but they typically had CSO monitoring and reporting requirements of some kind.

Relatedly, our review of 56 major NPDES permits for POTWs in the Lake Erie Basin revealed that all of them had monitoring requirements for Sanitary Sewer Overflows (SSO's), which can release untreated sewerage and nutrients from separated sewerage systems after major rain events.

We also reviewed required management programs relating to the control of storm water flows among large and small municipalities.[10] These programs are targeted toward large communities with 100,000 or more persons (Phase I storm water requirements) and smaller communities (Phase II storm water requirements) as well. Our review of the Ohio NPDES storm water program found that OEPA has[11]:

- Issued at least two Individual Phase I Municipal storm water permits within the Lake Erie Basin (Toledo and Akron).
- Covered 133 government entities in counties that are at least partially in the Lake Erie Basin under its small MS4 Phase II Storm Water General Permit.
- Covered 6,942 permittees in the counties that are at least partially inside of the Lake Erie Basin under its Construction Storm Water General Permit.
- Covered 1,265 permittees in those same counties under its Industrial Storm Water General Permit.

We also investigated the use of NPDES Concentrated Animal Feeding Operation (CAFO) permits to control polluted waters flowing from larger animal feeding operations in Ohio.[12] We found that:

- Currently, OEPA's NPDES CAFO program has permitted 35 operations in Ohio. However, only 15 appear to be within the 35 counties that are at least partially within the Lake Erie Watershed.
- There are currently 113 Ohio Department of Agriculture (ODA) permitted animal feeding facilities within Ohio counties that are at least partially within the Lake Erie Watershed. The

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[9] We identified a listing of communities in Ohio with CSOs from the OEPA website (http://www.epa.ohio.gov/dsw/cso/csoindex.aspx#116135672-how-many-csos-are-in-ohio). There were a total of 77 communities in the inventory that did not have an implementation status of "complete". Using ODNR's High Quality Watershed map we were able to identify the communities within the Lake Erie Watershed. We then reviewed the NPDES permit for each community in the watershed to see if the permits included CSO controls. Those that had CSO controls in their permits are included in the tallies presented.

[10] We reviewed the list of permittees under each type of storm water general permit (Phase II, Construction, Industrial, and Marina) and performed a simple count of the number of permittees covered under each general permit for the counties that are at least partially in the Lake Erie Watershed.

[11] County by county information is available upon request.

[12] We utilized the lists of permitted CAFO and CAFF facilities provided by OEPA and ODA on their respective websites and performed a simple count of the number of permitted facilities in the counties that are at least partially within the Lake Erie Basin.

ODA permitting program, the Combined Animal Feeding Facility (CAFF) program, applies to both large CAFO's and other animal feeding operations that do not meet NPDES CAFO requirements.

Overall, we found an abundance of regulatory interventions to reduce and/or control nutrient flows in the Lake Erie water basin. However, we found no central place or information source that would enable integrated management of these requirements, and we also found instances where current NPDES permits could potentially be strengthened to exercise greater control over nutrients than currently exists. In addition, for regulatory controls relating to wet weather related sources, it appeared that information on the actual implementation of compliance activities was not as readily available as it could be.

EXPENDITURES OF FUNDS AND RESOURCES TO REDUCE NUTRIENT FLOWS

During the course of our work, we identified multiple ways in which federal and state agencies expend funds and make investments to reduce nutrient flows. Below we summarize these efforts and the findings that stem from them. Two summaries are provided. One summary focuses on investments to reduce nutrient flows from point sources, where wastewaters are discharged through pipes and/or conveyances. A second summary focuses on non-point sources, where nutrients are released through diffuse flows of waters over land after rain events.

Investments in Point Source Wastewater Treatment

- WPCLF Logo (OEPA, 2014)

We investigated funding and expenditures that are made available to help control nutrients from wastewater treatment plants in Ohio and the Ohio Lake Erie Basin.[13] Ohio utilizes financial assistance programs that benefit from both state and federal funding to help local governments address point source issues. The Ohio Water Development Authority (OWDA) and the OEPA jointly manage and implement the Water Pollution Control Loan Fund (WPCLF), which provides below market rate loans to public entities for sewerage systems, Wastewater Treatment Plants (WWTPs), and related planning and construction projects. The OWDA also manages additional loan programs, such as the Fresh Water Fund, the Community Assistance Fund, and the Un-sewered Area Assistance Account. Between 2010 and 2015, these funding sources provided $2.769 billion to finance wastewater planning and construction projects in Ohio (OWDA Annual Reports 2011-2014; OEPA 2015 Annual Report). In 2015, OEPA offered a Nutrient Reduction Discount, where the agency provided an additional $1 million in loans available at a 0% interest rate for projects that include "equipment and facilities at POTWs to reduce levels of phosphorus and other nutrient pollutants" (OEPA, 2014).

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[13] We reviewed the OWDA's annual reports from 2011-2014, which provide an overview of the planning and construction loans for each year. We utilized the 2015 OEPA Annual Report to identify the total funding provided by the WPCLF (which may result in an underestimate of funding because it does not include the other OWDA loan programs highlighted above).

Investments in Nutrient Reduction from Non-Point Sources

There are numerous programs being implemented that are either directly or indirectly related to controlling nutrient flows from non-point sources (NPS's) to Lake Erie. The programs applicable to the Lake Erie Watershed are presented below. They have been separated into Federal and State funded programs. It should be noted some that some programs are federally funded but are implemented at the state level by state agencies.

Federal agencies implementing non-point source-related programs include:

- US Environmental Protection Agency (USEPA)
    - Clean Water Act Section 319 Grant Program
    - Urban Waters Grant Program
- US Department of Agriculture (USDA)
    - Multiple Farm Bill Programs[14]
- US Department of Interior (USDI)
    - Land and Water Conservation Fund Grant Program
    - US Fish and Wildlife Wetland Grant Programs
- US Department of Commerce
    - National Oceanic and Atmospheric Agency (NOAA) Coastal Management Grants
- Multi-Agency Grant Programs
    - Great Lakes Restoration Initiative
    - Sustain Our Great Lakes

- LWCF Logo

- Property protected by NOAA-ODNR Coastal Management Program

We investigated federal expenditures in these programs that support efforts to reduce nutrients in Ohio and the Lake Erie basin between 2010 and 2015, and found substantial expenditures across these programs.[15] The non-USDA programs listed above represent about $162 million in investments to reduce nutrient flows by the federal government from 2010-2015 in the Lake Erie Basin. Over 330 individual projects were supported that directly or indirectly impact nutrient flow reductions within the Lake Erie Basin. In 2014, these programs invested in 67 projects totaling about $33 million in expenditures. In 2014 alone, USDA Farm Bill programs

[14] Farm Bill Programs include: Agricultural Conservation Easement Program, Conservation Technical Assistance, Conservation Stewardship Program, Conservation Innovation Grants, Environmental Quality Incentives Program, Conservation Reserve Program, Conservation Reserve Enhancement Program, Forest Legacy Program.

[15] We relied on publically available reports on expenditures and projects funded by these federal programs between 2010 and 2015. We summarized information related to the numbers of projects funded, the funding amount, and any available details on the projects, such as their purpose and implementation agency. We also differentiated among projects based on whether they: 1) were explicitly focused on nutrient reduction; 2) appeared likely to reduce nutrient flows indirectly through other conservation measures, and; 3) focused on education and research relevant to nutrient flows in the Ohio Lake Erie basin.

made investments exceeding $90 million statewide.[16] This USDA funding in 2014 implemented Best Management Practices on over 700,000 acres of land in Ohio.

As we conducted research underlying this project, we found it difficult to locate geographically specific information on the use of federal funds to implement nutrient controls by farmers and the agricultural community. Without this kind of information it is difficult to determine the extent to which agricultural practices to reduce nutrient loads are being implemented or to gauge their effectiveness. It appears that the difficulties we experienced in this area are not unusual. Rather, they appear to be at least partially traceable to Section 1619 of the 2008 federal Farm Bill (which is codified in 7 U.S.C. 8791) (Chite, 2014), which prohibits the release of any information on agricultural operations that is tied to participation in federal farm programs. Ultimately, addressing nutrient enrichment problems in the Lake Erie basin and elsewhere depends on developing, maintaining, and using a solid base of information on nutrient loadings and efforts made to reduce them.

- Property protected in part by ODNR's Nature Works Program

We also investigated state funding for non-point source nutrient control efforts in the Lake Erie basin[17]. We found that the State of Ohio also funds a number of programs that directly target non-point sources of nutrient flows to the state's waters, as well as programs that indirectly help reduce nutrient flows from non-point sources. The state agencies tasked with implementing state-funded non-point source pollution reductions have included:

- Ohio Department of Agriculture (ODA)
    - Agricultural Pollution Abatement Program
- Ohio Department of Natural Resources (ODNR)
    - Nature Works Grant Program
- Ohio Environmental Protection Agency (OEPA)
    - Surface Water Improvement Fund Grant Program
- Ohio Development Services Agency (ODSA)
    - Green Storm-water Infrastructure Loan Program
- Ohio Public Works Commission (OPWC)
    - Clean Ohio Green Space Protection Fund
- Ohio Lake Erie Commission (OLEC)
    - Lake Erie Protection Fund

[16] We found that information on expenditures by some USDA programs more difficult to identify and collect than other nonpoint source expenditures. For this reason, we report only 2014 statewide figures here.

[17] We relied on publically available reports on expenditures and projects funded by state agencies between 2010 and 2015. We collected and summarized information on funding allocated, numbers of projects, and descriptive information on the funded projects (as available). This data collection approach is similar to the one that we used for assessing federal NPS expenditures.

The state funded non-point source programs represent about $135 million investment over the 2010-2015 time period (not including funds provided through the ODA Agricultural Pollution Abatement Program). Funds were utilized to support 244 projects that directly or indirectly targeted nutrient load reductions in Ohio's Lake Erie Basin.

Overall, the information presented above demonstrates that federal and state agencies have invested many millions of dollars in efforts to reduce nutrient flows in the Lake Erie basin. While these significant investments appear appropriate given the impacts of the HAB problem, it seems unlikly that significant investments – in and of themselves – represent an adequate remedy for the continuing HAB problems that confront the Lake Erie basin and the people and economy that depend upon it.

Strategy, Planning, and Communications Efforts Focused on Lake Erie

Through our web searches and discussions with Lake Erie water management professionals, we also identified multiple efforts by the State of Ohio, US federal agencies, and the international organizations with whom they work to define problems associated with nutrient flows to Lake Erie and communicate ways in which they can be addressed. Significant efforts and documents we encountered include the following:

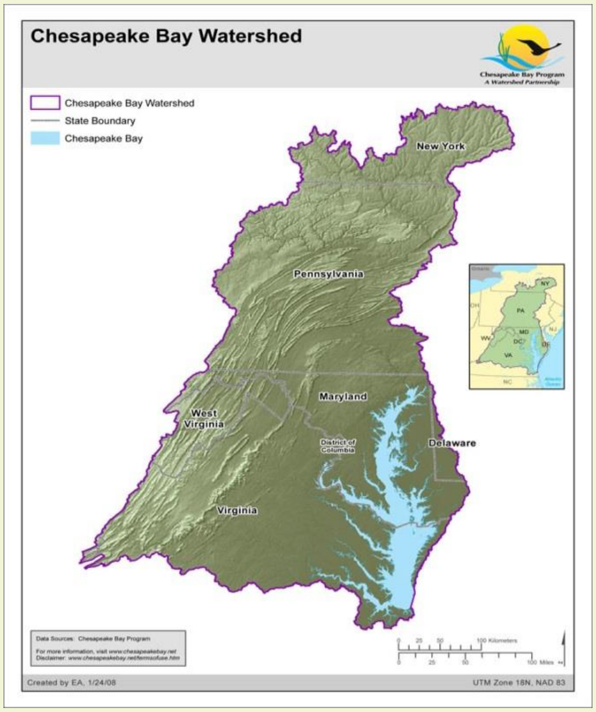
- Lake Erie Binational Nutrient Management Strategy (2011)
- Directors' Agricultural Nutrients and Water Quality Working Group Final Report and Recommendations (2012)
- Great Lakes Water Quality Agreement (Annex 4) (2012)
- Lake Erie Commission (2013): Lake Erie Protection and Restoration Plan
- Ohio Lake Erie Phosphorus Task Force II Final Report (2013)
- OEPA (with ODA and ODNR): Ohio Nutrient Reduction Strategy (2013)
- International Joint Commission's Lake Erie Ecosystem Priority (LEEP) Report (2014)
- Great Lakes Commission: A Joint Action Plan for Lake Erie (2015)
- US and Canada Agreement on Nutrient Reduction Targets (2016)

In most of these cases, these efforts were focused on characterizing the nutrient problem and signaling broad directions through which it could be addressed. However, in these cases, the focus was generally not on targeted communications with specific groups or individuals whose work or behaviors directly yielded nutrient flows to Lake Erie. While many of these documents were created in a collaborative fashion, involving multiple agencies and organizations, they often focused on broadly framing the issue and the strategies to address the problems. We were unable to identify a document that serves as a "one stop shop" that clearly delineates the roles and responsibilities of specific jurisdictions and agencies that can serve as a management framework. However, there may be an opportunity to create such a tool through the GLWQA Annex 4 process currently underway for Lake Erie.

Organization: Engaging Resources and Capacities to Reduce Nutrient Flows

We found ample evidence of government agency efforts to achieve nutrient reductions in the Lake Erie basin. We identified four federal agencies and six state agencies that have pursued this goal over the past five or six years, and these agencies funded and implemented numerous regulatory, financial assistance, and information dissemination programs. What we did not find, however, was any single organization or organizational effort that manages ongoing implementation of interventions to achieve the goal of nutrient reduction in the Lake Erie basin. The Lake Erie Commission plays a valuable coordination role, but it is a small organization that was built to advise on the development of policy, not to steer and guide policy implementation. This point is exemplified by the fact that it took our team substantial time to compile the basic information on policy efforts currently in place to reduce nutrient flows in the Lake Erie basin. This experience appears symptomatic of substantial policy fragmentation, and a need to better enable coordination of nutrient reduction efforts in Ohio. This picture of fragmentation in Ohio is exacerbated when one recognizes that Ohio is just one of a number of political jurisdictions that have an interest in the quality of water in Lake Erie and in the control of nutrients that have been giving rise to HABs.

Nutrient Reduction Policies in Other American Water Basins: Chesapeake Bay, Long Island Sound, and Tampa Bay



- Chesapeake Bay Watershed (CBP, 2012)

Chesapeake Bay: Background and Accomplishments

The Chesapeake Bay watershed lies in the mid-Atlantic region of the US, and spans more than 64,000 square miles. It encompasses parts of six states—Delaware, Maryland, New York, Pennsylvania, Virginia and West Virginia—and the entire District of Columbia. For decades now, the Chesapeake Bay has endured stresses relating to the release of nitrogen, phosphorus, and sediments to the bay. In 1983, Maryland, Virginia, and Pennsylvania – along with the District of Columbia, the Chesapeake Bay Commission, and the USEPA – established the Chesapeake Bay Partnership (Chesapeake Bay Agreement, 2014). Following years of voluntary efforts to address nutrient enrichment in the Bay, USEPA released its Chesapeake Bay TMDL, the largest TMDL developed to date in the US in December of 2010.

The Chesapeake Bay Program and the states with which it works have begun implementing the Chesapeake Bay TMDL, as well as an accountability framework that was established to enable its success (described briefly below). Through this process, they have systematically reviewed information provided by the states on steps taken to reduce nutrient loads to the bay. The results of this tracking process are publicly available on the Chesapeake Bay Program website.

According to information drawn from this publicly available tracking system in early 2016, there have been nutrient load reductions of 13% for nitrogen and 10.5% of phosphorus between 2009 and 2015. These reductions brought total estimated loadings of nitrogen to 242 million pounds and phosphorus to 17 million pounds. The nitrogen and phosphorus loading goals for 2015 are 207,571,430 pounds and 14,457,190 pounds, respectively, so there is still much work needed to meet the CBP's long term nutrient reduction goals (Chesapeake Bay Stat, 2016). Even so, these estimated loading reductions, along with some indicators of ambient water quality improvement in the bay (Langland et al, 2012; CBP, 2015) suggest that measureable progress in water quality improvement is being made.

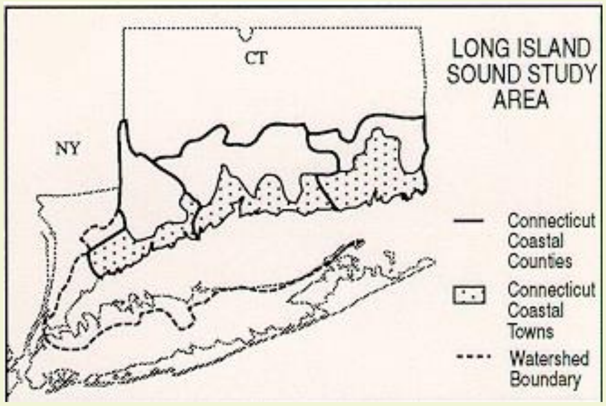
Policy Tools in the Chesapeake Bay

Through our efforts to review policies and programmatic efforts to address nutrients in the Chesapeake Bay Region we identified the following policy approaches and tools being developed and/or used in the Chesapeake Bay area that we did not encounter for the Ohio Lake Erie basin:

- A management framework consisting of 1) a scientific basis for integrated watershed and pollutant transport models across the entire Chesapeake Bay; 2) a tracking and accountability framework, and; 3) an organizational effort to implement nutrient controls that is commensurate with the scope of the bay's water quality problems.
- Water Quality Criteria and Standards for the Chesapeake Bay (USEPA and Delaware, Maryland, Virginia, and the District of Columbia)
- A TMDL framework for guiding nutrient reduction efforts
- Nutrient Management Requirements for Smaller Animal Feeding Operations (AFOs)
- Water Quality Trading Programs
- Agriculture Certainty Programs
- Innovative funding (Virginia's budget surplus Water Quality Improvement Fund)

Long Island Sound Study Program: Background and Accomplishments

The Long Island Sound drainage basin is about a quarter the size of the Chesapeake Bay's drainage basin, as it covers about 16,000 square miles of land from New York and Connecticut up into portions of New England (CCMP, 2015). One substantial water quality challenge facing the Sound is low levels of dissolved oxygen (DO), which yield anoxic zones that threaten fish and other aquatic species. This problem has been recognized for years now, and it is traceable to nitrogen loads released to the Sound from WWTP's in the surrounding urban areas, upstream agricultural sources, air deposition, and the ocean itself.



- Long Island Sound Study Area (LISS, 2016)

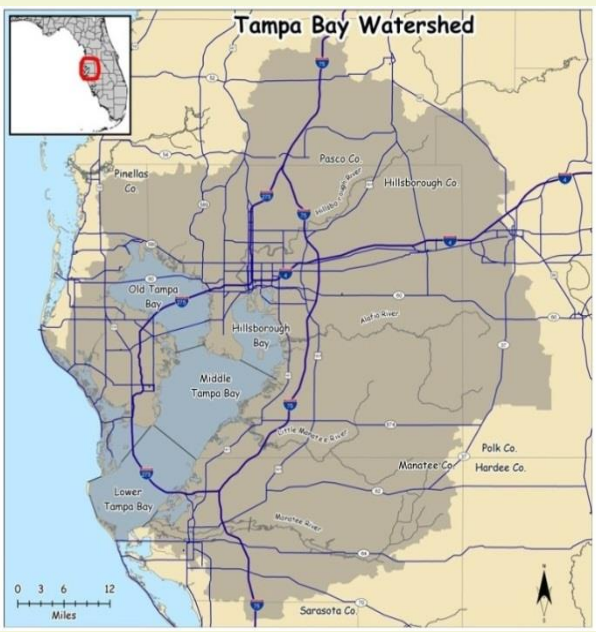
Through efforts at the state and federal levels, and the increased attention to the health of the Sound that was associated with them, USEPA and the states of New York and Connecticut established the Long Island Sound Study (LISS), "a Management Conference involving federal, state, interstate, and local agencies, universities, environmental groups, industry, and the public" (LISS, 2015). In 2001, the USEPA approved a multi-jurisdictional TMDL calling for a 58.5% reduction in nitrogen loads (LIS TMDL, 2000), a large proportion of which were to be achieved through upgrades to WWTP's.
In 2014, wastewater treatment facilities in the Sound's water basin were reported to have achieved "94% of the nitrogen reduction goal established in the USEPA approved DO TMDL, which means that 108,000 fewer pounds of nitrogen were discharged into the Sound every day" (LISS, 2015). In addition, the overall

trends appear to suggest that the size and durations of hypoxic areas size in the Sound appear to be diminishing.[18]

Policy Tools in the Long Island Sound

As was the case for other water basin programs reviewed during the course of this work, policy tools used in the Long Island Sound included regulatory interventions such as NPDES permits, funding support programs, and activities providing communications to broad audiences about the conditions of the Sound and the steps necessary to address them. Policy approaches and tools we encountered in the Long Island Sound that we did not identify in Ohio included:

- Connecticut's Effluent Trading Program which allows dischargers to trade their Waste Load Allocations across NPDES permits.
- New York's "Bubble" Permit Policy for New York City, through which the state has enabled the city to pool permitted nitrogen discharges together under two WLA "bubble" allocations. This enables the city to achieve its allocated reductions in whatever plants are most likely to yield the needed reductions in the most cost-effective fashion.
- A scientific network with ties to the Long Island Sound Study Program, which appears to serve as a cross-jurisdictional coordinating office.



-The Tampa Bay Watershed (TBEP, n.d.)

Tampa Bay Background and Accomplishments

While the Tampa Bay is much smaller than both the Chesapeake Bay and the Long Island Sound, it is the largest open water estuary in the State of Florida. Tampa Bay extends in a "Y" shape from the Gulf of Mexico, and covers about 400 square miles within a watershed of about 2600 square miles (Greening, 2014). Nitrogen is the nutrient of concern in the Bay, as it is reported to contribute to eutrophication and it affects both water quality in the Bay and the aquatic life that inhabits it.

In 1991, the Tampa Bay Estuary Program (TBEP) was established with support from USEPA. As with LISS, the TBEP benefited from funding provided through the NEP. The TBEP and its partners also adopted a Comprehensive Conservation and Management Plan (CCMP) that included measureable goals for the achievement of Tampa Bay's designated uses (Greening et al, 2014).

As a result of coordinated efforts between the public and private sectors in the Tampa Bay region, nutrient loadings are estimated to have been reduced by more than 50% since the 1970s (Greening, 2008).

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[18] While the trends suggested here are encouraging, it is important to recognize that they remain subject to both significant variability on a year to year basis and to long term change. In 2012, for example, the size of hypoxic area in the Sound increased dramatically due to climatic, temperature, and precipitation related factors.

However, because of population growth in the Tampa Bay Region, the overall level of nitrogen loading reduction is estimated to reflect an 80% reduction in per capita total nitrogen contributions to the Tampa Bay between the mid-1970's and 2010 (Greening, 2008). Importantly, these loading reductions have been accompanied enhanced compliance with "chloryphyll a" water quality targets for the Tampa Bay estuary in recent years (Greening et al, 2014).

Policy Tools in the Tampa Bay

As was the case with the other water basin programs we investigated, we found evidence of the use of regulatory permits, grant funding, and communications to broad audiences regarding to nutrient issues in Tampa Bay. We also identified policy approaches and tools in the Tampa Bay region that we did not identify in our Ohio Lake Erie Basin inventory. Some of these policy approaches and tools are outlined below:

- A public-private partnership, the Tampa Bay Nitrogen Management Consortium (TBNMC), comprised of public and private sector stakeholders who are concerned about water quality and economic vitality in the Tampa Bay area.
- A tracking and accountability system to measure ongoing progress in the implementation of nutrient reduction efforts
- State fertilizer law regulating turf grass fertilizer products and their application.
- Policies targeting Air Emissions of nitrogen
- Lawn Fertilizer Social Marketing Campaign
- An "Integrated Watershed-Groundwater-Circulation-Ecology Model" to provide a scientific foundation for nutrient reduction efforts.

Significance

The significance of this project lies in its compilation of information on nutrient reduction efforts in the Ohio Lake Erie basin and its review of othere basin-wide nutrient reduction efforts, as well as in the lessons and ideas it offers to inform policy discussions about ways to reduce nutrient flows to Lake Erie. We have found that federal government agencies and the State of Ohio are making *substantial* efforts to reduce nutrient flows in the Ohio Lake Erie basin. They are requiring many hundreds of federal and/or state permittees to assess and/or develop nutrient treatment and management capacities. They are spending many millions of dollars on nutrient reduction efforts. They are also collecting and disseminating information on nutrient enrichment, HABs, and ways in which these problems can be addressed. And finally, both federal and state governing entities are organizing multiple efforts to address and/or manage flows of nutrients to the Lake Erie water basin. In spite of these efforts, however, Ohio and its jurisdictional neighbors in the Great Lakes region continue to face challenges and threats associated with nutrient enrichment and HABs in Lake Erie.

Based on the information presented above, and other information compiled and analyzed during the course of this project, we offer at least two lessons for Ohio policymakers and natural resource practitioners. First, while the State of Ohio and federal government agencies are carrying out many activities to reduce nutrient flows, they appear to be fragmented. They do not appear to be implemented in a way that adheres to a single coordinated and focused nutrient reduction strategy targeted to the Ohio Lake Erie basin. Second, at least several other water basin programs around the US appear to be focusing their nutrient reduction efforts in strategic and coordinated fashion, and these efforts appear to be characterized by not only clearly articulated goals but also by tracking and accountability structures that measure progress toward achieving the goals that have been established. While the international character of the Lake Erie basin creates additional challenges not faced by other water basin programs, it seems appropriate to accept these challenges and address them as a part of an effort to achieve a focused and strategic implementation of Lake Erie nutrient reduction efforts.

Our inventory of current Ohio Lake Erie basin nutrient reduction policies and assessment of other American water basin programs has yielded multiple ideas regarding specific policies and practices that can be considered for the Ohio Lake Erie basin. Below, we present a number of these ideas in three broad categories: 1) Institutionalization of nutrient reduction efforts across organizations and jurisdictions; 2) Strengthening point source nutrient reduction efforts, and; 3) Strengthening nonpoint source nutrient reduction efforts.

It is important to recognize that we offer these policy ideas – at least at this point in time -- as a menu of possible options, rather than as recommendations for immediate implementation. This is because each of the suggestions below deserves more thought and evaluation than we could provide as a part of this assessment. For example, some of the ideas below could result in reduced nutrient flows, but at costs that far exceed their benefits in terms of nutrient flow reduction. In other cases, it may be that ideas presented below are already being initiated and/or implemented in some way. However, the use of a number of these policy approaches and tools by water basin programs that appear to be moving forward in productive fashion – along with the seriousness of the Lake Erie HAB problem -- makes these ideas worthy of active consideration.

<u>Institutionalizing Nutrient Reduction Efforts Across Organizations and Jurisdictions</u>

The State of Ohio should actively consider:

- Establishing and adequately funding one central organization and tasking it with responsibility for coordinating, tracking, and *assuring implementation* of nutrient reduction efforts across the Ohio Lake Erie basin.
    - As a part of this effort, this organization should develop a coordinated system for tracking implementation actions and assuring accountability for nutrient reduction efforts in the Ohio Lake Erie basin.
    - Over time, this effort should be coordinated with, and expanded to include, other jurisdictions in the Lake Erie basin.

- Working actively with other jurisdictions in the Lake Erie basin to establish an integrated basin-wide monitoring and modeling effort that enables an integrated basin-wide understanding of ways in which specific nutrient reduction efforts may yield improvement of water quality in Lake Erie. Our discussions suggest that valuable and significant water quality monitoring and modeling efforts are being undertaken, but it appears that there may be a need to enhance the integration and coordination of these effort across funding organizations and governing jurisdictions. The efforts made to model and monitor nutrient flows in the Chesapeake Bay area provide a potential model for consideration in this regard.

- Developing a consortium of entities with a stake (economic and otherwise) in the future of Lake Erie, and forming a broad private-public sector partnership or consortium devoted to reducing nutrient flows in the Lake Erie Basin. The nutrient management consortium managed in the Tampa Bay area provides a potential model for consideration in this regard.

- Developing and expanding pollution *abatement* strategies for nutrient flows, based on the GLWQA Annex 4 process and the water quality goals that underlying them.[19] However, over time, Ohio may also want to consider: 1) creating formal water quality standards for nutrients in Lake Erie (hopefully, ones that are consistent with the GLWQA agreement goals), and; 2) declaring impairment(s) of the Lake consistent with those standards after they are promulgated.

<u>Strengthening Point Source Nutrient Controls</u>

The State of Ohio should actively consider:

- Developing and implementing more comprehensive nutrient management requirements for animal feeding operations (and potentially other agricultural sectors), perhaps similar to those being implemented in Maryland.

- Reviewing existing NPDES nutrient-related permit requirements – as well as available data on nutrient concentrations in wastewater releases -- for dischargers in the Lake Erie basin. It appears as though Ohio is already moving in this direction to at least some extent. Requirements included in recent legislation (SB 1) mandate that major POTWs monitor total *and* dissolved reactive phosphorus pursuant to a new, renewed, or modified NPDES permit. In addition, all major POTWs that are not subject to a phosphorus limit will need to complete and submit to OEPA a study that evaluates the technical and financial capability of the existing treatment facility to reduce the final discharge of phosphorus to 1 mg/L. While these requirements appear to be useful steps, consideration should also be given to expanding nutrient reduction requirements to smaller POTWs and/or other NPDES permittees which are known to discharge phosphorus and nitrogen. In addition, for larger POTWs, consideration should be given to strengthening current nutrient controls, perhaps in ways that move toward and/or are consistent with the GLWQA's suggested .5 mg/l average monthly concentration limit for total phosphorus.

  Relatedly, the State of Ohio should review the manner in which it is currently funding and seeking to regulate both storm water flows for separated sewerage systems and animal feeding operations. While we found clear evidence of federal and state regulatory interventions in both of these areas, we did not encounter publicly available evidence demonstrating full implementation and compliance with these regulatory requirements. For this reason, it seems appropriate to evaluate current practices and funding levels for these programs to assure they can and are accomplishing their intended purposes.

- Investigating and considering water quality effluent trading and/or bubble permit programs for nutrient control in the Lake Erie basin. Here, lessons can be learned from the existing programs in Virginia, Connecticut, New York, and Ohio's Miami River basin.

[19] It is appropriate to acknowledge that this process already appears to be underway, to at least some extent.

Strengthening Nonpoint Source Control Efforts

The State of Ohio should actively consider:

- Engaging with the agricultural community in the state to enable greater generation and use of geographically based information on the implementation of BMPs and other nutrient reduction based agricultural management approaches. Exploring possible collaborative efforts with the agricultural community to allow for access to this information seems appropriate because privacy protection language included in the US Farm Bill appears to have made it difficult to gain access to the information needed to fully evaluate the effectiveness of existing agricultural cost share and financial assistance programs.

- Developing a budget surplus set aside policy that is targeted to support ongoing nutrient reduction efforts in the Lake Erie basin (and perhaps the Ohio River basin as well), perhaps similar to what has been done in Virginia.

- Developing and supporting voluntary initiatives to increase awareness and use of fertilizers on lawns and in other environments.

- Establishing state lawn fertilizer requirements, as we understand has been done in Florida.

- Developing a livestock exclusion and buffer zone support program to minimize nutrient flows to nearby water bodies, perhaps similar to the one that has been implemented in Virginia.

- Developing a Resource Management Plan and Agricultural Uncertainty program, perhaps modeled in part on the work that Virginia and Maryland have done in this area.

Over the next year or two, as political commitments within and across jurisdictions are solidified and allocations of nutrient reduction responsibility across jurisdictions and economic sectors are identified through the GLWQA agreement process, it will become important to identify ways to reduce nutrient flows to the Lake Erie basin in ways that will combat excessive nutrient enrichment and the HABs that are increasingly associated with it. Our hope is that the public management lessons and policy tools identified above can productively inform discussions about how best to accomplish the substantial reductions in nutrient flows that are necessary to accomplish these water quality goals.

References

Associated Press. “Record-breaking Lake Erie algae bloom was size of New York City”. Accessed November 11, 2015. http://www.cbsnews.com/news/record-breaking-lake-erie-algae-bloom-was-size-of-new-york-city/. Last modified 2015.

Austin, J., Anderson, S., Courant, P., Litan, R. “Healthy Waters, Strong Economy: The Benefits of Restoring the Great Lakes Ecosystem.” The Brookings Institution. 2007.

Chesapeake Bay Program (CBP). "Experts Consider Chesapeake Bay an Ecosystem in Recovery." Chesapeake Bay News. February 03, 2015. Accessed 2016. http://www.chesapeakebay.net/blog/post/experts_consider_chesapeake_bay_an_ecosystem_in_recovery .

Chesapeake Bay Program. "Water Quality: TMDL Tracking." Chesapeake Bay Program. February 2016. Accessed 2016. https://stat.chesapeakebay.net/?q=node/130.

Chesapeake Bay Watershed Agreement. "Chesapeake Bay Watershed Agreement." Chesapeake Bay. 2014. Accessed 2016. http://www.chesapeakebay.net/documents/ChesapeakeBayWatershedAgreemenetFINAL.pdf .

Chite, Ralph. "The 2014 Farm Bill (P.L. 113-79): Summary and Side-by-Side." Farmland Information Center. 2014.

Great Lakes Commission (GLC) Nutrient Management: A Summary of State and Provincial Programs in the Great Lakes – St. Lawrence River Region. 2012.

Great Lakes Commission (GLC). "An Interim Joint Action Plan for Lake Erie." Great Lakes Commission. June 2015. Accessed 2016. http://glc.org/files/projects/lent/FINAL-LENT-interim-Joint-Action-Plan-June-2015.pdf .

Great Lakes Water Quality Agreement (GLWQA). 2012. Retrieved from http://www.ijc.org/en_/Great_Lakes_Water_Quality.

Greening, Holly. “Tampa Bay Nitrogen Management Consortium: RA and TMDL Update”. Tampa Bay Estuary Program. November 2008.

Greening, Holly. “What’s Working in Tampa Bay: Nitrogen Management Consortium”. Tampa Bay Estuary Program. February 2014.

Greening, H., Janicki, A., Sherwood, E.T., R. Pribble, Johansson, J. *Ecosystem Responses to Long-Term Nutrient Management in an Urban Estuary: Tampa Bay, Florida.* Estuarine, Coastal and Shelf Science. 151 (2014) A1-A16.

Hood, Christopher. *The Tools of Government*. London, Macmillan, 1983.

International Joint Commission (IJC). "Phosphorus Management for the Great Lakes: Final Report of the Phosphorus Management Strategies Task Force." International Joint Commission. July 08, 1980. http://www.ijc.org/files/publications/D16.pdf

International Joint Commission (IJC). *A Balanced Diet for Lake Erie: Reducing Phosphorus Loadings and Harmful Algal Blooms*. 2014.

Langland, Michael, Blomquist, Joel, Moyer, Douglas, and Hyer, Kenneth. “Nutrient and suspended-sediment trends, loads, and yields and development of an indicator of stream water quality at non tidal sites in the Chesapeake Bay watershed”, 1985–2010: U.S. Geological Survey Scientific Investigations Report 2012-5093, 26 p. (2012). (Also available online at http://pubs.usgs.gov/sir/2012/5093/

Long Island Sound Study (LISS). "2015 Comprehensive Conservation and Management Plan." Long Island Sound Study. September 16, 2015. Accessed 2016. http://longislandsoundstudy.net/2015/09/2015-comprehensive-conservation-and-management-plan/

Long Island Sound (LIS) TMDL. A Total Maximum Daily Load Analysis to Achieve Water Quality Standards for Dissolved Oxygen in Long Island Sound. Prepared by the New York State Department of Environmental Conservation and the Conneticut Department of Environmental Protection. 2000.

Lucas County Board of Commissioners. Moving Forward: Legal Solutions to Lake Erie’s Harmful Algal Blooms. Report prepared for the Lucas County Board of Commissioners by Jack Tuholske and Ken Kilbert. 2015.

Ohio Clean Lakes Initiative. "Ohio Strengthens Efforts to Further Protect Lake Erie." Governor of Ohio, John R. Kasich. August 14, 2014. Accessed 2015. http://www.governor.ohio.gov/Portals/0/08.14.14%20Protecting%20Lake%20Erie%20Fact%20Sheet.pdf

Ohio Environmental Protection Agency. *State of Ohio Water Pollution Control Loan Fund: Final 2015 Program Management Plan.* 2014.

Ohio Environmental Protection Agency (OEPA)."Combined Sewer Overflow Control Program." 2015.Accessed April 04, 2016. http://www.epa.ohio.gov/dsw/cso/csoindex.aspx#116135672-how-many-csos-are-in-ohio

Ohio Phosphorous Task Force (Task Force). *Ohio Lake Erie Phosphorus Task Force II: Final Report*. (2013).

Ohio Sea Grant. “Harmful Algal Blooms in Ohio Waters: Ohio Sea Grant Fact Sheets”. Ohio State University, 2011

Ohio Water Development Authority. “2010-2014 Annual Reports”.

United States Environmental Protection Agency (USEPA) Press Release. "Governments of Canada and the United States Announce Phosphorus Reduction Targets of 40 Percent to Improve Lake Erie Water Quality and Reduce Public Health Risk." United States Environmental Protection Agency (EPA). February 22, 2016a. Accessed 2016. http://yosemite.epa.gov/opa/admpress.nsf/0/D6FB4CB50080797585257F610067D8BD

United States Environmental Protection Agency (U.S. EPA). "Supplemental Module: NPDES Permit Program." EPA. March 4, 2016b. Accessed 2016. https://www.epa.gov/wqs-tech/supplemental-module-npdes-permit-program .

Zingone, Adriana, and Henrik Oksfeldt Enevoldsen. "The diversity of harmful algal blooms: a challenge or science and management." *Ocean & Coastal Management* 43, no. 8 (2000): 725-748.

Appendix 1: List of Basin Programs in other States

NEP Programs

1. Albemarle-Pamlico National Estuary Program
2. Barataria-Terrebonne National Estuary Program
3. Barnegat Bay Partnership
4. Buzzards Bay National Estuary Program
5. Casco Bay Estuary Partnership
6. Charlotte Harbor National Estuary Program
7. Coastal Bend Bays and Estuaries Program
8. Delaware Center for the Inland Bays
9. Galveston Bay Estuary Program
10. Indian River Lagoon National Estuary Program
11. Long Island Sound Study
12. Lower Columbia Estuary Partnership
13. Maryland Coastal Bays Program
14. Massachusetts Bays Program
15. Mobile Bay National Estuary Program
16. Morro Bay National Estuary Program
17. Narragansett Bay Estuary Program
18. New York-New Jersey Harbor Estuary Program
19. Partnership for the Delaware Estuary
20. Peconic Estuary Program
21. Piscataqua Region Estuaries Partnership
22. Puget Sound Partnership
23. San Francisco Estuary Partnership
24. San Juan Bay Estuary Partnership
25. Santa Monica Bay Restoration Commission
26. Sarasota Bay Estuary Program
27. Tampa Bay Estuary Program
28. Tillamook Estuaries Partnership

Programs Outside NEP

29. Boston Harbor
30. Chesapeake Bay
31. Great Lakes Program
32. Gulf of Mexico

Appendix 2: List of Federal and State Programs Targeting Nutrients

State Non-Point Source Programs:

1. Ohio Department of Natural Resources (ODNR) – Nature Works
2. Ohio Environmental Protection Agency (OEPA)- Surface Water Improvement Fund
3. OEPA – Water Resource Restoration Sponsorship Program
4. Ohio Development Services Agency – Green Storm water Infrastructure loans
5. Ohio Public Works Commission – Clean Ohio Greenspace
6. Lake Erie Commission – Lake Erie Protection Fund

State Point Source Programs (these programs do receive federal funding as well)

1. OEPA and Ohio Water Development Authority (OWDA) - Water Pollution Control Loan Fund
2. OWDA - Fresh Water Fund
3. OWDA - Un-sewered area assistance account
4. OWDA - Community Assistance Fund

Federal NPS Programs

1. Environmental Protection Agency (EPA) – Clean Water Act Section 319 Grant Program
2. EPA – Urban Waters Small Grant Program
3. Department of Interior (DOI) – Land and Water Conservation Fund
4. DOI – US Fish and Wildlife Services (USFWS)- North American Wetlands Conservation Act
5. DOI - USFWS National Coastal Wetlands Conservation Grant Program
6. DOC-NOAA – Coastal Zone Program
7. Multi-agency – Great Lakes Restoration Initiative
8. Multi-agency – Sustain Our Great Lakes program
9. US Department of Agriculture (USDA) – Agriculture Conservation Easement Program
10. USDA – Conservation Technical Assistance
11. USDA – Conservation Stewardship Program
12. USDA – Conservation Innovation Grants
13. USDA – Environmental Quality Incentives Program
14. USDA – Conservation Reserve Program
15. USDA – Conservation Reserve Enhancement Program
16. USDA – Forest Legacy Program

Adopt Your Waterway

Basic Information

Title:	Adopt Your Waterway
Project Number:	2015OH482O
Start Date:	6/1/2015
End Date:	5/30/2017
Funding Source:	Other
Congressional District:	3rd
Research Category:	Not Applicable
Focus Category:	Education, Water Quality, Surface Water
Descriptors:	Membrane separations; water treatment; biomimetic
Principal Investigators:	Zuzana Bohrerova

Publications

There are no publications.

ACTIVITY PROGRESS REPORT

Adopt Your Waterway Training

Principal Investigators: Zuzana Bohrerova, OSU, Roxanne Anderson, OSU and Friends of the Olentangy Watershed NGO (Laura Fay) and Ohio Chapter Sierra Club (Elissa Yoder)

Problem and Information Transfer Objectives:
The tributaries of the Lower Olentangy River are under increasing urbanization pressures and lie in areas of aging urban infrastructure, resulting in decreases in native vegetation, development in close proximity to streams, and problems with sewer overflows. Therefore there is concern about current and potential future degradation of these streams. Together with our partners at Ohio Sierra Club Water Sentinel Program and Ohio Water Resources Center and Friends of the Lower Olentangy Watershed (FLOW) we will aim to educate over 100 citizens in the watershed on general water quality issues with the aim to more thoroughly educate and retain about 30 families/volunteers to become Water Stewards of the lower Olentangy Watershed. We are proposing to grow tributary champions via our educational efforts.

The selected tributaries for monitoring - Adena Brook, Glen Echo Ravine, Kempton Run, and Turkey Run - although dominantly suburban and urban, lie in diverse demographic and socioeconomic areas of the lower Olentangy River watershed and drain about 7.1 square miles of the watershed. Due to increasing urbanization pressures and continuing aging of old urban infrastructure, the resulting decrease in greenery, development in close proximity to streams, and problems with sewer overflows there is concern about current and potential future degradation of these streams. Therefore there is an urgent need to monitor their water quality, increase public knowledge about the effects of landscape changes, non-point sources of pollution and nutrient run off on water quality, and to thereby enhance individual stewardship behavior as well as community capacity.

Numerous people have contacted FLOW over recent years expressing an interest in water stewardship and water monitoring activities in an effort to become involved in monitoring and protecting their watershed. Additionally, two of the selected tributaries - Adena Brook and Glen Echo Ravine – have active volunteering groups present that are focused on promoting the services these streams provide to their communities. We will also focus on direct local participation around the selected streams, and identifying and encouraging champions for these target areas, a practice which is often connected to long term success
of volunteer programs.

There is previous data on the water quality and in some instances macroinvertebrate sampling of these tributaries collected by Ohio EPA, although most of the data are ten of more years old. However, in recent years positive changes have been made in terms of watershed health; the city is increasingly working on reducing stormwater overflows and on decreasing point and non-point source pressure might have led to mixed effects on the waterways. Comparing citizens' collected data with previous archived data will provide additional learning opportunities for the public and enhance their motivation to individual and community action by experiencing the effects of changes in the watershed on water quality.

a. **Collaborators:** This project brought together three previous FLOW Science Committee members that had not collaborated on a project before: Friends of the Lower Olentangy Watershed (FLOW), Sierra Club's Ohio Chapter and the Ohio Water Resources Center. Our collaboration has been successful and yielded significant amounts of coordination, knowledge and resources leveraged. We ended up not working closely with the ODNR Scenic River Program due to their staff changes in the program, but their expertise on macroinvertebrate sampling was filled by the FLOW watershed coordinator, Roxanne Anderson. This expertise was essential in leading and coordinating the project in the summer and fall 2015. Roxanne became the key personnel in the project, having field experience and previous event coordination and management experience. Roxanne has a bachelors' of science in Environmental Science, with a focus on water science. The collaboration between FLOW and the Sierra Club is considered a success by both parties. Sierra Club has donated the WARN Training Materials and Water Sentinel Chemistry Sampling kits as match. They are pleased to have so many new stream monitoring volunteers but were unfamiliar with locations that would be valuable to be sampled!

b. **Project Activities:** The list of our activities to date are:
   Preparing materials, presentations and kits for volunteer training;
   Broad public education on watershed issues (Water Alert Reporting Network WARN training (n=3);
   Hands-on Volunteer Training to Monitor Stream Water Quality
      macroinvertebrate volunteer training (n=8):
      water sentinel volunteer training (n=3);
   Train the Trainer Sessions (n=2);
   Stream Monitoring - sampling all four tributaries each at two locations in summer and fall (first sampling with trainer, second by volunteers themselves);
   Follow-up meeting with FLOW water stewards (November 2015);
   Data sharing and dissemination – presented information about the program and some initial data at the Olentangy Watershed Forum and the Water Management Association of Ohio fall conference.
   Survey of Current Volunteers (January 2016)
   Development of Water Stewards Facebook page (February 2016)

   The first WARN training was organized in the beginning of June, immediately after receiving the award, so we could maximize summer sampling. We advertised the program via emails and flyers posted at Whetstone Library, the Northmoor Park Kiosk, Upper Arlington Library and Lucky's Market. All of the participants were interested in hands on volunteer monitoring of the tributaries. We recently recruited quite a few volunteers from our public meeting in November 2015, and our Constant Contact messages online. After the end of sampling year we conducted additional WARN training in January to educate more volunteers and prepare additionally volunteers for next sampling year.

In order to increase the number of citizens more broadly educated about their streams in our watershed, we will organize at least two other WARN workshop in 2016. We will also create newsletter articles dedicated to our monitoring program, explaining what a healthy urban stream looks like as opposed to unhealthy urban stream signs. We will disseminate the newsletters with information about volunteer monitoring in the neighborhoods of our existing and proposed streams monitoring. Additionally, we will dedicate some of our monthly public meetings this year to talk about streams in our watershed.

According to the proposed project plan, we held two types of hands on volunteer trainings, one to monitor water chemistry (Water Sentinel) using Sierra Club methods and the other for macroinvetebrate sampling. We were able to train 16 volunteers and the majority of them wanted to perform both types of stream monitoring, which was close to our goal of 15 volunteers this year. Volunteers, after signing commitment forms to sample 3 times per year, received a sampling kit for macroinvetebrates (Appendix A) and a chemistry analysis kit via a loan agreement with the Sierra Club. Our volunteers sampled in groups of 2-3, all four tributaries twice in 2015, and each tributary was sampled at two points (see map, Attachment B). The volunteers were accompanied in the field during their first sampling by Roxanne, therefore gaining confidence in sampling by themselves (was proposed in project as one-on-one detailed training). At our November meeting volunteers expressed their confidence in sampling.

The group of volunteers were supported by our trainer – Roxanne Anderson. We proposed one trainer per ~10 volunteers. Although other FLOW volunteers were trained to be trainers (Zuzana Bohrerova, Laura Fay and Lisa Daris), they were not needed this year to help with volunteer support in the field.

c. Attendance – As stated before we expected a higher number of people attending our initial training with some people dropping out due to complexity but all the volunteers we trained have stayed with the program! We are working on increasing broader education and interest in stream water quality monitoring in our watershed, via different techniques than initially proposed, such as newsletter articles and public meetings.
Currently our volunteers are a diverse group of people including retirees, young professionals, OSU students, and Upper Arlington high school students along with their science teacher. Seven Science Classes at Upper Arlington High School are each monitoring a location on Turkey Run and helping us with other educational projects like videos, calendars and write ups for our Watershed Wiki!

d. **Highlights** – We were able to meet with our volunteer monitors in the fall after their last sampling and had a presentation for them of the program so far. We showed them the collected data, had chance to hear from them and meet some new prospective volunteers. We were happy to see that all our planned tributaries were sampled this year and the current volunteers are committed to continue the program in 2016 as well. We anticipate that in 2016 we will be able to expand our sampling to 5 additional tributaries of the lower Olentangy River.

e. **Educational objectives and survey:** We surveyed our current volunteers about their monitoring experiences from the first year. We got 5 responses back (50%). Some of the analysis of the survey are displayed in the table below. Answers were rated 1 - 5, where 5 means very positive answer/impacts and 1 the lowest or none satisfaction, and the weighted average is displayed.

Question/Rank	AVR
How much of an impact do you feel your volunteer work had?	3.3
How convenient were the volunteer training sessions at FLOW?	3.7
How useful were the volunteer training sessions at FLOW?	4.4
How easy was it to get along with the other volunteers at FLOW?	4.6
How friendly are the staff at FLOW?	4.7
How appreciated did your volunteer supervisors make you feel?	4.7
Overall, were you satisfied with your volunteer experience with FLOW, neither satisfied nor dissatisfied with it, or dissatisfied with it?	4.6
How likely are you to continue volunteering at FLOW in the future?	4.6
How likely are you to recommend FLOW to others as a place to volunteer?	4.7

From the comment it was evident that the three biggest negative comments are that volunteers felt isolated from each other (except of the groups sampling together), volunteers did not see what impact their data have and some volunteers would like to have more resources about sampling available to them.

To remedy some of these comments we created closed facebook group for our water steward where they can post photos, questions, communicate about their sampling. We also added additional resources to this website with more photos of macroinvertebrates, additional sampling techniques etc. We are regularly posting events related to the water steward group, such as training refresher and times to sample. We also posted summary of the volunteers' data for 2015.

f. Photographs


Field supplies for macroinvertebrate monitoring


"Train the Trainer" session on Adena Brook with Chris Skalski from Ohio EPA


Glen Echo Volunteers sampling macroinvertebrates in the summer with Roxanne

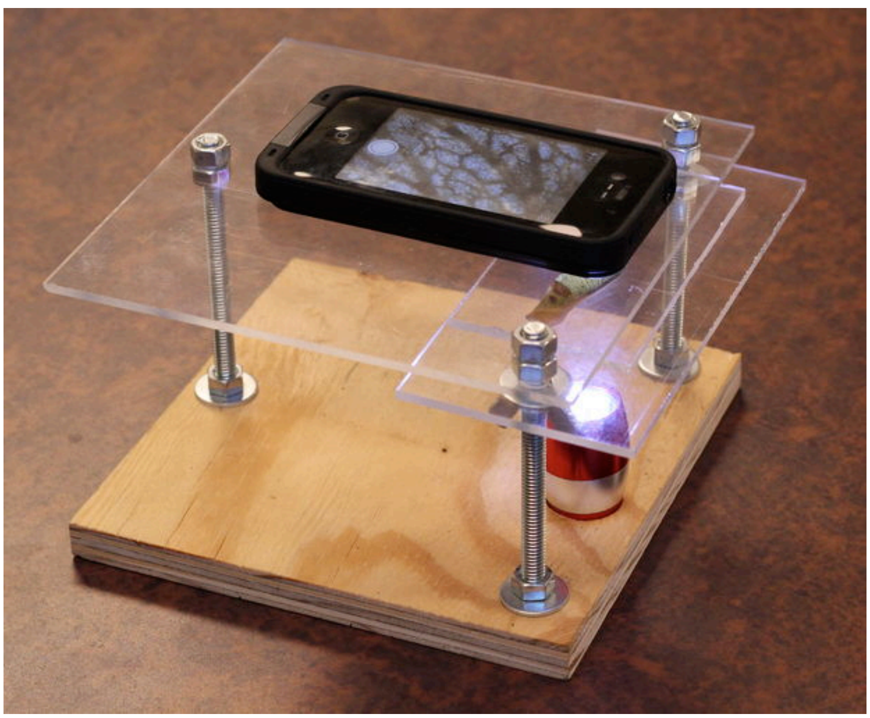

Volunteers sampling Kempton Run

Appendix A – List of supplies in macroinvetebrate kit, loaned to volunteer groups

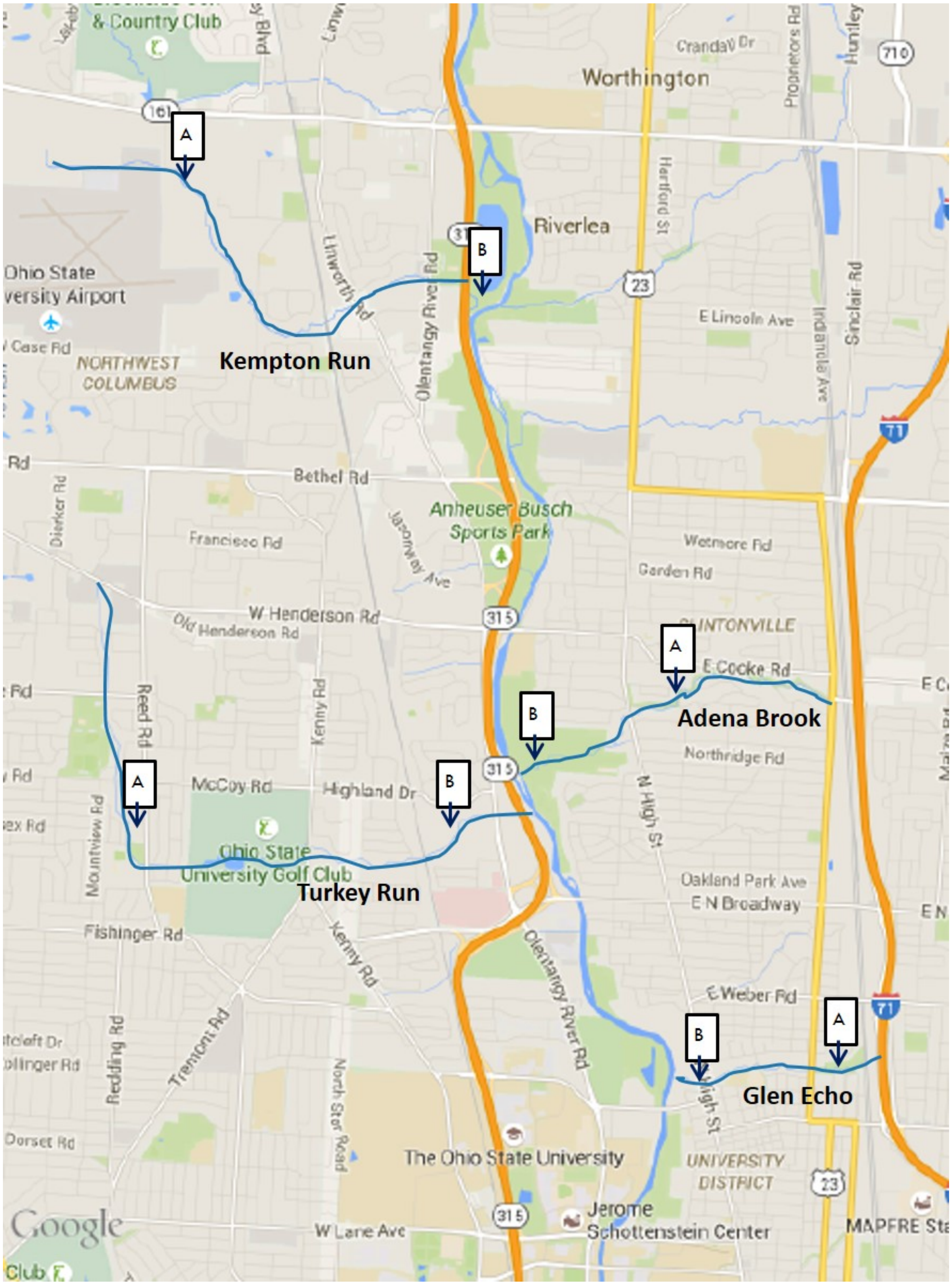
- clipboard
- 1 mechanical pencil
- ODNR Guide to Stream Quality Monitoring
- 1 pair of forceps
- 2 eyedroppers
- macroinvertebrate scoring sheets on waterproof paper
- D-shaped dip net
- 2 plastic sorting trays
- hand sanitizer
- 5 gallon bucket for storage and for water sampling

-100 foot measuring cord

- smartphone digital microscope with flashlight (shown below)



Appendix B – Current Sampling Locations



Appendix C. Survey Questions for Stream Quality Monitoring Volunteers

Stream Water Quality Monitoring Program Survey

1. What prompted you to participate in the program?

2. What supports, tools or practices have been most helpful in your time as a volunteer with the program?

3. Please provide a specific example of a hurdle you faced in volunteering with this program? What do you think would have helped you?

4. Have you encouraged others to get involved with the program? If so how?

5. What do you think would be the best way for the program to recruit more volunteers?

6. What suggestions would you offer volunteers new to the program?

7. If you could change one thing about the program, what would it be?

8. If you could change one thing about your experience with the program, what would it be?

9. How much of an impact do you feel your volunteer work had?

- A great deal of impact
- A lot of impact
- A moderate amount of impact
- A little impact

- Not any impact at all

10. How convenient were the volunteer training sessions at FLOW?

- Extremely convenient
- Quite convenient
- Moderately convenient
- Slightly convenient
- Not at all convenient

11. How useful were the volunteer training sessions at FLOW?

Extremely useful
- Quite useful
- Moderately useful
- Slightly useful
- Not at all useful

12. How easy was it to get along with the other volunteers at FLOW?

- Extremely easy
- Very easy
- Moderately easy
- Slightly easy
- Not at all easy

13. How friendly are the staff at FLOW?

- Extremely friendly
- Quite friendly
- Moderately friendly
- Slightly friendly
- Not at all friendly

14. How appreciated did your volunteer supervisors make you feel?

- Extremely appreciated
- Quite appreciated

- Moderately appreciated
- Slightly appreciated
- Not at all appreciated

15. Overall, were you satisfied with your volunteer experience with FLOW, neither satisfied nor dissatisfied with it, or dissatisfied with it?

- Extremely satisfied
- Quite satisfied
- Somewhat satisfied
- Neither satisfied nor dissatisfied
- Somewhat dissatisfied
- Quite dissatisfied
- Extremely dissatisfied

16. How likely are you to continue volunteering at FLOW in the future?

- Extremely likely
- Quite likely
- Moderately likely
- Slightly likely
- Not at all likely

17. How likely are you to recommend FLOW to others as a place to volunteer?

- Extremely likely
- Quite likely
- Moderately likely
- Slightly likely
- Not at all likely

18. Other thoughts or comments?

Appendix D. Materials Produced

Become a Water Steward! Flyer

Stream Quality Monitoring Protocol Document

Water Alert River Network (WARN) Training card and contact information for Franklin County

USGS Summer Intern Program

None.

Student Support					
Category	**Section 104 Base Grant**	**Section 104 NCGP Award**	**NIWR-USGS Internship**	**Supplemental Awards**	**Total**
Undergraduate	14	0	0	1	15
Masters	11	0	0	1	12
Ph.D.	6	0	0	1	7
Post-Doc.	0	0	0	0	0
Total	31	0	0	3	34

Notable Awards and Achievements

2013OH297B Mark Mitchell (Graduate Assistant) - American Scandinavian Foundation Research Award ($20,000)

2013OH297B Mark Mitchell (Graduate Assistant) - Graduate School Dean's Fellowship Award, University of Cincinnati ($20,000, full stipend)

2013OH297B Steven Doyle (Undergraduate student) - Steven Doyle – Selected as undergraduate mentee in GSUM/SUMR-UC program

2013OH297B Steven Doyle (Undergraduate student) - 2015 UC-wide McKibbin Award nominee – for outstanding Environmental Studies student

2014OH312B Graduate student, Mr. Aashish Shrestha was ranked first position under student poster competition categories in Ohio River Basin Consortium for Research and Education (ORBCRE) symposium

2013OH436O Dr. Chin-Min Cheng's project received: Undergraduate student fellowship - $3,500 College of Engineering Undergraduate Summer Research Fund

2015OH481O (Boccelli) Dr. Chen recently received an Honorable Mention (2nd place) in the 2016 University Council on Water Resources (UCOWR) Ph.D. Dissertation Award within the Natural Science and Engineering category.

2015OH481O (Boccelli) The current Ohio Water Resources Center funding also led to additional funding from the National Science Foundation for the project entitled "Data Assimilation and Forecasting for Real-Time Drinking Water Distribution System Modeling" (started Aug 2015, $336,000).

Publications from Prior Years

1. 2013OH300B ("Characterizing the influence of surface chemistry and morphology on biofilm formation of ceramic membranes in wastewater treatment") - Articles in Refereed Scientific Journals - JK Krinks, M Qiu, IA Mergos, LK Weavers, PJ Mouser, H Verweij (2015). Piezoceramic membrane with built-in ultrasonic defouling. Journal of Membrane Science. 494:130–135
2. 2013OH292B ("Source tracking of Microcystis blooms in Lake Erie and its tributaries") - Articles in Refereed Scientific Journals - TW Davis, GS Bullerjahn,T Tuttle, RM McKay, SB Watson (2015). Effects of Increasing Nitrogen and Phosphorus Concentrations on Phytoplankton Community Growth and Toxicity During Planktothrix Blooms in Sandusky Bay, Lake Erie. Environmental Science and Technology. 49:7197–7207
3. 2013OH435O ("") - Articles in Refereed Scientific Journals - P Wagha, G Parungaob, RE Violab, IC Escobar (2015). A new technique to fabricate high-performance biologically inspired membranes for water treatment. Separation and Purification Technology. 156(2):754–765