Water Resources Research Center Annual Technical Report FY 2010

Introduction

This report covers the period March 1, 2010 to February 28, 2011, the 45th year of the Massachusetts Water Resources Research Center (WRRC). The Center is under the direction of Dr. Paula Rees, who holds a joint appointment as Director of the WRRC and as Director of Education and Outreach of the Engineering Research Center for Collaborative Adaptive Sensing of the Atmosphere at the University of Massachusetts Amherst (UMass).

Several research projects were supported by the Massachusetts Water Resources Research Center. The Water Center funded a research project headed by Dr. Ellen Douglas of UMass Boston entitled "Developing a Physically-Based and Policy-Relevant River Classification Scheme for Sustainable Water and Ecosystem Management Decisions." Four graduate student projects were also funded: “An Assessment Methodology for Differential Impact on Environmental Justice Populations of Releases of Industrial Toxics to Water in Massachusetts” under PI Dr. Michael Ash of UMass Amherst; “Impact of the Hemlock Woolly Adelgid on the Water Cycle in New England: Differences in Hydrologic Fluxes Between Hemlock and Deciduous Forest Stands” under PI Dr. Andrew Guswa of Smith College; “Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing” under PI Qian Yu of UMass Amherst; and “Surface Water-Groundwater Interactions on the Deerfield River” under PI Dr. David Boutt of UMass Amherst. The “Acid Rain Monitoring Project,” led by WRRC Associate Director Marie-Françoise Hatte, was continued for another year in order to document trends in surface water acidification. One Technology Transfer award supported the Eighth Annual Water Resources Conference, organized by the Water Center on the University of Massachusetts Amherst campus.

Progress results for each project are summarized for the reporting year in the following sections.

Research Program Introduction

Eight research projects were conducted this fiscal year. One research project was funded through the USGS 104G program, and one research project received a no-cost extension for funds received through the previous USGS 104B program. Five new projects were funded through the 104B program and were completed this year. One research project was funded by the US Army Corps of Engineers for "Evaluation of Plausible Risk to Lake Superior Regulation and Upper Great Lakes Amid Climate Variability and Change" led by PI Dr. Casey Brown of UMass Amherst and administered by the Water Center.

Environmental Behaviors of Engineered Nanoparticles in Water

Basic Information

Title:	Environmental Behaviors of Engineered Nanoparticles in Water
Project Number:	2007MA73B
Start Date:	3/1/2007
End Date:	5/31/2010
Funding Source:	104B
Congressional District:	First
Research Category:	Water Quality
Focus Category:	Water Quality, Toxic Substances, Solute Transport
Descriptors:	
Principal Investigators:	Baoshan Xing, Baoshan Xing

Publications

1. Wang, X.L., J.L. Lu and B. Xing, 2008. Sorption of organic contaminants by carbon nanotubes: Influence of adsorbed organic matter. Environ. Sci. Technol. (42)3207-3212
2. Iorio, M, B. Pan, R. Capasso and B. Xing, 2008. Sorption of phenanthrene by dissolved organic matter and its complex with aluminum oxide nanoparticles. Environ. Pollut. (in press).
3. Pan, B., D.H. Lin, H. Mashayekhi and B. Xing, 2008. Adsorption and hysteresis of bisphenol A and 17 -ethinyl estradiol on carbon nanomaterials. Environ. Sci. Technol. (accepted).
4. Lin, D.H. and B. Xing, 2008. Phytotoxicity of zno nanoparticle: inhibition of ryegrass growth. Preprints of Environ. Chem. Div. Extended Abstracts of the 235th ACS national meetings. 48(1): 276-280
5. Mashayekhi, H., W. Jiang and B. Xing, 2008. Metal oxide nanoparticles show toxicity to bacteria. Preprints of Environ. Chem. Div. Extended Abstracts of the 235th ACS national meetings. 48(1): 326-328.
6. Pan, B. and B. Xing, 2008. Sorption of endocrine disrupting chemicals on carbon nano materials. Preprints of Environ. Chem. Div. Extended Abstracts of the 235th ACS national meetings. 48(1): 206-208.
7. Wang, X.L. and B. Xing, 2008. Dissolved organic matter affects sorption of organic contaminants on carbon nanotubes. Preprints of Environ. Chem. Div. Extended Abstracts of the 235th ACS national meetings. 48(1): 220-223.
8. Wang, Z.Y., Z.J. Tian, F.M. Li, and B. Xing, 2008. Effects of five nanomaterials on gymnodinium breve. Preprints of Environ. Chem. Div. Extended Abstracts of the 235th ACS national meetings. 48(1): 315-319.
9. Yang, K. and B. Xing. 2009. Adsorption of fulvic acid by carbon nanotubes from water. Environ. Pollut. 157: 1095-1100.
10. Lin, D.H., N. Liu, K.Yang, L.Z. Zhu, Y. Xu and B. Xing. 2009. The effect of ionic strength and pH on the stability of tannic acid-facilitated carbon nanotube suspensions. Carbon, 47: 2875-2882.
11. Yang, K., D.H. Lin and B. Xing. 2009. Interactions of humic acid with nanosized inorganic oxides. Langmuir, 25(6): 3571-3576.
12. Yang, K. and B. Xing. 2009. Sorption of phenanthrene by humic acid-coated nanosized TiO2 and ZnO. Environ. Sci. Technol. 43(6): 1845-1851.

13. Ghosh, S., Hamid Mashayekhi, P. Bhowmik and B. Xing. 2010. Colloidal stability of Al2O3 nanoparticles as affected by coating of structurally different humic acids. Langmuir, 26 (2): 873–879.
14. “Influence of Natural Organic Matter and Synthetic Polyelectrolytes on Colloidal Behavior of Metal Oxide Nanoparticles” Ph.D. Dissertation by Saikat Ghosh. UMass.
15. Mashayekhi, H. S. Ghosh and B. Xing. “Aggregation Behavior of C60 Fullerene Water Suspension in the Presence of Natural Organic Materials.” International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 36.
16. Iorio, M., A. De Martino, R. Capasso and B. Xing. “Screening of nine different nanoparticles for the removal of MCPA from water.” International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 49.
17. Zia, J., D. Amarasiriwardena and B. Xing. “Investigation of the adsorption characteristics of Sb, Cd, and pB by nano- and micro-particle titania.” International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 51.

Methodology

Batch sorption techniques, DSL, liquid scintillation counting, HPLC detection, TEM, AFM and SEM examinations.

Principal Findings and Significance

The colloidal stability of three structurally different humic acid (HA) coated Al_2O_3 nanoparticles (HAs-Al_2O_3 NPs) was studied in the presence of Ca^{2+}. HAs were obtained after sequential extractions of Amherst Peat Soil. Highly polar HA1-coated Al_2O_3 NPs exhibited strong aggregation in the presence of Ca^{2+}. HA3 and HA7-coated NPs showed weaker aggregation due to their increased aliphaticity and low polarity. HA7-Al_2O_3 NPs displayed the weakest aggregation behavior even at relatively high Ca^{2+} concentration. The inverse stability ratio (α=1/W) was the lowest for HA7-Al_2O_3 NPs reflecting that strong steric stabilization enhanced colloidal stability. Atomic force microscopy (AFM) of pure Al_2O_3 NPs on Ca^{2+}-saturated mica clearly demonstrated significant aggregation following classical DLVO model for hard spheres. On the contrary, weakly polar HA fraction produced approximately 10 nm thick corona of adsorbed layer around each Al_2O_3 NP, thus, stabilizing coated NPs suspension through steric effect. Under alkaline conditions and at low ionic strength adsorbed HA chains swelled, increasing their osmotic potential, which in turn resulted in stabilization of the colloids. Inherent structural variations of NOMs played a significant part in colloidal stability of the coated NPs. Thus, development of sterically stabilized NPs may have potential application for water remediation in marine and high salinity conditions.

Publications and Conference Presentations

Several Platform and poster presentations will be given at the International Conference of Environmental Application and Implication of Nanotechnology, June 9-11, 2009, Amherst, MA.

Articles in Refereed Scientific Journals

Ghosh, S., Hamid Mashayekhi, P. Bhowmik and B. Xing. 2010. Colloidal stability of Al_2O_3 nanoparticles as affected by coating of structurally different humic acids. Langmuir, 26 (2): 873–879.

Yang, K. and B. Xing. 2009. Sorption of phenanthrene by humic acid-coated nanosized TiO_2 and ZnO. Environ. Sci. Technol. 43(6): 1845-1851.

Yang, K., D.H. Lin and B. Xing. 2009. Interactions of humic acid with nanosized inorganic oxides. Langmuir, 25(6): 3571-3576.

Lin, D.H., N. Liu, K.Yang, L.Z. Zhu, Y. Xu and B. Xing. 2009. The effect of ionic strength and pH on the stability of tannic acid-facilitated carbon nanotube suspensions. Carbon, 47: 2875-2882.

Yang, K. and B. Xing. 2009. Adsorption of fulvic acid by carbon nanotubes from water. Environ. Pollut. 157: 1095-1100.

Dissertations

"Influence of Natural Organic Matter and Synthetic Polyelectrolytes on Colloidal Behavior of Metal Oxide Nanoparticles" Ph.D. Dissertation by Saikat Ghosh. UMass.

Conference Proceedings

Mashayekhi, H. S. Ghosh and B. Xing. "Aggregation Behavior of C60 Fullerene Water Suspension in the Presence of Natural Organic Materials." International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 36.

Iorio, M., A. De Martino, R. Capasso and B. Xing. “Screening of nine different nanoparticles for the removal of MCPA from water.” International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 49.

Zia, J., D. Amarasiriwardena and B. Xing. “Investigation of the adsorption characteristics of Sb, Cd, and pB by nano- and micro-particle titania.” International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 51.

Student Support, Department of Plant, Soil and Insect Sciences

- Mr. Hamid Mashayekhi, PhD Candidate
- Mr. Saikat Ghosh, PhD Candidate
- Miss Wei Jiang, PhD Candidate

Bacterial Toxicity of Oxide Nanoparticles and Their Adhesion

Basic Information

Title:	Bacterial Toxicity of Oxide Nanoparticles and Their Adhesion
Project Number:	2009MA177B
Start Date:	4/1/2009
End Date:	3/31/2010
Funding Source:	104B
Congressional District:	First
Research Category:	Water Quality
Focus Category:	Toxic Substances, Water Quality, Geochemical Processes
Descriptors:	None
Principal Investigators:	Baoshan Xing

Publications

1. Jiang, W., H. Mashayekhi and B. Xing. 2009. Bacterial toxicity comparison between nano- and micro-scaled oxide particles. Environ. Pollut. 157:1619-1625.
2. Jiang, W., H. Mashayekhi and B. Xing "Bacterial toxicity of oxide nanoparticles and their adhesion to bacteria cell walls". International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 27.
3. Wang, H., R. Wick and B. Xing . "Toxicity of nanoparticulate and bulk ZnO, Al2O3 and TiO2 to the nematode Caenorhabditis Elegans". International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 31.
4. Jiang, W. and B. Xing. "Behavior of nanoparticles at the bacteria-water interface" The 7th Annual Massachusetts Water Resources Conference, Amherst, April 8, 2010. Abstract book, p. 25.
5. Jiang, W., H. Mashayekhi and B. Xing. 2009. Bacterial toxicity comparison between nano- and micro-scaled oxide particles. Environ. Pollut. 157:1619-1625.
6. Jiang, W., H. Mashayekhi and B. Xing "Bacterial toxicity of oxide nanoparticles and their adhesion to bacteria cell walls". International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 27.
7. Wang, H., R. Wick and B. Xing . "Toxicity of nanoparticulate and bulk ZnO, Al2O3 and TiO2 to the nematode Caenorhabditis Elegans". International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 31.
8. Jiang, W. and B. Xing. "Behavior of nanoparticles at the bacteria-water interface" The 7th Annual Massachusetts Water Resources Conference, Amherst, April 8, 2010. Abstract book, p. 25.

Problem and Research Objectives

Oxide nanoparticles (NPs) are widely used, and they are potentially toxic. The goal of this work was to evaluate the toxicity of several engineered oxide NPs to common bacteria species and the adhesion of NPs to the bacteria surface.

Methodology

Batch experiments, FTIR, Characterization of Nanoparticles, Toxicity evaluation, AFM and TEM imaging.

Principal Findings and Significance

Toxicity of nano-scaled aluminum, silicon, titanium and zinc oxides to bacteria (*Bacillus subtilis, Escherichia coli* and *Pseudomonas fluorescens*) was examined and compared to that of their respective bulk (micro-scaled) counterparts. All nanoparticles but titanium oxide showed higher toxicity than their bulk counterparts. Toxicity of released metal ions was differentiated from that of the oxide particles. ZnO was the most toxic among the three nanoparticles, causing 100% mortality to the three tested bacteria. Al_2O_3 nanoparticles had a mortality rate of 57% to *B. subtilis*, 36% to *E. coli*, and 70% to *P. fuorescens*. SiO_2 nanoparticles killed 40% of *B. subtili*s, 58% of *E. coli*, and 70% of *P. fluorescens*. TEM images showed attachment of nanoparticles to the bacteria, suggesting that the toxicity was affected by bacterial attachment. Bacterial responses to nanoparticles were different from their bulk counterparts; therefore nanoparticle toxicity mechanisms need to be studied thoroughly.

Publications and Conference Presentations

Articles in Refereed Scientific Journals

Jiang, W., H. Mashayekhi and B. Xing, 2009. Bacterial toxicity comparison between nano- and micro-scaled oxide particles. Environ. Pollut. 157:1619-1625.

Conference Proceedings

Jiang, W., H. Mashayekhi and B. Xing "Bacterial toxicity of oxide nanoparticles and their adhesion to bacteria cell walls". International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 27.

Wang, H., R. Wick and B. Xing . "Toxicity of nanoparticulate and bulk ZnO, Al_2O_3 and TiO_2 to the nematode *Caenorhabditis Elegans".* International Conference on the Environmental Implications and Applications of Nanotechnology, Amherst (MA), June 9-11, 2009, p. 31.

Jiang, W. and B. Xing. "Behavior of nanoparticles at the bacteria-water interface" The 7th Annual Massachusetts Water Resources Conference, Amherst, April 8, 2010. Abstract book, p. 25.

Student Support

Mr. Hamid Mashayekhi and Miss Wei Jiang, Ph.D. Department of Plant, Soil & Insect Sciences

Notable Achievements and Awards

One graduate student, Wei Jiang, won a first place for her poster presentation at the "Water Dependencies in New England", the 6th Annual Conference: http://www.umass.edu/psis/news/ne_water_conf.html

Impact of Nanoparticles on the Activated Sludge Process: Effects on Microbial Community Structure and Function

Basic Information

Title:	Impact of Nanoparticles on the Activated Sludge Process: Effects on Microbial Community Structure and Function
Project Number:	2009MA178B
Start Date:	4/1/2009
End Date:	3/31/2010
Funding Source:	104B
Congressional District:	5th
Research Category:	Not Applicable
Focus Category:	Toxic Substances, Wastewater, None
Descriptors:	None
Principal Investigators:	Juliette Rooney-Varga, Deepankar Goyal

Publications

1. Goyal, D., G. Doyle, X. J. Zhang, J. N. Rooney-Varga. Impact of Multi-Walled Carbon Nanotubes on the Structure of Activated Sludge Microbial Communities. Eastern New England Biological Conference, Lowell MA, April 2009.
2. Goyal, D., G. Doyle, X. J. Zhang, J. N. Rooney-Varga. Impact of Multi-Walled Carbon Nanotubes on the Structure of Activated Sludge Microbial Communities. Eastern New England Biological Conference, Lowell MA, April 2009.

Problem and Research Objectives

Nanotechnology, or the ability to create and use materials at the scale of 1 to 100 nanometers, is a rapidly expanding sector that is generating materials with unique physical and chemical properties. In particular, carbon nanotubes (CNTs) are known for their unique mechanical, electronic, and biological properties and have far-reaching potential applications as components of personal care products, pharmaceuticals, electronic devices, energy storage devices, stains and coatings, and new environmental clean-up technologies (Masciangioli and Zhang 2003; Boczkowski and Lanone 2007; Chen 2007; Erdem 2007; Rivas et al., 2007; Kislyuk and Dimitriev 2008; Prato et al., 2008; Theron et al., 2008). Massachusetts is poised to be a leader in nanotechnology research and development and this sector is expected to be a major component of the Commonwealth's economy for the foreseeable future. However, while the potential for nanotechnology is vast, relatively little is known about the health and environmental risks posed by nanomaterials (Colvin 2003). Indeed, those watching the industry have commented that concern over unknown risks of nanomaterials is a major determinant of the future success of nanotechnology (Colvin 2003).

Through nanomanufacturing and widespread use of nanomaterials, CNTs and other nanomaterials will inevitably be released into wastewater streams and enter wastewater treatment plants (Wiesner 2006). All publicly owned wastewater treatment facilities rely on the 'activated sludge process,' which relies on controlled microbial degradation of waste materials to remove chemical and biological contaminants (Wagner et al., 2002). However, little is known about how CNTs will affect the complex microbial communities that are responsible for the activated sludge process and whether microorganisms are capable of removing them. Effluent from wastewater treatment plants ultimately is released to the environment, where it can impact aquatic ecosystems and drinking water. Any toxicity to microorganisms exhibited by CNTs has the potential to dramatically reduce the efficacy of the activated sludge process, resulting in the release of untreated sewage, pathogenic microbes, and CNTs into the environment. Shock-loading of other contaminants has shown that treatment performance can be affected for weeks or months, resulting in a reduction in treatment efficiency, environmental release of toxic contaminants, and operation problems that may require months to recover (Boon et al., 2003; Henriques et al., 2007). In addition, the ability of CNTs to strongly adsorb organic matter can reduce the bioavailability and, therefore, microbial degradation of organic pollutants, which would then effectively bypass the treatment process.

The composition and function of activated sludge microbial communities has received considerable attention, although the function of many specific phylogenetic groups and the factors that control are not yet well understood. In broad terms, activated sludge contains microbial eukaryotes ("microeukaryotes"), including protozoa, fungi, and metazoans, as well as a wide diversity of bacteria responsible for varied metabolic functions, including oxidation of organic compounds and removal of nitrogenous pollutants and phosphates (Weber et al., 2007). Within this microbial community, many complex ecological interactions are thought to be necessary for the effective functioning of activated sludge. For example, ciliated protozoa and fungi have been found to form tree-like and filamentous colonies, respectively, that form a back-bone for bacterial colonization, resulting in the production of flocs, which readily settle out of the liquid phase and are collected for effective removal from treated wastewater (Weber et al., 2007). Both bacteria and microeukaryotes are likely to contribute to the formation of extracellular polymeric substances (EPS), which are high molecular weight compounds with adhesive properties that are critical to the formation and integrity of flocs (and biofilms more generally) (Raska et al., 2006; Weber et al., 2007). In addition, specific taxonomic groups of bacteria are known to carry out key functions in activated sludge. For example, members of the *Planctomycetes* are responsible for anaerobic ammonium oxidation; several lineages within the kingdom *Euryarchaea* produce methane; members of the genus *Nitrospira* oxidize nitrite to nitrate (Juretschko et al., 2002); *Actinomycetes* may contribute to the production of foam and reduce the quality of effluent; and members of the *Chloroflexi* have been associated with bulking events (Kragelund et al., 2007). Thus, analysis of microbial community composition can provide meaningful insight into various activated sludge functions, as well as the factors that control them (Liu et al., 1997; Forney et al., 2001).

Relatively little is known about the potential toxicity of CNTs to activated sludge microorganisms and studies on pure cultures or defined mixed cultures have yielded conflicting results. There is strong evidence for CNT toxicity to pulmonary cells, as well as potential toxicity to epithelial, brain, and liver cells (Lam et al., 2006; Smart et al., 2006; Warheit 2006). Single-walled CNTs (SWCNTs) have been reported to be highly toxic to *Escherichia coli* str. K12 cells that come in direct contact with them (Kang et al., 2007). (Ghafari et al., 2008) found a moderate impact of SWCNTs on *E. coli*-gfp viability, although they did not differentiate between planktonic cells and those in contact with SWCNT aggregates. Interestingly, they found that *Tetrahymena thermophila,* a ciliated protozoan that is an important member of wastewater treatment microbial communities, ingested CNTs. As a result, the protozoan's ability to ingest and digest bacterial cells was impeded, suggesting a negative impact on an important function of these protozoa, namely bacterivory (Ghafari et al., 2008). Conversely, the CNTs may also have a positive impact on activated sludge processes, as they caused an increase in the production of exudates by ciliates, which may benefit floc formation and, therefore, sludge settleability.

While pure culture studies have shown that nanomaterials can act as antimicrobial agents, the complexity of the activated sludge community make it unlikely to respond to CNTs in the same manner as simple pure culture systems. Our objective was to use state-of-the-art molecular techniques to determine the impact of CNTs on microbial community dynamics in batch reactors that model the activated sludge process.

Methodology

Experimental set-up

Fresh activated sludge was collected from an aeration basin at the Lowell Regional Wastewater Treatment Facility, Lowell. This facility is designed to treat primarily municipal wastewater through conventional primary and secondary treatment processes. Whole unscreened samples were transported to the laboratory and processed within 30 minutes of sample collection. Experimental conditions for batch-scale reactor studies were previously described by Yin et al., (2009). In order to distinguish between effects of CNTs and potential toxic effects of impurities associated with them (such as amorphous carbon and metal catalysts), triplicate CNT-exposed reactors were compared to triplicate reactors exposed to impurities alone. CNTs used in the current study consisted of >90% pure CNTs (Sigma-Aldrich, Inc., St. Louis MO) characterized by Raman spectroscopy (Table 1). Reactors were filled with 2 L of fresh activated sludge, with an initial soluble chemical oxygen demand (sCOD) of 20 mg L^{-1} from the aeration basin effluent (Yin et al., 2009). The sludge was fed with peptone and aerated prior to and during the experiment as described by Yin et al., (Yin et al., 2009). Sub-samples for microbial community analysis were taken aseptically immediately after adding CNTs or impurities (T_0), at 1.25 hr (T_1) after initial exposure, and at 5 hr (T_4). The samples were placed in cryovials, and stored at -80°C until further processing.

DNA extraction and analysis
Genomic DNA from was extracted and purified from 400 µL sub-samples of sludge using the FastDNA Spin kit for Soil (MP Biomedicals Inc., Solon, OH). ARISA-PCR was performed as previously described (Fisher and Triplett 1999), with minor modifications. Reaction mixtures contained 1× AmpliTaq PCR buffer (Applied Biosystems, Inc., Carlsbad, CA), 2.5 mM $MgCl_2$, 400 ng μL^{-1} bovine serum albumin (BSA), 200 µM each dNTP, 400 nM each primer, 2.5 U of *Taq* DNA polymerase, and 1, 5, 10, or 20 ng of genomic DNA in a final volume of 50 µL. The primers used were 1392F (5'-G [C/A] ACACACCGCCCGT-3') and 23SR (5'GGGTT[C/G/T] CCCCATTC[A/G]G-3'). The 5'end of primer 1392F was labeled with 6-carboxyfluorescein (6-FAM). The following thermal profile was used for PCR: denaturation at 94°C for 3 min, followed by 30 cycles of amplification at 94° C for 30 s, 56° C for 30 s, and 72° C for 45 s, followed by a final extension of 72° C for 7 min. PCR products were analyzed by electrophoresis in 1% agarose gels (Ausubel et al., 1997) and were purified using QiaQuick PCR Purification Kits (Qiagen, Inc., Valencia CA).

20 ng each purified PCR product were lyophilized and subjected to automated capillary electrophoresis (CE) analysis in conjunction with a 50 – 1200 bp size standard labeled with LIZ™ (Applied Biosystems, Inc.) at the Center for AIDS Research, UMass Medical School, Worcester MA. ARISA conditions were optimized by comparing profiles generated from multiple DNA template amounts (1, 5, 10, or 20 ng per 50 µL PCR) and PCR product amounts (5, 10, or 20 ng PCR product per well). Comparison of these conditions indicated that the highest diversity (species richness and evenness) and signal to noise ratios were achieved using 1 ng DNA template DNA for PCR and 20 ng PCR product for CE analyses, which were used in subsequent analyses.
ARISA profiles were analyzed using PeakScanner software (Applied Biosystems Inc.) and processed as described by Brown et al., (Brown et al., 2005). The programs Interactive and Automatic Binner were used to bin peaks, with a window size (WS) of 3 bp and a shift value (Sh) of 0.1 (Ramette 2009). Peak areas were normalized to total peak area per sample and peaks representing <1% total peak area for a given sample were considered indistinguishable from background and removed from the analysis. Data visualization and ordination analyses were conducted using the packages Ecodist (Goslee and Urban 2007) and Vegan (http://vegan.r-forge.r-project.org/) in the R statistical programming environment (Goslee and Urban 2007). Pairwise Bray-Curtis distances between samples were calculated using the Ecodist package and a hierarchical clustering algorithm with average linkage clustering were used to construct a dendrogram depicting relationships among the samples' ARISA profiles. Correspondence analysis (CA), which assumes a unimodal relationship between relative abundance (i.e., normalized peak area) and ordination axes, was used to analyze relationships between samples. The R package Vegan was used to determine whether CA ordination axes were correlated with environmental variables. The latter included the experiment from which samples were analyzed (E1 for the experiment comparing CNTs to CNT-associated impurities, conducted on June 28, 2007; E2 for the experiment comparing CNT-associated impurities to a control conducted on July 19, 2007); time elapsed from the initiation of the experiment to sampling (0, 1.25, or 5 hours); and treatment (CNTs, associated impurities, or feed alone). "Dummy" variables were assigned for categorical variables and set to 0 or 1 depending on the presence of a given variable. The "envfit" goodness of fit test with 1000 permutations was used to assess the fit of environmental variables to ordination axes.

In order to determine the phylogenetic identity of dominant community members, as detected by ARISA, phylogenetic analysis of 16S rRNA genes contiguous with fragments analyzed in ARISA was used (Brown et al., 2005). DNA amplicons

containing partial 16S rRNA genes and associated intergenic spacer (IGS) regions were generated from selected activated sludge genomic DNA samples using primers 338F and 23SR (5'GGGTT[C/G/T] CCCCATTC[A/G]G-3') (Amann et al., 1990; Brown et al., 2005). The resulting amplicons were cloned using the TOPO TA Cloning Kit for Sequencing with One Shot® TOP10 Chemically Competent *E. coli*, as described by the manufacturer (Invitrogen Corp., Carlsbad, CA). 90 cloned inserts were analyzed using ARISA, as described above, except that the template DNA for PCR consisted of *E. coli* clone cell lysates (obtained by suspending individual colonies in 0.1 M Tris-Cl, pH 8.0, and incubating them at 99˚ C for 2 minutes). ARISA peaks from cloned inserts were considered to match OTUs from environmental community ARISA patterns if their peak size was placed within the same 3 bp bin as a given OTU from environmental samples.
At least one cloned insert representative of each ARISA OTU was sequenced in both directions by Beckman Coulter Genomics Inc. (Danvers MA, USA) with M13 primers. Vector and primer sequences were trimmed, trimmed sequences were aligned to the Silva database, and phylogenetic relationships among aligned sequences and their 40 nearest neighbors in the Silva database were analyzed using ARB (Ludwig et al., 2004; Pruesse et al., 2007). Trimmed sequences were deposited in GenBank under accession numbers HM205112 - HM205114.

Principal Findings and Significance

Results

Effects of CNTs and their associated impurities

Analysis of ARISA profiles revealed several differences between bacterial community structure in batch reactors exposed to CNTs for five hours when compared to those exposed to associated impurities alone. For example, the relative peak areas of dominant OTUs represented by peaks 419, 794, and 839 bp were significantly different in communities exposed to CNTs vs. those exposed to CNT-associated impurities (Fig. 1). Similarly, a Chi-square goodness-of-fit test of correspondence analysis (CA) axes revealed that the effect of CNTs on community structure was significant (p =0.043), while exposure to impurities alone was not (p= 0.604). In order to assess the effect of CNTs without interference from the strong effects of time and experiment, CA ordination was repeated with only the time T4 samples from the experiment comparing CNTs to impurities alone (E1). A statistically significant effect of CNTs was observed ($p<0.001$), while a similar analysis of the effects of impurities alone (CA with experiment E2, time T4 samples) revealed no effect ($p=0.316$), as was also evident from direct inspection of ARISA profiles (Fig. 1). Samples taken after only 1.25 hours exposure (time T1) revealed no clear differences in ARISA profiles between either CNT- and impurities-exposed reactors or between reactors exposed to impurities and control reactors), indicating that exposure for 1.25 hours was insufficient for CNT effects to be detected via the approach used here.

Both hierarchical clustering and correspondence analysis (CA) of all samples revealed strong effects of the amount of time elapsed prior to sampling (0, 1.25, or 5 hours) and the date of the experiment (Fig. 2). Baseline (T_0) communities for E1 and E2 were fairly similar. However, these communities diverged substantially over the short experimental time period of five hours, with the resulting communities sharing only 14/29 total OTUs and 4/9 total "dominant" (considered here to be those with average relative peak areas > 5%) OTUs.

Three of the OTUs found in environmental samples were identified among the 90 cloned inserts analyzed here. These included peaks corresponding to 419, 740, and 812 bp (Fig. 1). Phylogenetic analysis placed these OTUs within the families *Sphingomonadaceae* (419 bp) and *Cytophagaceae* (740 bp), and the genus *Zoogloea*

(812 bp). Two representative of OTU 812 were sequenced and found to be identical. The closest relatives of the sequences representing OTUs 419, 740, and 812 were: an uncultivated *Sphingomonadaceae* bacterium from snow (97.1% similarity); an uncultivated *Cytophagaceae* bacterium from activated sludge (89.5% similarity); and *Zoogloea resiniphila*, a denitrifier isolated from activated sludge (99.8% similarity).

Discussion

While CNTs have the potential to be highly toxic to microbial cells, their impact under the complex abiotic and biological conditions found in environmental microbial communities remains poorly understood. The current study revealed changes in microbial community structure in activated sludge batch reactors exposed to CNTs, while no effects of CNT-associated impurities were detected. Yin et al. (2009) analyzed bulk parameters and performance from the CNT-exposed batch reactors described here and similarly found that CNTs, but not their associated impurities, had several effects on sludge performance. These effects included: increased organic carbon removal primarily through organic carbon adsorption; less negative surface charges of activated sludge flocs; and improved sludge settleability (Yin et al., 2009). Other parameters such as pH, dissolved oxygen, specific resistance to filtration, and relative hydrophobicity were not significantly impacted (Yin et al., 2009). These findings suggest that CNTs impacted community structure through toxicity to some community members, by reducing organic carbon bioavailability, and/or by altering floc properties.

The fact that CNT effects on microbial community structure were detected was especially interesting given that, unlike some previous studies, the experimental conditions used did not maximize CNT-cell interactions. For example, an assay for cytotoxicity developed by Kang et al., (2007) relies on drawing planktonic cells onto a filter that is coated with nanoparticles and observing the resulting effects on cellular membrane integrity over time. Under these conditions, direct cell-nanoparticle contact is artificially induced and CNTs demonstrated high levels of toxicity to Gram-negative (*Escherichia coli* and *Pseudomonas aeruginosa*) and, to a lesser extent, Gram-positive (*Staphylococcus epidermis* and *Bacillus subtilis*) cells (Kang et al., 2009). In contrast, here, CNTs were added to activated sludge bioreactors in suspension, making CNT-cell contact much less likely. In addition, the presence of extracellular polymeric substances (EPS) and high concentrations of DOC in the batch reactors used here may have mitigated CNT toxicity to some extent, as CNTs are likely to become embedded in EPS and thereby prevented from coming in direct contact with cell membranes (Neal 2008; Luongo and Zhang 2010). Lastly, the exposure time was kept short in order to avoid confounding effects of starvation and/or accumulation of waste products in closed-system batch reactors. Despite the use of short incubation times, changes in community structure with both CNT exposure and time over the course of the experiment were found (Fig. 1 and 2). Previous studies have shown that cellular inactivation increased with time of exposure (Kang et al., 2009), indicating that use of longer incubation times in continuous reactors may increase effects of CNTs on community structure.

Phylogenetic analysis of cloned inserts that were matched to ARISA peaks revealed the presence of three phylogenetic groups that are responsible for important functions in activated sludge communities, including the members of the families *Sphingomonadaceae* (OTU 419) and *Cytophagaceae* (OTU 740) and the genus *Zoogloea* (OTU 812) (Manz et al., 1996; Neef et al., 1999; Juretschko et al., 2002; Wagner et al., 2002; Li et al., 2008). Of these, the sphingomonad (OTU 419) showed a trend of decreased relative peak intensity with exposure to CNTs (Fig. 1), indicating an adverse impact of CNTs on this group compared to other community members. Within wastewater treatment microbial communities, sphingomonads are thought to have wide metabolic diversity, are capable of degrading some xenobiotics, and contribute to the formation of flocs (Neef et al., 1999; Wagner et al., 2002). Although directly measuring these parameters was beyond the scope of the current study, the potential for negative impacts on CNTs on these microbial functions deserves further attention.

Differences in the 'baseline' (T_0) community structure from one sampling date to another corroborate results obtained by Wittebolle et al., (2005), who observed that large community shifts occurred over a period as short as a few days in a given wastewater treatment plant and that community structure was related to performance of biological treatment. These findings underscore the need to analyze microbial community structure when assessing the effects of emerging contaminants on environmental systems, as differences in the starting community composition may alter the observed impacts on community performance.

In conclusion, our results indicate that the structure of activated sludge microbial communities is impacted by exposure to CNTs, even when such exposure is limited to a short time period, and that these effects were not due to impurities associated with CNTs. Community shifts found here indicated that CNTs differentially affect microbial species, as has been found under pure culture conditions (Kang et al., 2009). These results raise the concern of CNT impact on biological functions carried out by the activated sludge process.

Table 1. Characteristics of CNTs used in the current study.

Purity	
Carbon nanotubes	>90%
Single-walled nanotubes	>50%
Impurities	
Amorphous carbon	<5%
Co	0.6%
Mg	1.2%
Mo	0.1%
Silicates	0.1%
Average outside diameter	1–2 nm
Density	1.7–2.1
Length	5–15 um
Specific surface area	>400 m2/g

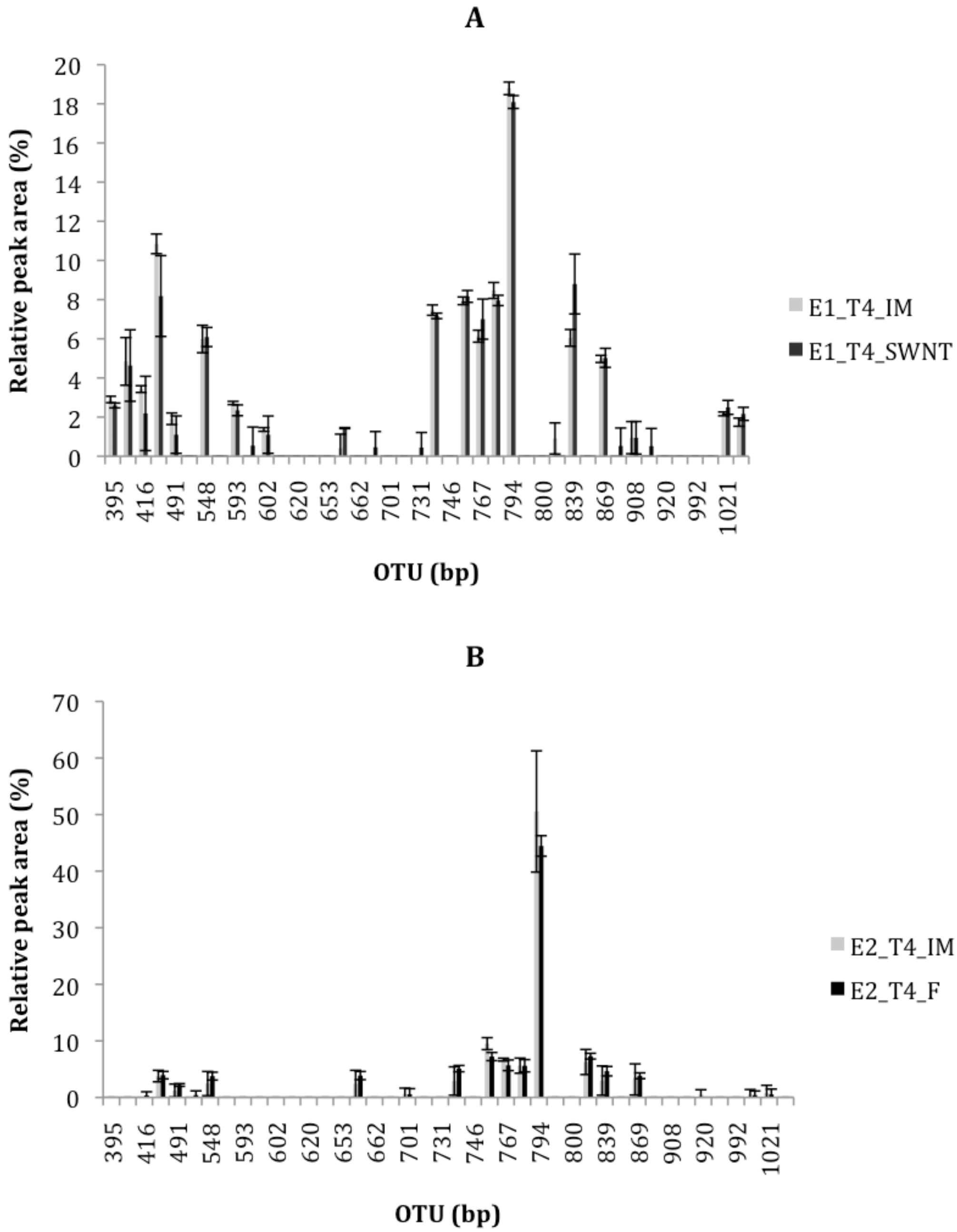

Figure 1. ARISA profiles of activated sludge bacterial communities exposed to CNTs, their associated impurities, or synthetic feed alone at the end of the experiments (T4). Comparisons were made between CNT- and impurities-exposed (IM) reactors during one experiment (designated E1; panel A) and between impurities-exposed and control reactors receiving feed alone (F) in a second experiment (E2; panel B). Means and standard deviations of relative peak areas from triplicate batch reactors are shown.

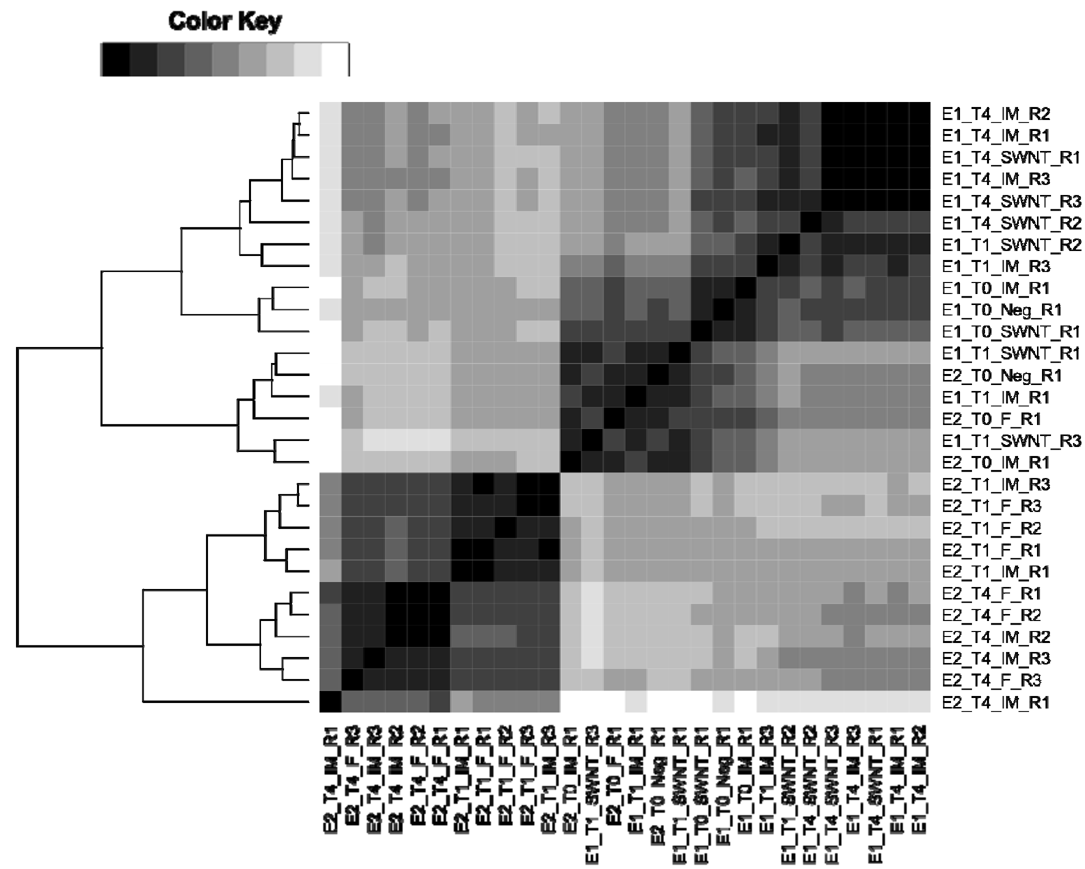

Figure 2. Hierarchical clustering analysis and heatmap of Bray-Curtis distances among samples taken from the first and second experiments (E1 and E2, respectively), at times 0, 1.25 hours, and 5 hours (T0, T1, and T4, respectively), and exposed to CNTs, impurities, or feed alone (SWNT, IM, or F, respectively).

Publications and Conference Presentations

Goyal, D., J. X. Zhang, J. N. Rooney-Varga. 2010. Impacts of single-walled carbon nanotubes on microbial community structure in activated sludge. *Submitted.*

Goyal, D., G. Doyle, X. J. Zhang, J. N. Rooney-Varga, 2009. Impact of Multi-Walled Carbon Nanotubes on the Structure of Activated Sludge Microbial Communities. Eastern New England Biological Conference, Lowell MA, April 2009.

Student Support

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Gregory Doyle, B.S., Biological Sciences

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Assessing the Transport and Fate of Effluent Organic Nitrogen in the Connecticut River and Long Island Sound Using Mass-Mapping Proteomics Technology

Basic Information

Title:	Assessing the Transport and Fate of Effluent Organic Nitrogen in the Connecticut River and Long Island Sound Using Mass-Mapping Proteomics Technology
Project Number:	2009MA186B
Start Date:	4/1/2009
End Date:	5/31/2010
Funding Source:	104B
Congressional District:	MA-001
Research Category:	Water Quality
Focus Category:	Water Quality, Acid Deposition, Nutrients
Descriptors:	
Principal Investigators:	Chul Park

Publications

1. – Westgate, P. and Park, C. (In revision) Evaluation of proteins and organic nitrogen in wastewater treatment effluents. Environmental Science and Technology.
2. – MS Thesis: Characterization of Proteins in Effluents from Three Wastewater Treatment Plants that Discharge to the Connecticut River, MS Environmental Engineering, Aug 2009
3. – Westgate, P. and Park, C. (2009) Evaluation of Effluent Proteins: Towards Characterization of Effluent Organic Nitrogen, Water Environment Federation 82nd Annual Technical Exhibition and Conference (WEFTEC 2009), October 2009, Orlando, FL.
4. – Westgate, P. and Park, C. (In revision) Evaluation of proteins and organic nitrogen in wastewater treatment effluents. Environmental Science and Technology.
5. – MS Thesis: Characterization of Proteins in Effluents from Three Wastewater Treatment Plants that Discharge to the Connecticut River, MS Environmental Engineering, Aug 2009
6. – Westgate, P. and Park, C. (2009) Evaluation of Effluent Proteins: Towards Characterization of Effluent Organic Nitrogen, Water Environment Federation 82nd Annual Technical Exhibition and Conference (WEFTEC 2009), October 2009, Orlando, FL.

Problem and Research Objectives

Significant efforts have been made to reduce the nitrogen released from wastewater treatment plants (WWTPs) and this has been mainly achieved by upgrading the facility for enhanced nitrogen removal through nitrification and denitrification. Though these processes are effective for removing inorganic nitrogen (ammonia and nitrate) organic nitrogen remains little changed, presumably due to its recalcitrant nature, which leads to organic-N becoming a substantial fraction of the N in the final effluent. Thus, one major issue with effluent organic-N is whether it degrades and becomes bioavailable in receiving waters.

Our research group proposed a unique research plan that bases on proteomics analysis to characterize effluent proteins and to assess their fate in receiving waters. Better characterization of effluent proteins and better understanding of their fate in receiving water are critical as proteins comprise a major fraction of effluent organic nitrogen. Furthermore, as proteins can be characterized at a molecular level, profiling of effluent proteins and tracking them in a well defined laboratory bioassay (that mimics receiving waters) will further enable us to determine the fate of proteins, thus a significant fraction of organic-N, in receiving waters. The specific objectives of this project are as follows:

- Determine and characterize proteins in wastewater effluents from major dischargers to the Connecticut River, thus to Long Island Sound.
- Perform a laboratory bioassay and apply proteomics analysis before and after the bioassay to evaluate the bioavailability of effluent proteins in receiving waters.

Methodology

This research has been conducted in two phases: 1) collecting effluents samples and characterizing effluent proteins from various wastewater treatment plants and 2) performing a laboratory bioassay to investigate the fate of effluent proteins and organic nitrogen in receiving waters.

Collection of samples. Primary and secondary effluents were collected from three wastewater treatment facilities that discharge to the Connecticut River in Western Massachusetts. The Northampton and Amherst facilities use conventional activated sludge while the Springfield Regional Wastewater Treatment Facility uses the Ludtzac Ettinger process to treat their wastewater. Samples were collected in plastic containers kept on ice until processed later the same day. Total suspended solids (TSS), volatile suspended solids (VSS) and chemical oxygen demand (COD) measurements were taken the day of collection, while samples were frozen for later measurement of protein, total nitrogen, ammonium, nitrate and nitrite. The secondary effluent from each sample was also separated through a 0.45 μm filter.

Quantification of Proteins in Effluent. The Frølund adaptation of the Lowry method (1995) was mainly used to quantify proteins in wastewater effluents. This method can account for the interference of humic compounds in protein measurement. This method, however, often produced falsely negative protein concentrations for unconcentrated effluent samples. Thus, in this research the quantity of proteins in unconcentrated effluent was measured using the original Lowry method, while the protein concentration in ammonium sulfate concentrated samples were measured using the Frølund adaptation of the Lowry method. Light absorbance was measured with the Thermospectronic Genesys 10 UV Spectrophotometer (Thermo Spectronic, Madison, WI, USA) and concentrations calculated from a standard curve created from 0, 10, 25, and 50 mg/L BSA standards.

Ammonium Sulfate Precipitation. In order to visualize proteins using sodium dodecyl sulfate polyacrylamide gel electrophoresis (SDS-PAGE), secondary effluent and secondary filtered effluent were concentrated with 50% ammonium sulfate. The appropriate mass of ammonium sulfate was combined with 150 ml of primary effluent, 1.2 L of secondary effluent, and 2.2 L of 0.45 μm filtered secondary effluent, in 500 ml centrifuge bottles and one glass pyrex bottle (for 1 L of the 0.45 μm filtered secondary effluent). Precipitation procedures were conducted on ice for more than 12 hours, followed by centrifuging at 11,730 g for 45 minutes. The precipitate was re-suspended in a known volume in phosphate buffered saline (PBS: 10mM NaCl, 1.2mM KH_2PO_4, 6.0mM Na_2HPO_4), then dialyzed extensively in the same buffer with multiple changes at 4°C using a 6-8 kDa cellulose membrane.

Sodium Dodecyl Sulfate – Polyacrylamide Gel Electrophoresis. The SDS-PAGE was performed according to the method of Laemmli (1970). Ammonium sulfate concentrated samples were prepared for size separation on polyacrylamide gels by incubating at approximately 95°C for at least 10 minutes with a 3.3x sample buffer consisting of XT Mops sample buffer and a reducing agent (Bio-Rad, Hercules, CA, USA). Some samples were heat concentrated for up to one hour. Following heat concentration, samples were centrifuged at

12,000 rpm for 3 minutes and the supernatant was used for SDS-PAGE. Prepared samples were loaded onto pre-cast Criterion XT 4-12% gradient gels (Bio-Rad, Hercules, CA, USA) and separated on the gels by a current of 80V for 20 minutes, followed by 100V for two hours. After electrophoresis, gels were stained with silver nitrate or coomassie brilliant blue using Bio-Rad's Silver Stain Kit or Bio-Safe stain (Bio-Rad, Hercules, CA, USA). Gel images were digitally recorded using a CanoScan 8800F desktop scanner (Canon, Tokyo, Japan).

Zymogram analysis. Samples were subjected to zymogram analysis to determine if they contained active proteolytic enzymes. Enzyme activity was determined by separating proteins using electrophoresis in a casein infused gel (Bio-Rad, Hercules, CA, USA). Before electrophoresis the samples were combined with zymogram buffer (Bio-Rad, Hercules, CA, USA) and centrifuged at 12,000 rpm for 3 minutes; the supernatant was used for the zymogram analysis. Gel images were digitally recorded using a CanoScan desktop scanner (Canon, Tokyo, Japan).

Chemical Analysis. Total protein in each of the effluents, both raw and concentrated with ammonium sulfate, was measured using the Lowry method (1951) and determined with a calibration curve generated with bovine serum albumin (Fisherbrand Scientific, Pittsburg, PA, USA). On the day of sample collection, COD, TSS and VSS were measured for primary and secondary effluents according to Standard Methods (2005). COD was measured for secondary effluent filtered through a 0.45 μm filter, as well. Light absorbances for COD tests were determined using the Thermospectronic Genesys 10 UV Spectrophotometer (Thermo Spectronic, Madison, WI, USA) and concentrations calculated from a standard curve using 0, 10, 75, and 150 mg/L KHP standards.

Nitrogen species. Total nitrogen concentrations in primary and secondary effluent, and 0.45 μm filtered secondary effluent were determined using the persulfate method (Hach, Loveland, CO, USA) and confirmed using a Shimadzu TN analyzer (Shimadzu TOC-VCPH with TNM-1, Shimadzu North America, SSI Inc., Columbia, MD, USA). Ammonium, nitrate and nitrite ions in the solution phase (<0.45 μm) of primary and secondary effluents were measured using a Metrohm ion chromatograph (Metrohm, Herisau, Sz). Organic nitrogen was estimated by subtracting the sum of the nitrogen ions from the total nitrogen.

Laboratory bioassay. Several incubation conditions have been tested in an effort to establish the final protocol of laboratory bioassay for this research. Some earlier incubation conditions included: 1) no mixing, 2) intermittent mixing, and 3) continuous mixing of the bioassay bottles. Each set of bioassay included a killed control set to make sure that changes in proteins and organic nitrogen during the bioassay were caused by biological activity. The earlier experiments also tested different dilution sets between effluents and the Connecticut River water at 1:9 and 5:5. For this laboratory bioassay, effluents from Springfield Regional Wastewater Treatment Facility were mainly used. The final bioassay protocol includes following conditions:

1) Filter river water using 100 μm filter.
2) Use 5:5 ratio for river water and secondary effluent for incubation.
3) Perform a separate bioassay on dissolved and whole fraction of secondary effluents.
4) Provide continuous mixing during the incubation.
5) Place the incubation bottles under natural sunlight conditions.

During this laboratory bioassay we also performed Tangential Flow Filtration (TFF) to effectively concentrate the sample before conducting all protein related analysis. Following the concentrating stage, proteins were separated by sodium dodecyl sulfate polyacrylamide gel electrophoresis (SDS-PAGE). Some effluent concentrate samples were sent to another laboratory for liquid chromatography tandem mass spectrometry (LC-MS/MS) analysis to identify the proteins. In addition, various effluent parameters such as TSS, total organic carbon (TOC), cations, anions, and inorganic nitrogen species, as described earlier, were also measured.

Principal Findings and Significance

The current project has revealed several important and new findings regarding effluent organic nitrogen and effluent proteins. The most important finding of this research is that facilities with more advanced N removal processes contain a greater amount of organic nitrogen with a higher diversity of proteins and active enzymes in their final effluents. This indicates that effluent organic-N in advanced wastewater treatment plants differs significantly than that found in conventional wastewater treatment systems. The full effects of released enzymes and proteins in the receiving ecosystem are unknown, but are thought to increase the bioavailability of natural organic matter and to modulate nutrient cycling in the receiving water. As advanced removal of N becomes mandatory in wastewater treatment, it is imperative that this process and potential unintended consequences be fully understood.

The consistent differences between effluent protein profiles from each of the treatment plants investigated further suggest that effluent proteins and enzymes could serve as "fingerprints" of distinct wastewater treatment works and provide a means to track their fate in receiving waters. This fingerprinting concept was employed in the bioassay during the later part of this research and partially used to track the fate of preselected proteins during the incubation. Other major findings and significance of this research can be further summarized as below.

- The research revealed that proteins are significantly correlated with organic nitrogen in effluent from each of the wastewater treatment facilities, demonstrating the significance of protein molecules in effluent organic nitrogen. We believe that there has been no precedent study that has found this relationship or addressed the issue of proteins being an indication, or representative, of effluent organic nitrogen.
- We believe that this is also the first study showing changes in protein profiles, at a molecular level, across processes in a wastewater treatment plant. The results from this approach allowed direct evidence that some influent wastewater proteins persist through the wastewater treatment process and some of these proteins are actually active proteolytic enzymes.
- The research also showed that some bacterial proteins and enzymes that are generated during a biological treatment do indeed end up in the secondary effluent, as so-called soluble microbial products (SMP).
- The finding of active enzymes and proteins in filtered effluent samples is also important to note since the addition of a filtration process to a facility, such as microfiltration, is not likely to improve the capture of these potentially biological compounds.
- Because of these protein results, we gained new knowledge that different treatment works release different sets of proteins (thus, organic nitrogen) and proteolytic enzymes: this information could not be achieved by simple quantitative data or conventional size fractionation techniques. These results are important in that we do not know how these proteins and enzymes behave in the receiving water and what ecological and environmental impacts they may have. The study has provided us a chance to better characterize and identify effluent proteins and enzymes, which can be tracked thoroughly in well defined laboratory bioassays or even directly in the receiving water.

- The bioassay that was designed to mimic the reaction of wastewater effluents in natural receiving waters requires natural sunlight and continuous and uniform mixing during the incubation.
- Concentrations of both inorganic nitrogen (ammonia and nitrate) and organic nitrogen changed greatly during the bioassay, indicating a degradation of organic nitrogen and release of newly generated organic nitrogen.
- After the incubation, new soluble protein bands were detected along with a substantial increase in algal biomass. This observation suggests that receiving waters utilized available effluent nitrogen, including organic nitrogen, and release of proteins from grown algal biomass contributed to a new set of dissolved organic nitrogen remaining in the bioassay.

Characterization of Flow and Water Quality of Stormwater Runoff from a Green Roof

Basic Information

Title:	Characterization of Flow and Water Quality of Stormwater Runoff from a Green Roof
Project Number:	2009MA199B
Start Date:	4/1/2008
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	3
Research Category:	Water Quality
Focus Category:	Water Quality, Hydrology, Water Quantity
Descriptors:	
Principal Investigators:	Paul Mathisen, Suzanne LePage

Publications

1. LePage, Suzanne “An investigation of the hydrologic and geochemical processes contributing to green roof performance”, MS Thesis, Worcester Polytechnic Institute, completed in May 2010
2. LePage, Suzanne and Paul Mathisen, “An investigation of the hydrologic and geochemical processes contributing to green roof performance.” presentation at the American Society of Civil Engineers (ASCE) World Environmental and Water Resources Congress 2010. Providence, Rhode Island. May 20, 2010.
3. LePage, Suzanne, “An Investigation into the Water Quality Impacts of a Green Roof”, poster presentation at WPI Graduate Achievement Day, Worcester Polytechnic Institute, Worcester, MA on March 31, 2010.
4. LePage, Suzanne, “An Investigation into the Water Quality Impacts of a Green Roof”, poster presentation at the 7th Annual Water Resources Research Conference, Amherst, MA on April 8, 2010

Problem and Research Objectives

Low Impact Development (LID) techniques for site design are increasingly being utilized to mitigate the negative impacts associated with stormwater runoff, and green roofs are one such application. The ability of green roofs to reduce the total and peak volumes of stormwater runoff has been fairly well documented, but performance varies in different climate zones, and there is limited information available regarding green roof effectiveness in New England, a region whose weather patterns are notoriously variable from season to season and often even day-to-day. Additionally, there are questions regarding the impact that green roofs have on water quality. While there seems to be a general consensus that green roofs will leach phosphorus, and sometimes other contaminants, into stormwater runoff within the first few years after installation, it is assumed that this phenomenon will not continue after the green roof vegetation has been established. However, it is still unclear whether or not this assumption is valid, and very few research projects have attempted to provide the necessary insight into the hydrologic and chemical processes that are contributing to this question.

Accordingly, the goals of this research were to provide insight into the hydrologic and geochemical processes that contribute to green roof performance. The specific objectives included the following:

- Determine the effectiveness of a green roof in attenuating stormwater flow
- Document a green roof's impact on water quality, specifically regarding phosphorus
- Identify the key components of the processes that are likely leading to the highest variability in observed water quality parameters – hence, the highest potential that a change in design could lead to significant improvements

In addition to providing insight into green roof performance, these objectives are intended to provide a foundation for future research efforts to explore the behavior of phosphorus in soil solutions and its implications for stormwater treatment.

Methodology

The methodology for achieving the project objectives combined field monitoring and laboratory testing and analyses to characterize the quality of runoff associated with the Nitsch/Magliozzi Green Roof, an extensive green roof located on top of a new residence hall at WPI. This roof, which was donated to enhance the sustainability of the building and foster continued research and education, provided the context for this project. The research tasks included field monitoring of the roof drainage, laboratory testing of green roof panels under simulated rainfall conditions, bench-scale testing of phosphorus desorption from the growing medium, and laboratory analyses of water quality, soil characteristics, and plant phosphorus content. The methodology provided a basis for gaining a better understanding of the relationship between rainfall and runoff volumes, phosphorus sorption/desorption in the growing medium, and plant uptake processes.

The field monitoring program focused on the seasonal variations of water quality throughout a complete growing season. Two flow meters and sampling ports have been installed within the storm drain system of the residence hall: one to measure drainage from the green roof; and the other to measure drainage from the "non-green" portion of the roof. Using these sampling ports, a total of 25 grab samples from each roof were collected and analyzed between June 2009 and April 2010.

The laboratory testing and analysis program was developed to characterize both the stormwater retention performance and water quality characteristics of the green roof. For this program, two (2) of the green roof panels were brought into a greenhouse maintained at WPI by WPI's Biology Department. A stand was constructed which allowed for the application of simulated rainfall and collection and measurement of runoff for each panel. For water quality monitoring, runoff from each panel was detoured through a flow-through device attached to a water quality monitoring sonde (Hach MS5 Hydrolab unit), and grab samples were collected at key points during the simulated storms. The Hach MS5 Hydrolab units, one of which was acquired using support from this grant, were important components of this system. Soil and plant samples were also collected, and additional bench-scale tests were completed to characterize the nature of the phosphorus desorption from the media. All samples of water, plant, and soil were analyzed in the water quality laboratory in WPI's Department of Civil and Environmental Engineering.

Principal Findings and Significance

In regards to storm-water flow attenuation, results from the greenhouse experiments showed that green roof performance was more effective for smaller storms, and was influenced by the soil properties (including field capacity and moisture content). Overall, these results are consistent with the published literature. For example, the reduced retention capacity observed during higher flow conditions is a common trend that has been reported for extensive green roof performance. At high rainfall intensities, the field capacity of the green roof panels is quickly exceeded, and the thin layer of the extensive green roof design does not provide much storage capacity. However, while the growing medium did not provide much storage during the heavier simulated rain event, the green roof vegetation's ability to rapidly uptake water when it becomes available did provide a stormwater retention benefit. The improved performance during the lower flow conditions was found to be more heavily influenced by the soil than by the plants. The highest retention rates in the simulated rain events were observed when the antecedent moisture content was low (9-11%). In contrast, for a light rain event, the moisture content of the soil at the beginning was the highest of all tests (26%), and the green roof panels retained only 38% of the influent volume, despite the fact this simulated storm used the smallest volume of water of all simulated events. Clearly the growing medium's field capacity is a critical design factor that is indicative of green roof performance.

In regards to water quality, phosphorus concentrations observed in runoff during green house tests, were similar in magnitude to the concentrations in samples collected from the green and white roofs, which were relatively high. These high concentrations were found to be primarily influenced by phosphorus in the growing medium, which quickly desorbs in response to flushing due to storm events. For all greenhouse tests, the phosphorus concentrations (and other constituents as well) showed up in the "first flush" runoff samples and continued to increase throughout the duration of the storm and after the simulated rainfall had stopped. This trend was consistently observed in all storms, regardless of their size or intensity. These results indicate that the desorption of phosphorus from the growing medium happens quickly, and the soil is not rapidly depleted of its phosphorus content. Also, the green roof panel whose soil was higher in phosphorus concentration (Stand B) also produced runoff with higher phosphorus concentrations than the other panel tested in the greenhouse (Stand A). Meanwhile, the growth of green roof plant material and its associated nutrient uptake processes did not appear to reduce the amount of phosphorus that ended up in the runoff. These results confirm that the growing medium is the source of phosphorus in runoff. However, while a bench-scale laboratory experiment indicated that phosphorus levels in runoff may decrease over time, the rate of desorption is not constant and cannot be easily predicted. Additional investigations will be needed in order to predict the long-term impact of a green roof on phosphorus loading.

With consideration to the design of green roofs, a number of key processes/factors were defined. First, this research showed that soil storage and soil moisture content are particularly important considerations with respect to green roof performance. Soil storage is heavily influenced by antecedent moisture content, and soil moisture content is a function of both weather, which cannot be controlled, and plant variety, which generally can be controlled. These results should help future designers determine whether the weather patterns in a particular location where a green roof is being considered will be hindrance to the effectiveness of a green roof. Areas experiencing significant amounts of rainfall that may keep the soil at field capacity would not be a good choice. However, selecting plant varieties that quickly uptake water, such as sedum and delosperma, will provide the ability to regenerate the holding capacity of the growing medium and will improve the performance of green roofs. Also, efforts should be taken to engineer new soil media that will maximize the field capacity of green roof designs. Second, the research showed that the leaching of phosphorus from the growing medium must be taken into consideration when designing a green roof. Previous studies have made assumptions that the leaching of phosphorus will decrease over time and many have predicted that the phenomenon will only occur for a few years after installation. However, the results of this study indicate that this assumption may not be valid. The long-term phosphorus loading resulting from a green roof may continue longer than previously assumed. Until additional investigations are conducted to develop a prediction model, the impacts of a green roof must be given careful consideration if being installed where phosphorus levels in stormwater are a concern. Further, it is recommended that phosphorus use be minimized in the growing medium. The typical green roof plant varieties, such as those studied here, do not appear to uptake very much of this nutrient, even in their first few establishment years.

In general, these results provide a basis for developing improved predictions of storm-water retention performance, gaining deeper insight into the transformations of phosphorus in the green roof panels, and developing a process by which continued, in-depth study could be performed under controlled laboratory and field conditions.
Publications and Conference Presentations:
The results summarized in this summary report are also described in more detail in a Master of Science thesis prepared by Suzanne LePage in partial fulfillment of the requirements for her Master of Science degree at Worcester Polytechnic Institute. The results were also disseminated via a presentation at the EWRI Congress of the American Society of Civil Engineers (ASCE), and via a poster presentation at the Annual Water Resources Conference in Amherst, MA. The details of these items are included in the following listing:

Publications and Conference Presentations

Dissertations / MS Theses

LePage, Suzanne, 2010. An investigation of the hydrologic and geochemical processes contributing to green roof performance. MS Thesis, Worcester Polytechnic Institute, completed in May 2010

Other Publications and presentations

LePage, Suzanne and Paul Mathisen, 2010. *An investigation of the hydrologic and geochemical processes contributing to green roof performance.* Presentation at the American Society of Civil Engineers (ASCE) World Environmental and Water Resources Congress 2010. Providence, Rhode Island. May 20, 2010.

LePage, Suzanne, 2010. *An Investigation into the Water Quality Impacts of a Green Roof*. Poster presentation at WPI Graduate Achievement Day, Worcester Polytechnic Institute, Worcester, MA on March 31, 2010.

LePage, Suzanne,2010. *An Investigation into the Water Quality Impacts of a Green Roof*. Poster presentation at the 7th Annual Water Resources Research Conference, Amherst, MA on April 8, 2010 (see note below on award).

Student Support

This project provided equipment that assisted the research program of 1 graduate student, Suzanne LePage, at Worcester Polytechnic Institute. The matching funds designated in this grant included the student time and effort as part of an independent study project (ISP) completed in the fall of 2009.

Notable Achievements and Awards

2nd Place award for Poster entitled "*An Investigation into the Water Quality Impacts of a Green Roof*", which was presented at the Seventh Annual Water Resources Research Conference Poster Contest on April 8, 2010

Characterizing and Quantifying Recharge at the Bedrock Interface

Basic Information

Title:	Characterizing and Quantifying Recharge at the Bedrock Interface
Project Number:	2009MA213G
Start Date:	9/1/2009
End Date:	8/31/2012
Funding Source:	104G
Congressional District:	1st
Research Category:	Ground-water Flow and Transport
Focus Category:	Groundwater, Water Supply, Water Quantity
Descriptors:	None
Principal Investigators:	David Boutt, Stephen B. Mabee

Publications

1. Characterizing Groundwater Recharge Across the Surficial/Bedrock Interface. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Resource Center Annual Conference. April 8, 2010. (Poster session).
2. Characterizing Groundwater Recharge Across the Surficial/Bedrock Interface. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Resource Center Annual Conference. April 8, 2010. (Poster session).
3. Weider, K. and D.F. Boutt, Heterogeneous water table response to climate revealed by 60 years of ground water data, Geophys. Res. Lett., doi:10.1029/2010GL045561, 2010.
4. Boutt, D.F., Poroelastic response of an unconsolidated aquifer to daily releases of water from an upstream dam, Ground Water, doi:10.1111/j.1745-6584.2009.00,663.x, 2010.
5. Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US. Bevan, L.B., D.F. Boutt, S.B. Mabee. American Geophysical Union Conference. December, 2010. (Poster session).

Title Characterizing and Quantifying Recharge at the Bedrock Interface

Project Number 2009MA213G

Start Date 9/01/2009

End Date 8/31/2012

Funding Source 104G

Research Category Groundwater Flow and Transport

Focus Categories GW WS WQ

Descriptors Groundwater, Bedrock, Fractured-Rock Aquifers, Recharge

Primary PI Dr. David Boutt

Other PIs Dr. Stephen B. Mabee

Problem and Research Objectives

Evaluating the sustainability of fractured bedrock as a groundwater resource and understanding the environmental impacts of water withdrawals from the bedrock on nearby streams, wetlands, ecosystems and unconsolidated aquifer systems requires an estimate of the recharge and an understanding of the advective flux across the bedrock –overburden boundary. Few published studies address this issue with direct measurement (Rodhe and Bockgard, 2006, White and Burbey, 2007) while others use tracers (e.g. Rugh and Burbey, 2008) and numerical models to study the distribution of groundwater flow in the soil and bedrock (eg., Harte and Winter, 1995; Tiedeman et al., 1998). However, quantifying the flux of water between the overburden and bedrock remains one of the major sources of uncertainty in numerical models (Lyford et al., 2003). The only long term monitoring well that is screened in bedrock in Massachusetts shows an alarming downward trend in hydraulic head over a period of almost 20 years while the hydraulic head from a nested piezometer in the surficial material above the bedrock piezometer does not show a similar trend. Understanding the dynamics of how systems like these interact is fundamental to improving our ability to manage and regulate these important resources.

The objectives of this project are to evaluate water flux across the overburden-bedrock interface under ambient and stressed conditions and to estimate its hydraulic conductivity in three typical hydrogeologic settings in the glaciated terrain of eastern Massachusetts. The hydrogeological conditions that will be examined include thick till overburden, thin till-shallow bedrock and coarse-grained stratified deposits. The work will be conducted in the Assabet River watershed because this watershed has previously been modeled by the USGS (DeSimone, 2004). The project complements past and ongoing work by the USGS in the New England region that evaluates water availability and the impacts of pumping on shallow aquifers and riparian systems (eg., DeSimone et al., 2002, DeSimone, 2004; Carlson et al., 2008). The proposed project is also designed to complement a project underway by the USGS Water

Science Center in Northborough, MA that is assessing the factors affecting bedrock well yields in the Nashoba terrane. Many of these projects benefited from several years of cooperation between the Office of the Massachusetts State Geologist, University of Massachusetts, USGS and the MA Department of Environmental Protection.

The expected outcome of this work is a clearer understanding of the groundwater flux across the overburden-bedrock boundary and how the coupled systems respond to seasonal changes, individual recharge events and potential stresses due to pumping. Acquisition of these data will provide a basis for calibrating numerical models that investigate the effects of groundwater withdrawals (both surficial and bedrock) on stream baseflows and ecosystems.

Methodology

Site Selection

A groundwater monitoring site in Berlin, MA was chosen in the context of the existing USGS groundwater flow models. The site known as the Gates Pond Reservoir lies in areas where the Nashoba formation outcrops and subcrops and is covered by till or has exposed bedrock. The Town of Hudson Department of Public Works, the water supply agency that regulates the Gates Pond granted access to the site in May 2010. The Gates Pond Reservoir has four pre-existing monitoring wells that are 6 inches in diameter and terminate in bedrock at depths that range between 150 and 245 meters below surface level (bsl). The fractured bedrock intersected by these wells have been extensively characterized for their hydrologic properties as reported in Boutt et al., 2010.

The site was surveyed in June 2010 and maps were developed using resources provided by The Massachusetts Office of Geographic Information, the USGS. In addition to topographical surveys, 2D seismic refraction and electrical resistivity surveys (Figure 1) were conducted in the areas adjacent to the bedrock wells as well as till covered slopes to identify the location of the water table and bedrock interface.

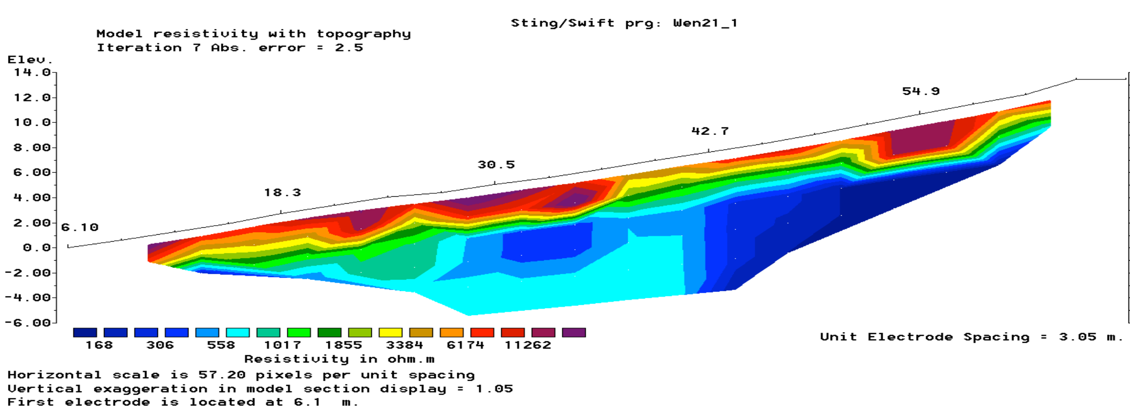

Figure 1. Inverse model of data collected from a East to West Wenner resistivity array on the eastern edge of the thick till (drumlin) deposit.

Site Instrumentation

Two bedrock wells (BMW-1 and BMW-2) were instrumented in May 2010 with Solinst LevelLoggers to measure time-series of temperature and pressure within each of the wells. To correct for barometric pressure influences on bedrock well water levels, a Solinst BaroLogger was installed above ground near BMW-1 to measure atmospheric pressure and air temperature. All four bedrock monitoring wells are instrumented to date with BMW3 and BMW4 monitoring beginning in Winter 2011. Two 1.75 inch diameter surficial wells were installed in both the thin till and thick till packages. The wells were installed to a depth of 1 and 2 meters respectively. Each of these wells was instrumented with a Solinst LevelLogger pressure and temperature sensor. Additionally, pond level was also measured with a LevelLogger. All LevelLoggers installed are logging pressure and temperature data at 5 minute intervals. In addtion to the water level and temperature probes, Four Decagon Devices 5TM soil moisture probes were installed within both the thick and thin till. These probes are installed at the top of a slope and at the base of the slope at depths of .5 meters and 1.2 meters. The 5TM probe from Decagon Devices collects volumentric water content (VWC) by measuring and converting the soils dielectric permittivity into soil moisture content using the Topp equation:

$$VWC = 4.3x10-6\ ea3 - 5.5x10-4\ ea2 + 2.92x10-2\ ea - 5.3x10-2$$

Data collected from the 5TM probes are logged and transmitted to an off site computer via a Decagon Devices EM50G wireless cellular datalogger. Precipitation measurements are taken from a USGS precipitation gage located in the nearby town of Sterling, Massachusetts. The USGS gage collects precipitation data at 15-minute intervals.

Two distributed temperature sensing (DTS) fiber optic probes were built for both BMW1 and BMW2. These probes are deployed in the field and have only been implemented for limited time periods.

<u>Data Collection and Site Characterization</u>

In the spring of 2011 we performed a pump test at the proposed field site (Gates Pond) in the Nashoba terrane. Gates Pond consists of 4 bedrock wells with depths on the order of 250 meters. The wells are located in two NE-SW trending lineaments that are separated by a bedrock ridge, with wells 1 and 2 in the western-most lineament, and wells 3 and 4 the eastern-most lineament . Pumping was performed in well 1, and drawdown was observed in wells 2, 3 and 4. Well 2 displayed significant drawdown during pumping; however, wells 3 and 4 do not display significant drawdown. These results suggest that wells 1 and 2 are hydraulically connected, whereas wells 3 and 4 are not hydraulically connected to the pumping well. We interpret these observations to indicate a strong NE-SW hydraulic conductivity anisotropy. The orientation of this anisotropy is coincident with the orientation of Foliation Parallel Faults (FPF), suggesting that FPFs are playing a major role in permeability anisotropy between wells 1 and 2. Wells 3 and 4 are located in a lineament that is east of the pumping well, suggesting that E-W oriented fractures are less permeable than the NE-SW trending FPFs. What is more, the orientation of the regional maximum horizontal stress in New England is approximately NE-SW, and thus the maximum permeability orientation interpreted from the pump test is parallel to the maximum horizontal stress, and fractures oriented obliquely to the maximum horizontal stress are less permeable.

Data collection began in May 2010 in BMW1 and BMW2. The surficial well, pond, soil moisture measurements began in earnest September 2010. Currently at the site, 22 separate pieces of data (air temperature, barometric pressure, 4 soil moisture contents, 8 hydraulic heads, and 8 water temperatures) will be recorded at 5-minute intervals resulting in 6,336 pieces of data per day. The additional of the 15-minute precipitation measurements at the USGS Sterling, MA gage (96 per day), brings the total number of pieces of data collected per day to 6,432 pieces of data per day to be considered.

Water samples were taken for stable isotope analysis from all surficial and bedrock wells, as well as from Gates Pond Reservoir. Isotope samples may give some insight into subsurface fractionation processes that are expressed within recharge.

Soil cores were taken in the locations of the surficial wells and were analyzed for grain size, porosity, specific yield, total organic content and vertical saturated hydraulic conductivity via falling head permeameter testing.

In order to determine in-situ saturated hydraulic conductivity, reverse slug tests were performed in the surficial wells by removing a volume of water from the well and observing the rate of replenishment in the well via Solinst LevelLogger. Pumping tests were executed using a GeoTech submersible pump in BMW1 and BMW3 respectively and drawdown was monitored in all four bedrock monitoring wells concurrently.

Data Reduction, Interpretation, Modeling

Data collected from the instruments on site is time synced to ensure proper analysis. A MATLAB code was developed to process all of the collected time series data from the deployed instruments. The data was normalized to an arbitrary datum that was established by topographical survey. Portions of the surficial well level data was detrended and cross correlated with the bedrock well level data in order to determine the lagged time response in the bedrock wells from a precipitation input (Lee et al., 2006). A cross correlation is essentially the plotted coefficients of determination (R^r) of a cross plot of the bedrock and surficial wells at a series of lagged sampling intervals. The higher the R^2 value, the more likely there is a similarity in the well responses at that particular lag time. Currently, there is an effort being made to filter the detrended data in order to remove the periodic barometric loading effect. The barometric loading may be providing false correlations.

A

B

C

D

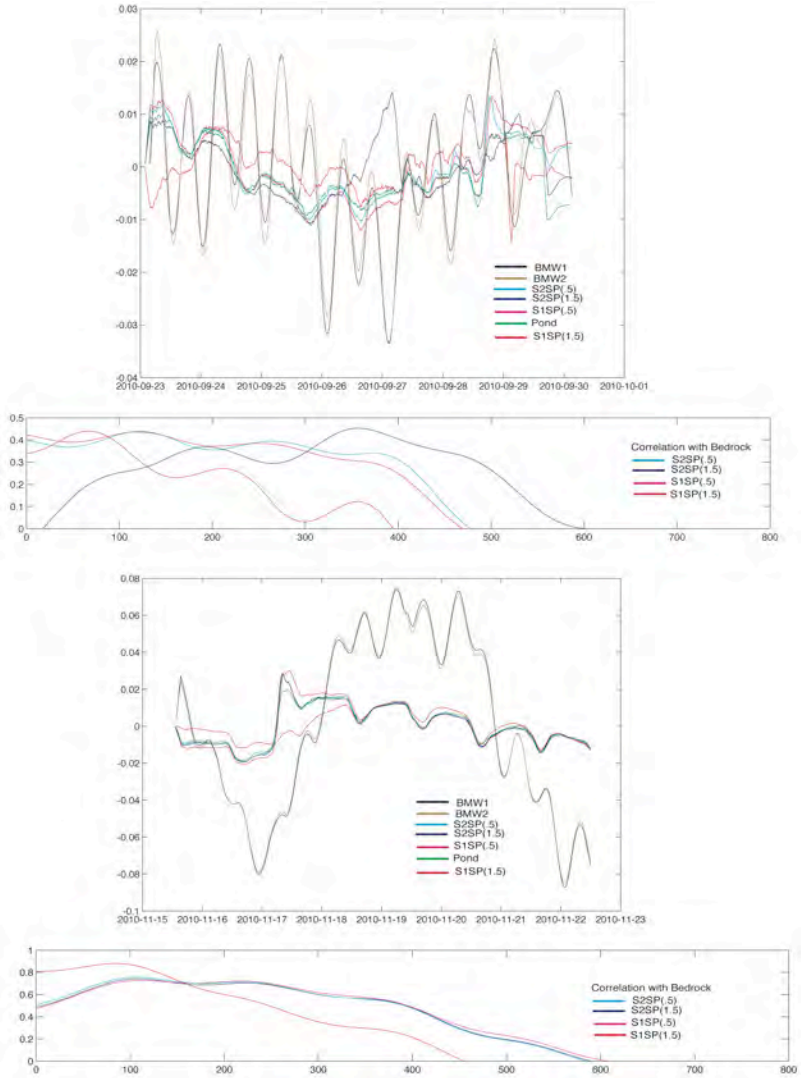

Figure 2. a) Detrended water level data from one week in September 2010. b) Cross correlated data between surficial and bedrock wells for the detrended data in 2a. c) Detrended water level data from one week in November 2010. d). Cross-correlated data between surficial wells for the detrended data in 2c. Note that the architecture of the correlation structure changes dramatically. There implies that the signals are responding to a common input. A low pass filter will need to be worked into the MATLAB code in order to refine the signal and eliminate the barometric loading bias that is apparent in all of the signals.

The hydraulic data together with the temperature data will be used to build 1-dimensional coupled saturated-unsaturated water flow and temperature models using the general finite-element method solver COMSOL multiphysics (Fleming, 2009). Hydraulic and heat boundary conditions will be provided by the field measurements. The integration of both head and temperature data into a model such as this reduces the degrees of freedom and will allow an estimation of the vertical flux across various boundaries (water table, till-bedrock interface) present in the model (Anderson, 2005) that is constrained by observations of head and temperature. A split data approach will be used to calibrate the model reserving the other half of the data to predict water flux under different hydraulic conditions throughout the data set. The hydraulic properties of the bedrock-till interface will be the main calibration parameter.

For each a site a detailed quantitative model will be developed to understand the movement of water from the surface through to the bedrock. Recharge rates to the till (where present) will be estimated and the amount of leakage (recharge) to the bedrock will also be determined using the data collected. Results from this analysis will yield a detailed set of water fluxes for the hydrologic periods during data collection. The hydraulic properties of the bedrock-till interface will be an important calibration and model result.

Preliminary Findings and Significance

Work on this project began on September 1, 2009 with site selection and characterization and continues today with time series analysis and aquifer hydraulic characterization. Figure 3 includes a site locus as site map showing the location of monitoring equipment as well as geophysical survey lines. Time series data collected from wells BMW1 and BMW2 have captured bedrock recharge timing and magnitude at the Gates Pond Site (Fig. 4).

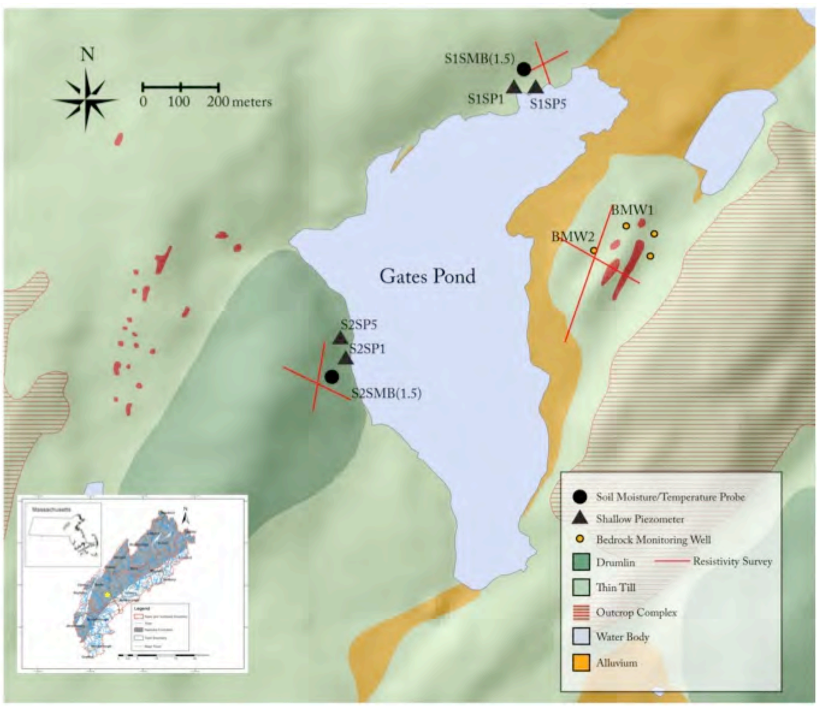

Figure 3. Site map of the Gates Pond Reservoir. In the lower left corner is a site locus showing the location of Gates Pond in relation ship to the Nashoba Formation in Massachusetts.

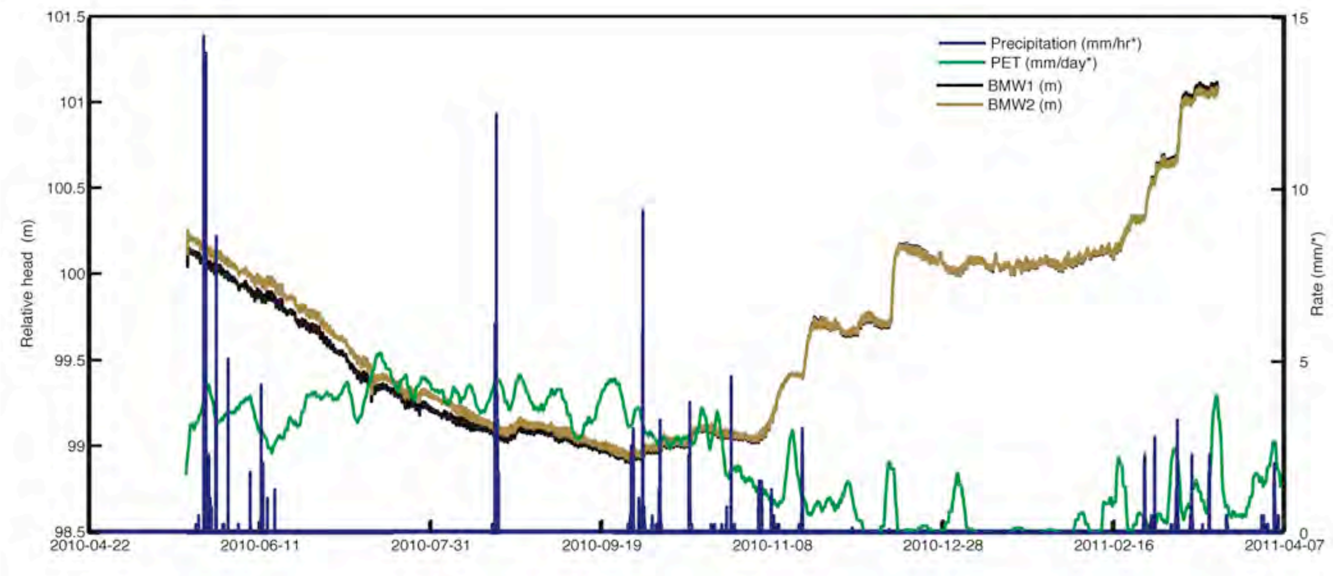

Figure 4. Time series of normalized head from BMW1 and BMW2. Potential evaporation rate as calculated using the Thorthwaite approximation and precipitation rate are also plotted.

Summer precipitation events do not to appear to have an appreciable effect on the trend of the bedrock head. Instead, bedrock recharge occurs during times with reduced potential evapotranspiration. The bedrock recharge occurs more rapidly as the Fall season progresses. During the earliest part of Fall, bedrock wells and the surficial wells respond with approximately the same magnitude until the middle of Fall when the bedrock response to precipitation increases in its magnitude (Fig 5). This sudden change in how the bedrock system reacts suggests that there is a threshold in the system that must be met in order for there to be an appreciable amount of recharge to the bedrock. The investigators believe that the threshold is represented by the hydraulic conductivity in the till. While at lower soil moisture values, the tills have little ability to transmit water through it matrix. As soil moisture content increases (Figure 6), the till's hydraulic conductivity increases until an effective hydraulic conductivity is achieved. It is at this effective hydraulic conductivity that water may transmit water to deeper depths. Earlier

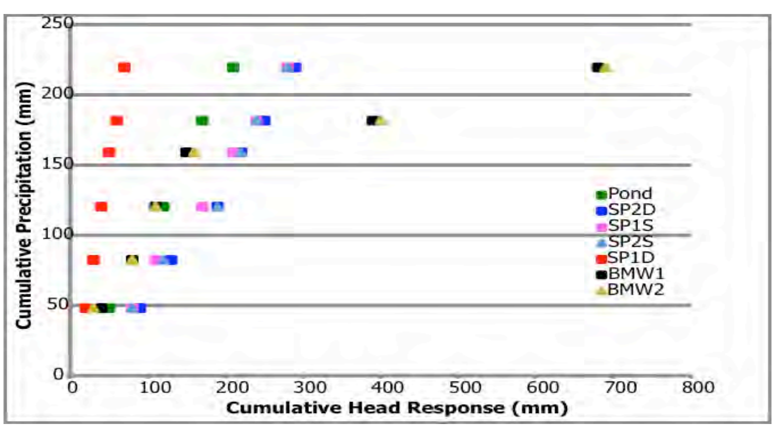

Figure 5. Cumulative plot of the early fall when recharge began to occur in the bedrock monitoring well. Note that the bedrock wells (BMW1 and BMW2) respond to precipitation events similarly to the surficial wells until a point after which the bedrock monitoring wells respond very differently.

There is also a steepening of the slope associated with the bedrock response to recharge that may be associated with an increase in the soil moisture content. Preliminary observations indicate that bedrock recharge is highly dependent upon soil moisture dynamics and overburden thickness.

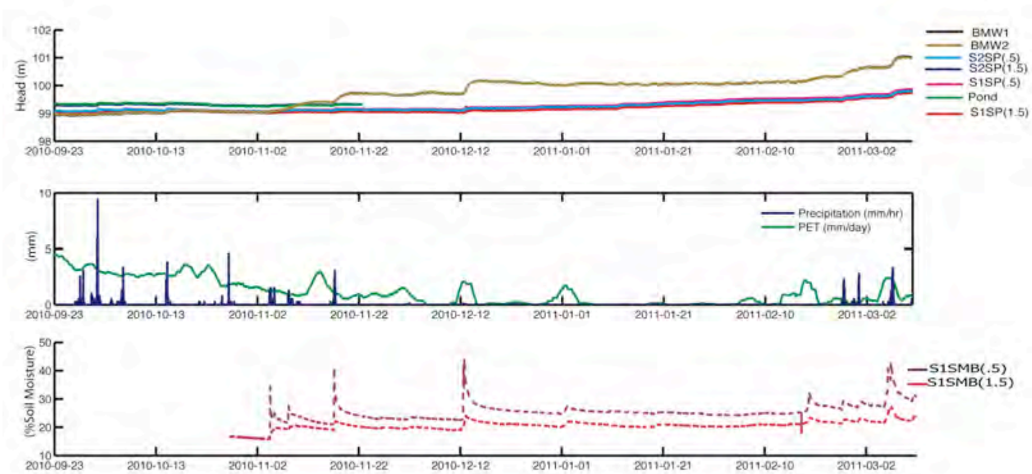

Figure 6. This figure illustrates the responses in the surficial water table compared to bedrock water table

Soil hydraulic conductivities were taken at each site at two different depths. The slug test results, when solved with the Hvorslev method, there is a distinct non-linearity that is appears in late time during the test. This represents radial flow and can be attributed to the limited vertical connectivity to the aquifer due to the well construction techniques that were used. This non-linearity is shown in Figure 7.

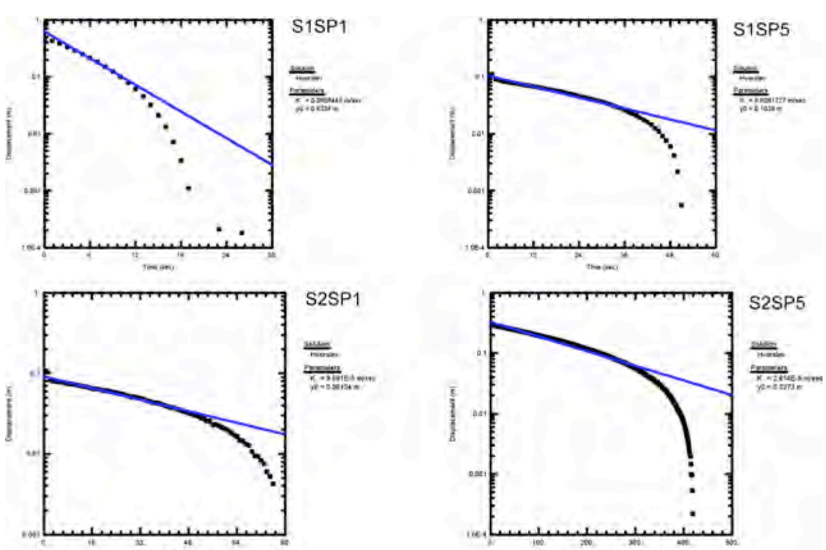

Figure 7. Slug test results as analyzed using the Hvorslev method. The non-linearity confirms the dominance of radial flow and therefore, results with a more appropriate solution (like Cooper et al.) can

be used to get the horizontal hydraulic conductivity. This value can be compared to the falling head permeameter results to determine horizontal/vertical anisotropies in the system.

Coincidental to the bedrock recharge, the soil moisture begins to increase, suggesting that the hydraulic conductivity of the soil must also increase. Hydraulic conductivity varied between the thin and drumlin till sites and is summarized in Table 1 below.

Site	Slug Test	Falling Head
Thin Till		
S1SWS	8.4e-4 m/s	1.28E-7 m/s
S1SWD	1.7e-4 m/s	1.8E-7 m/s
Drumlin		
S2SWS	9.7e-5 m/s	3.1E-7 m/s
S2SWD	2.6e-5 ,/s	3.5E-7 m/s

Table 1. Hydraulic conductivities of surficial materials located at Gates Pond Reservoir.

Future Work

Plans are currently being developed to install an additional bedrock monitoring well at the Gates Pond site. While installing the well, the bedrock/till interface will be cored and characterized. The new bedrock well will be instrumented with fiber optic DTS probes as well as instrumented to measure hydraulic head at multiple depths. The well will also be geophysically logged for resistivity, with a heat pulse flow meter, imaged and interpreted via optical televiewer and caliper. Time series data will continue to be collected and analyzed from all wells, soil probes and the pond. Stable isotope analysis will also be performed from regular sampling at the site. Stable isotopes will give the Investigators insight as to the origin of the water onsite and will allow the investigators to determine whether responses in bedrock wells are the result of advection across the bedrock/till interface or an expression of the pressure wave associated with hydraulic diffusion and surface loading of meteoric water mass.

Publications and Conference Presentations

Weider, K. and **D.F. Boutt**, *Heterogeneous water table response to climate revealed by 60 years of ground water data*, Geophys. Res. Lett., doi:10.1029/2010GL045561, 2010.

Characterizing Groundwater Recharge Across the Surficial/Bedrock Interface. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Resource Center Annual Conference. April, 2010. (Poster session).

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US. Bevan, L.B., D.F. Boutt, S.B. Mabee. American Geophysical Union Conference. December, 2010. (Poster session).

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US. Bevan, L.B., D.F. Boutt, S.B. Mabee. Massachusetts Water Research Center Annual Conference. April, 2011. (Poster session). First place poster submission.

Developing a Conceptual Model for Bedrock Recharge in the Glaciated Northeastern US. Bevan, L.B., D.F. Boutt, S.B. Mabee. Novel Methods for Subsurface Characterization Conference. May, 2011. (Poster session).

Student Support

Liam B. Bevan is fully supported by this project. He is pursuing an M.S. degree in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

Evan Earnest-Heckler is partially supported by this project. He has been assisting with field work and developing a detailed characterization of the fractured bedrock of the site. He is pursuing a PhD in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

Shakib Ahmed used data collected from this project for his Senior thesis. He recently completed his B.S. in geology in the Department of Geosciences at the University of Massachusetts, Amherst.

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Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing

Basic Information

Title:	Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing
Project Number:	2010MA231B
Start Date:	3/1/2010
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	Massachusetts's 1st congressional district
Research Category:	Climate and Hydrologic Processes
Focus Category:	Water Quality, Hydrogeochemistry, Models
Descriptors:	None
Principal Investigators:	Qian Yu, Weining Zhu

Publications

1. Zhu, W.N., Q. Yu, Y.Q. Tian, R.F. Chen, and B.G. Gardner, 2011, Estimation of chromophoric dissolved organic matter in the Mississippi and Atchafalaya river plume regions using above surface hyperspectral remote sensing, Journal of Geophysical Research-Oceans, 116, C02011
2. Yu, Q., Y. Q. Tian, R.F. Chen, A. Liu, G.B. Gardner and W.N. Zhu, 2010, Functional linear analysis for estimating riverine CDOM in coastal environment using in situ hyperspectral data, Photogrammetric Engineering and Remote Sensing, 76(10), 1147–1158. (Early Career Best Paper Award, AAG Remote Sensing Specialty Group)

Annual Report (from March 1, 2010 to February 28, 2011)

Title: Monitoring and Modeling Chromophoric Dissolved Organic Matter in Neponset River and Boston Harbor Using GIS and Hyperspectral Remote Sensing

Project Number: S11500000000037

Start Date: March 1, 2010

End Date: February 28, 2011

Funding Source: USGS State Water Resources Research Institute Program

Research Category:

Focus Categories: WQL, MOD, HYDGEO

Descriptors:

Primary PI: Qian Yu

Problem and Research Objectives:

(1) Problems

As one of the major components of DOM, dissolved organic carbon (DOC) is a key factor for water quality. The organic molecules that make up the DOC pool in fresh and coastal waters come from natural sources, mainly from the decay of terrestrial and aquatic plants, algae and bacteria. Meanwhile, anthropogenic activities, such as sewer release and agricultural fertilization, also have strong impact on DOC contents and compositions in watershed regions with high population density. The watershed flux of DOC from terrestrial landscape to rivers has wide-ranging consequences for aquatic chemistry and biology. DOC affects the complexation, solubility, and mobility of metals as well as the adsorption of pesticides to soils, and is therefore a critical water quality parameter important for human health. In addition, DOC also plays an indispensable role in the cycle of terrestrial and atmospheric CO2, and hence implies the climate change. Currently, due to physical and biological complexity, multiple scales of freshwater systems, and biogeochemical reactivity at the land-water interface, changes to DOC fluxes in response to terrestrial sources and climate change are not well-known, and so as to be hard to evaluate their potential impact on water quality. To quantify the seasonal or interannual variation of DOC flux, the first thing is to accurately estimate DOC amount in riverine and estuarine regions. This is not only crucial to analyze their sources and transport mechanisms, but also subsequently helpful for monitoring and controlling water quality and for understanding the regional and global carbon cycle.

Our study site is the Neponset River and Boston Harbor regions. The Neponset River locates in eastern Massachusetts, starting at the Neponset Reservoir and meandering generally northeast for approximately 29 miles to its mouth at Dorchester Bay of Boston Harbor. The Neponset River is fed by a drainage basin or watershed of approximately 130 square miles, including numerous aquifers, wetlands, streams, surrounding upland areas, and about 330,000 populations. The Neponset has been heavily polluted before, but at present it is clearer due to much remediation. Several water quality monitoring programs are running for this river, for example, the Massachusetts Water Resources Authority's harbor and river monitoring program monitors the impacts of combined sewer overflows (CSOs) on the Harbor and tributary rivers. The sampling crew measures temperature, bacteria, algae, water clarity, nutrients, and suspended solids in Boston Harbor and Neponset rivers. However, there are some shortcomings of current monitoring programs. First, DOM/DOC is not usually directly measured. The conventional water quality parameters, such as ions concentrations, pH values, and dissolved oxygen, are not appropriate to estimate DOC's concentration. Second, even DOM/DOC are measured sometimes, most of them are only point sampling. Such field survey is insufficient or less reliable for evaluating the distribution and dynamic of organic matters at a large spatial scale. Third, our previous study shows DOC has strong seasonal and even daily variations. Frequent and synoptic DOC measurement is in great need for monitoring DOC flux spatial temporal variation in a large area.

Inversion of DOC concentration from satellite images has great potential to overcome the shortcomings of field survey. The concentrations of in-water components influence water's absorption and backscattering coefficients and hence change the radiance received by satellite sensors. Therefore, based on satellite images, we can inversely predict those components. Although DOC is unable to be fully estimated from satellite images since part of them are not photoactive substances, the photoactive fraction of DOM, chromophoric dissolved organic matter (CDOM), could be used as the tracer of DOC. Many observations provide the evidence of a good correlation between CDOM and DOM/DOC loading across the different subcatchments, despite the absence of this co-variation in a few cases. CDOM absorbs primarily ultraviolet and blue light, and is fluorescent (350nm – 440nm). Together with the other two ocean-color components, chlorophyll and non-algal particles (NAP), CDOM have a significant contribution to the signals received by satellites and therefore is detectable by remote sensing.

(2) Research objectives

Our study will focus on two aspects: 1) rapidly quantifying CDOM via remote sensing-based inversion and 2) watershed-based modeling to understand CDOM sources and degradation.

Remote sensing-based inversion is to observe CDOM concentrations in freshwater and coastal regions from in situ spectral data measured by ASD FieldSpec® 3 and satellite hyperspectral images EO-1 Hyperion. Hyperion sensor bears relatively high resolutions both in spectral (10nm) and spatial (30m), and hence provides a good platform for optical inversion of CDOM. Specifically, we will improve and calibrate our algorithm, QAA-E (Extended Quasi Analytical Algorithm), to inverse the ag440 (the absorption coefficients of CDOM at 440nm, typically denoting CDOM concentrations in ocean-color science) from satellite images. Our previous study on algorithm development and test shows QAA-E is able to retrieve ag440 with acceptable accuracy in Mississippi River and Atchafalaya River plumes. The implement of this objective is the base for further work of modeling.

Watershed-based modeling is to better understand the DOC sources and its relationship and coupling to a number of environmental factors associated to watershed (vegetation coverage and density, topography, soil type, land cover and land use, etc), hydrology (precipitation, flow, runoff, etc), water quality (salinity, dissolved oxygen, etc) and aquatic optical components (chlorophyll and sediments). Especially, the Vegetation-CO2-DOC-CDOM chain relationship may give us a whole picture of DOC dynamic, transportation and cycle on regional watershed ecosystem, as well as the possible reason and impact of anthropogenic activities on water quality. The SWAT (Soil and Water Assessment Tool) model will serve as a good tool to model the interactions between watershed features, environmental factors, and land use practice. This objective is challenging since the related systems are fairly complex.

However, our goal is to build a model capturing the major factors controlling CDOM flux from Neponset River to Boston Harbor.

Methodology:

(1) In situ measurement:

Our field data collection was conducted in Sept. 25 and Nov. 04, 2009. The in situ CDOM concentration and spectral data were measured on the vessel R/V Neritic, cooperating with researchers of UMass Boston, over the low Neponset River and Boston Harbor (Fig. 1). The data acquisition activities include (1) continuous above-surface hyperspectral measurements, (2) continuous underwater measurements of the IOPs (attenuation and absorption coefficients), salinity, density, dissolved oxygen, UV radiance, CDOM fluorescence, chlorophyll fluorescence, and optical backscattering for suspended sediments, and (3) discrete water sampling and analysis in laboratory.

The water above-surface remote-sensing reflectance was measured by a portable spectroradiometer (ASD FieldSpec®), with a full spectral range (350 – 2500 nm). The spectral sampling interval of output is 1 nm. In the entire cruise, we collected approximate 1,500 hyperspectral samples. The underwater measurements were carried out by the MiniShuttle, a towed, undulating vehicle based on the Nu-Shuttle manufactured by Chelsea Instruments. It is a synthetic function instrument with multiple devices, including a SeaTech fluorometer measuring the fluorescent dissolved organic matter. The resolution of our underwater measurements is very high and we consequently made a large dataset containing about 45,000 samples. In addition, about 25 discrete water samples were also collected and analyzed to calibrate the real time underway measurements.

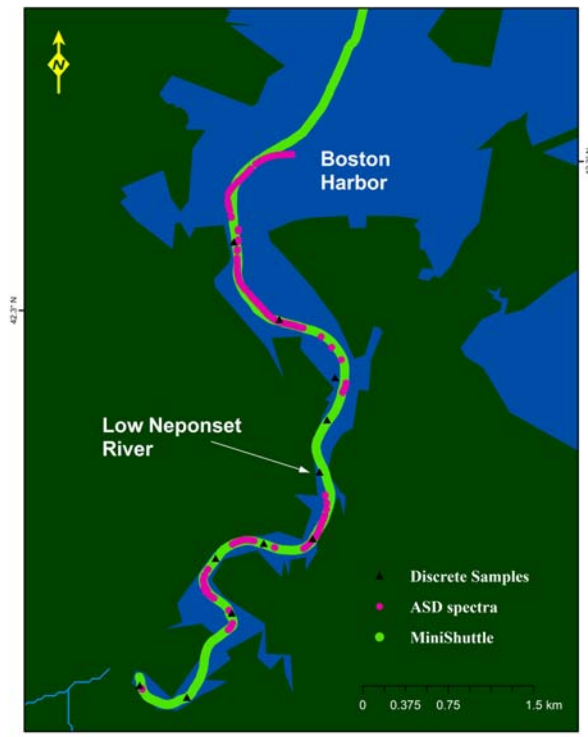


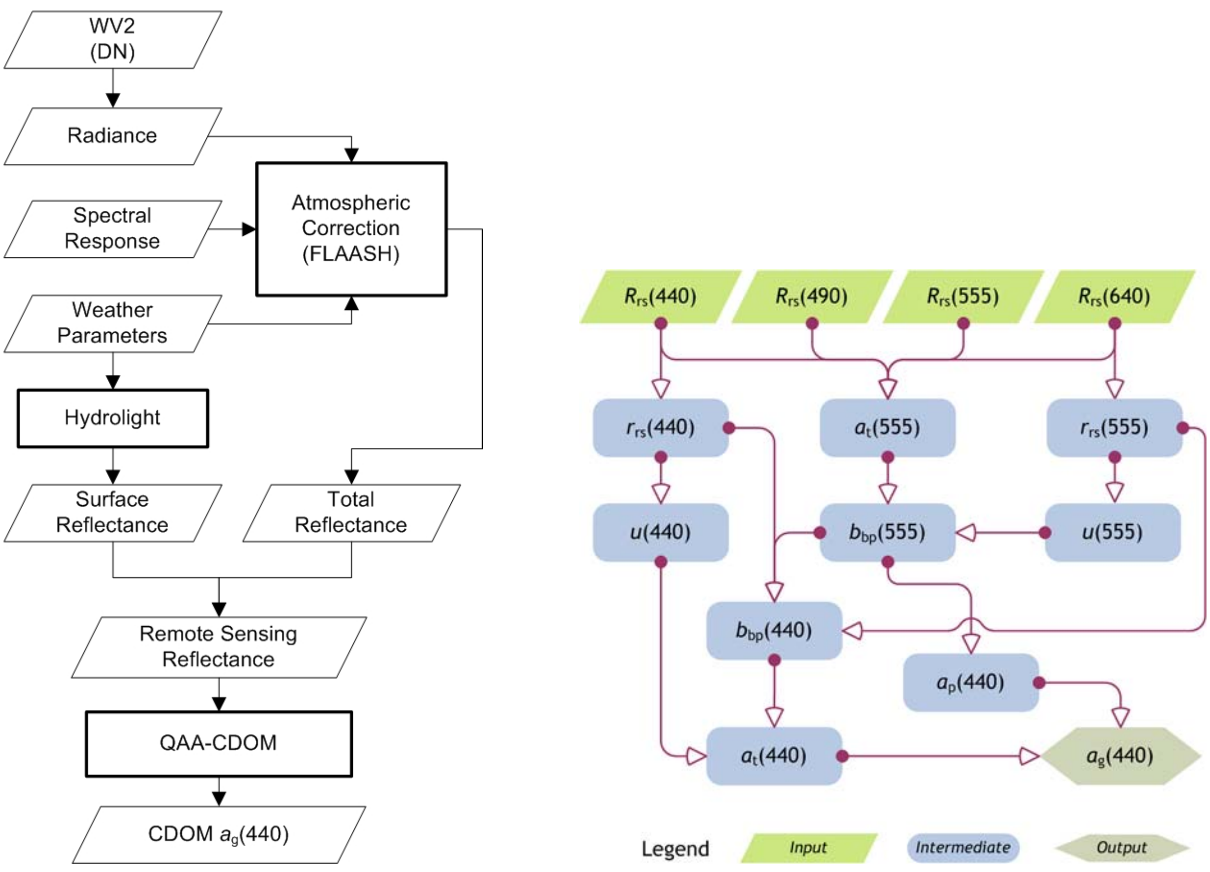
Figure 1. Study site map and in situ measurement tracks of MiniShuttle and ASD, and locations of discrete samples in Low Neponset River and Boston Harbor regions.

The EO-1 Hyperion images in the same regions were requested during fieldtrip dates. The image acquired in Nov. 04 is cloud free. Hyperion images provide a high spectral resolution 400 – 900 nm with 10 nm interval, and a high spatial resolution 30 m. We also acquired a latest WorldView-2 (WV2) image covering our study site. WorldView-2 is a multispectral sensor which bears 8 bands, including a coastal blue band (400 – 450 nm), and also is with very high spatial resolution (~1.8 m).

(2) Remote sensing inversion algorithm and processing:

The whole processing of CDOM remote sensing inversion from WV2 can be referred to the Fig. 2(a). Given a WV2 image, we need to convert the digital number (DN) to the radiance and then make an atmospheric correction, using FLAASH module provided by ENVI®, to obtain the total reflectance, and then compute the remote sensing reflectance, R_{rs}, using HydroLight simulation to remove the water surface reflectance, and finally input these R_{rs}' into remote sensing inversion model to derive CDOM concentration $a_g(440)$ (absorption coefficient at 440 nm). For Hyperion images, some additional preprocessing are needed, including replacing dark lines, destriping and denosing.

To retrieve CDOM's $a_g(440)$, we developed a QAA-CDOM algorithm (Fig. 2(b)). QAA-CDOM is based on Lee's QAA algorithm and its extension, QAA-E. QAA is a quasi-analytical level-by-level algorithm combining a series of empirical, semi-analytical, and analytical algorithms. QAA only requires R_{rs} at several wavelengths (410, 440, 490, 555, and 640 nm) as input data, and at different levels, it outputs r_{rs}, absorption and backscattering coefficients of water's total (a_t, b_{bp}), chlorophyll (a_{ph}, b_{ph}) and CDM (a_{dg}, b_d) (CDOM and NAP together) for all given *Rrs*' wavelengths. QAA has been tested and used in many applications. QAA's output $a_{dg}(440)$, however, has been proven too rough to represent $a_g(440)$ in estuarine and coastal turbid waters, so that QAA's extension, QAA-E, has been developed, in which $a_g(440)$ is exactly derived, using either a_d-based or a_g-based methods. Recently, QAA-E has been further improved to QAA-CDOM in which a QAA's original function and a few parameters has been optimized by integrating synthetic data, very high spatial resolution *in situ* data from our Mississippi cruise and other field data (NOMAD) collected globally during the last decades. QAA-CDOM has been tested with not only excellent inversion accuracy (<25%), but also suitable for high CDOM variability (0.01 – 13.3 m^{-1}).



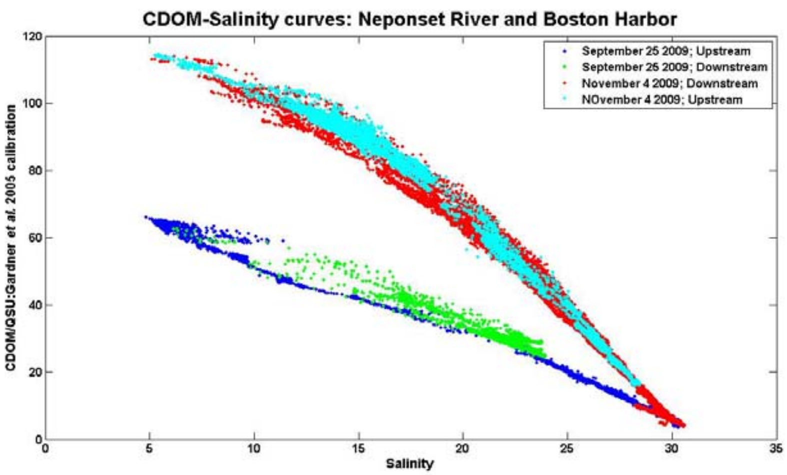
(a) (b)

Figure 2. The simple flow charts of (a) whole processing of WV2 satellite images and (b) QAA-CDOM algorithm.

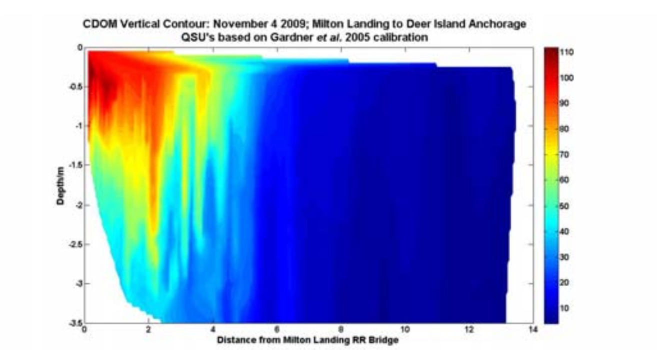
Principal Findings and Significance:

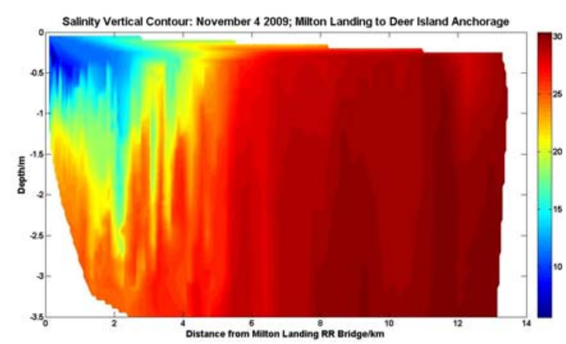
(1) In situ CDOM concentration

According to our measurements in Nov. 4, 2009, CDOM in the Neponset River and Boston Harbor regions ranged from 8.17 to 86.75 QSU, with mean value 34.7 QSU. This wide range shows CDOM in the Neponset River is highly varied complicated. Also, CDOM and salinity demonstrate high negative correlation (Fig. (3)). However, their correlation coefficients change at different times (Sept. 25, 2009 vs. Nov. 04, 2009) but are similar at different locations (upstream vs. downstream), indicating that CDOM seasonal or temporal variations are more significant than its spatial variations.



(a)





(b) (c)

Figure 3. CDOM-Salinity relationship in the Neponset River and Boston Harbor regions. (a) CDOM-Salinity curves; (b) CDOM vertical contour; (c) Salinity vertical contour; (Figure courtesy of G. Bernard Gardner).

(2) CDOM high-resolution remote sensing inversion.

CDOM distribution in the low Neponset River and Boston Harbor has been mapped in very high spatial resolution (~ 1.8 m) from a WV2 image, see Fig. 4. This resultant image and statistical comparisons (Fig. 5) show that QAA-CDOM is able to invert CDOM absorption coefficients with excellent accuracy ($RMSE$ = 0.11, R^2 = 0.73). Our results also indicate that $a_g(440)$ in the Neponset River and Boston Harbor is slightly underestimated, particularly for the fresh water in the upstream. According to our analysis, this underestimation is possibly due to the interference of very high concentrations of phytoplankton (chlorophyll) growing in the same regions.

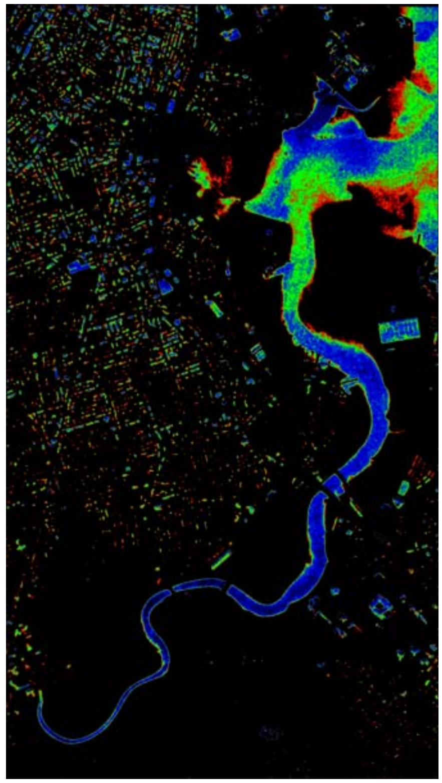


Figure 4. CDOM distribution in the Low Neponset River and Boston Harbor regions, derived from WV2 satellite images.

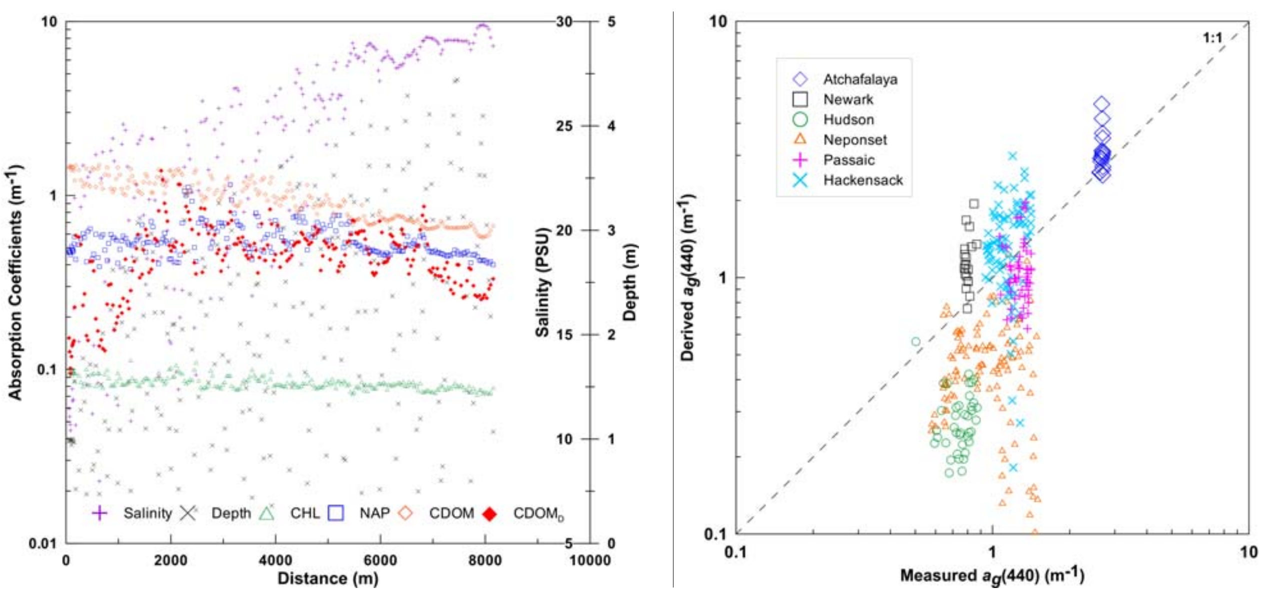


Figure 5. Derived CDOM and properties of other in-water components. (a) Satellite-derived CDOM and in situ measured concentration of CDOM, CHL, and non-algal particles, and water depth and salinity, along our tracks. (b) Comparisons between derived and measured CDOM in six locations, including the Neponset River.

(3) Uncertainty of in situ measurement and remote sensing inversion

The uncertainties related to the whole inversion processing were analyzed in three levels: uncertainties in the processing of in situ measurement (level 1), satellite image preprocessing (level 4) and remote sensing inversion model (level 5), respectively. We found that in the level 1, the uncertainties of the in situ CDOM measurement in the Neponset River and Boston Harbor is approximately 1.547, which is about 10 times larger than our other study sites in the Mississippi River plumes and Hudson River estuary. This result indicates that the true CDOM distribution in the Neponset River is complex and highly varied even in a small water volume. The overall average level 1 uncertainty of our three study sites is 0.262%. This normalized uncertainty is stood on the unit of volts. If we convert it to QSU by multiplying 30, then we get 7.86%. This value, closing to 0.1, implies that in the unit of QSU, keeping 1 digital number is significant for shuttle's measurement. If we further convert it to absorption coefficient, then we obtained for $a_g(440)$, the normalized uncertainty is about 0.32%. Similarly, it indicates that in the unit of absorption coefficient (m^{-1}), the first two digits are significant. If we exclude the Neponset data, the normalized uncertainties for volts, QSU, and absorption coefficient are 0.122%, 3.66%, and 0.22%, respectively.

The uncertainties of level 4 and level 5 are listed in the Table 1 and Table 2, respectively.

Table 1. Uncertainties of above surface spectrum. Star symbol indicates the results were calculated from strong wave areas. The values in the '*Err*' columns multiple 100 is the error percentage. The Err_4 and Err_{49} are the mean value of the 4 bands (440, 490, 555, 640) and the 49 bands of Hyperion sensor.

Err_{49}	Err_4	*Err* (440)	*Err* (490)	*Err* (555)	*Err* (640)	U_{A49}	U_{A4}	U_A (440)	U_A (490)	U_A (555)	U_A (640)
0.128	0.654	1.121	0.655	0.482	0.356	0.009	0.014	0.021	0.013	0.014	0.009

Table 2. Uncertainty of QAA-CDOM remote sensing inversion model.

n	Measured			Derived			U_A	U_{A2}	*Err*% mean	*Err*% Abs mean	Err_{log} mean	RMSE	$RMSE_{log}$
	Min	Avg	Max	Min	Avg	Max							
1143	0.57	0.98	1.53	0.09	0.47	1.42	0.0179	0.0189	-0.488	0.494	-0.346	0.607	0.426

Publications and Conference Presentations:

Provide citations for publications resulting from all projects supported using your grant and required matching funds, including base grants. Please provide the citations in the format requested.

a. <u>Articles in Refereed Scientific Journals</u>

- Zhu, W.N., Q. Yu, Y.Q. Tian, R.F. Chen, and B.G. Gardner, 2011, Estimation of chromophoric dissolved organic matter in the Mississippi and Atchafalaya river plume regions using above-surface hyperspectral remote sensing, *Journal of Geophysical Research-Oceans*, 116, C02011
- Yu, Q., Y. Q. Tian, R.F. Chen, A. Liu, G.B. Gardner and W.N. Zhu, 2010, Functional linear analysis for estimating riverine CDOM in coastal environment using in situ hyperspectral data, *Photogrammetric Engineering and Remote Sensing*, 76(10), 1147–1158. (Early Career Best Paper Award, AAG Remote Sensing Specialty Group)

b. <u>Book Chapter</u>

c. <u>Dissertations</u>

Weining Zhu, Dept. of Geosciences, Inversion And Analysis of Chromophoric Dissolved Organic Matter In Estuarine And Coastal Regions Using Hyperspectral Remote Sensing

d. <u>Water Resources Research Institute Reports</u>

e. <u>Conference Proceedings</u>

f. <u>Other Publications</u>

- Presentation, Weining Zhu and Qian Yu, 2011, *Uncertainty analysis of remote sensing of colored dissolved organic matter: evaluations and comparisons for three rivers in North America*, AAG Annual Meeting, Seattle, WA, April 12-16.
- Presentation, Yu, Q., W.N. Zhu, Y.Q. Tian and R.F. Chen, 2011, *High resolution estimation of colored dissolved organic carbon in riverine and plume area*, AAG Annual Meeting, Seattle, WA, April 12-16.
- Presentation, Tian, Y.Q., Q. Yu, and R.F. Chen, *Sensitivity of terrestrial DOC export to climate change from urban landscape*, AAG Annual Meeting, Seattle, WA, April 12-16.
- Presentation, Weining Zhu, Qian Yu, 2010, *Inversion of chromophoric dissolved organic matter in coastal and estuarine regions using satellite hyperspectral remote sensing*, AAG Annual Meeting, Washington, D.C., April 14-18.
- Presentation, Yu, Q., W.N. Zhu, R.F. Chen, Y.Q. Tian, and G.B. Gardner, 2010, *Examining seasonal variation of CDOM concentration in coastal river using hyperspectral data*, AAG Annual Meeting, Washington, DC, April 14-18.
- Presentation, Tian Y.Q., R.F. Chen, Q. Yu, K. Cialino, W. Huang, and G.B. Gardner, 2010, *Variation of DOC exports from urban watershed to marine water in response to climate change: A case study for the Northeast of the United States*, AAG Annual Meeting, Washington, DC, April 14-18.

Student Support

Number of students supported by grant or matching funds, the degree they are pursuing, and their major.

Name: Weining Zhu
Degree: Ph.D. candidate
Major: Geosciences

Notable Achievements and Awards

Provide a brief description of any especially notable achievements and awards resulting from work supported by section 104 and required matching funds and by supplemental grants during the reporting period.

Using this grant as seed fund and preliminary study, we submitted a NSF proposal and it was successfully funded.
Qian Yu (PI), Co-PI: Anna Liu, Collaborated with Yong Tian and Robert Chen at UMass-Boston, *Collaborative Research: Modeling DOC dynamics from landscapes to coasts: hydrological connectivity and estuary processes*, NSF Collaboration in Mathematical Geosciences (CMG), #1025547, $517,987 ($329,346 on Amherst Campus), Sept 2010 - Aug 2013.

Surface water-groundwater interactions on the Deerfield River

Basic Information

Title:	Surface water-groundwater interactions on the Deerfield River
Project Number:	2010MA237B
Start Date:	3/1/2010
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	1
Research Category:	Ground-water Flow and Transport
Focus Category:	Surface Water, Groundwater, Methods
Descriptors:	
Principal Investigators:	David Boutt

Publications

There are no publications.

Annual Report (from March 1, 2010 to February 28, 2011)

Title: Surface water-ground water interactions on the Deerfield River

Project Number: 2010MA237B

Start Date: 3/1/2010

End Date: 2/28/2011

Funding Source: NIWR 104B Competition Massachusetts Water Resources Center

Research Category: **GroundWater Flow and Transport**

Focus Categories: **SW GW MET**

Descriptors:

Primary PI: David F. Boutt

Problem and Research Objectives: Near daily hydroelectric dam releases from Fife Brook Dam in Rowe, MA raise the stage of the river and alter downstream interaction between water in the river and that in the riparian aquifer. Several studies (Sawyer 2009, Arntzen 2006) have shown that these anthropogenic "flood" events in a typically gaining river setting can reverse the hydraulic gradient between the stream and the aquifer, causing the stream to temporarily lose water as the dam release flood-wave passes. Bank storage, the natural analogue of this hydraulic gradient reversal, is an important phenomenon for attenuating hydrograph spikes during natural floods. Past work on anthropogenic stage increases has typically focused on an individual site below a dam, thereby oversimplifying the downstream heterogeneity of stream and aquifer morphologies. In this study, the goal was to investigate how different geologic settings along a 20km reach of river may influence the magnitude of dam-induced bank storage events. We hypothesized that areas of greater aquifer transmissivity would account for greater exchange of water between the river and adjacent riparian aquifer.

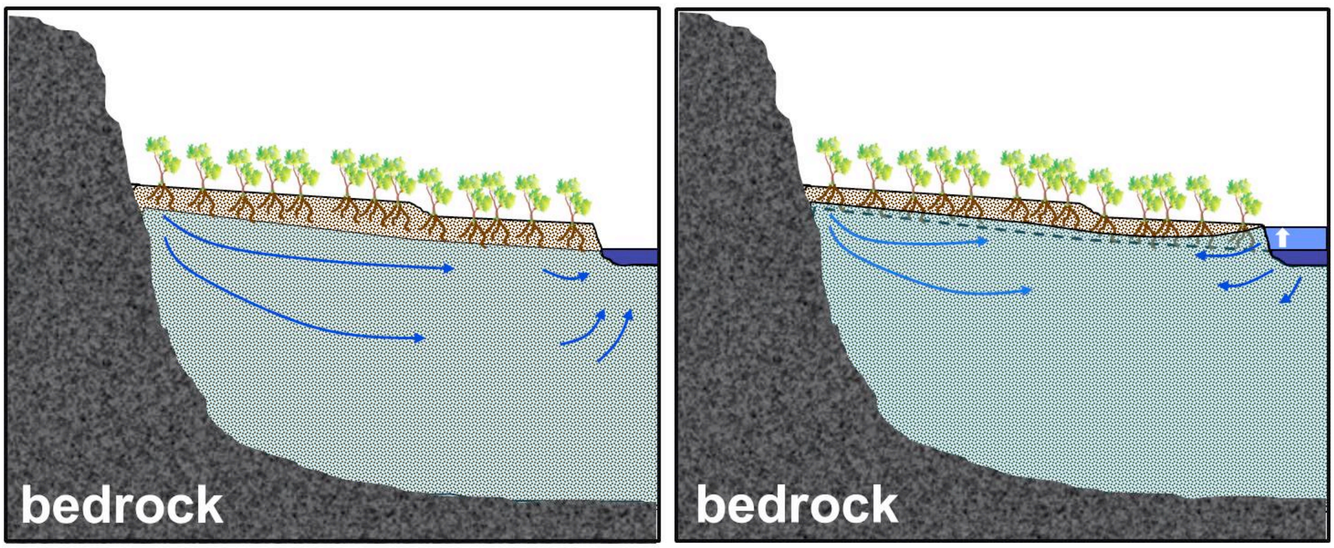


Figure 1 - At low stage (left) the water table slopes towards the river and the river is gaining. The hydraulic gradient is reversed during a dam release, causing the river to lose.

Methodology: Three distinct strategies were employed to better understand the controls on dam-induced bank storage: 1) Several sites were selected for in-stream monitoring of water fluxes across the river bed. Direct Darcy-based methods with piezometers and pressure transducers provided direct observation of the hydraulic gradient between river water and groundwater. Vertical thermistor arrays collected temperature profiles of the subsurface to enable the use of river heat as a natural tracer, giving us a second means by which to observe fluxes across the riverbed. 2) Two-dimensional finite-element modeling of the riparian aquifer was used to simulate bank storage events to evaluate how different aquifer dimensions mediate surface water-groundwater exchange. 3) A water budget for the 20km river reach was created by accounting for known inputs to and outputs from the river to evaluate how dam-induced bank storage affects the river at the reach scale (see figure 2).

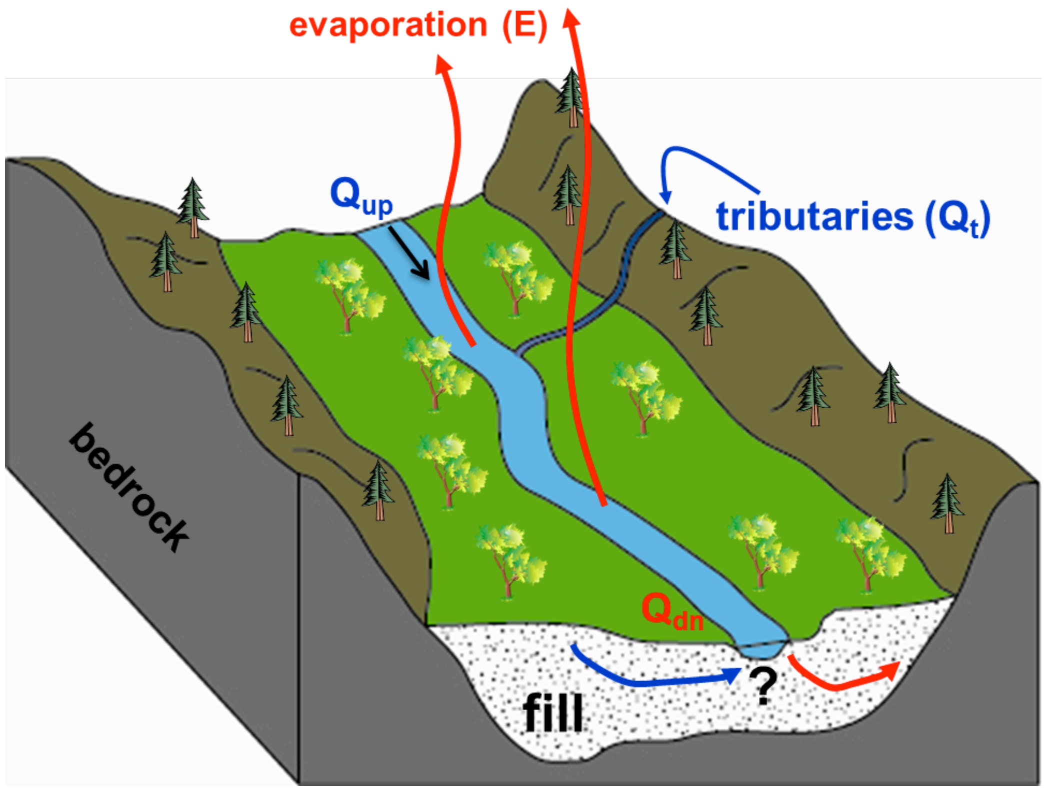


Figure 2 - The water budget for the 20km river reach accounted for inputs from the upstream dam and tributaries, and outputs from the downstream end of the reach and direct evaporation.

Principal Findings and Significance: From June through September, the 20km study reach of the Deerfield River below Fife Brook Dam loses 5-10% of its discharge. Unregulated rivers in this region typically gain during all seasons. Water budget analysis of the entire reach, as well as direct observations from in-stream piezometers and vertical temperature arrays all indicate that the river loses water in a variety of geologic settings.

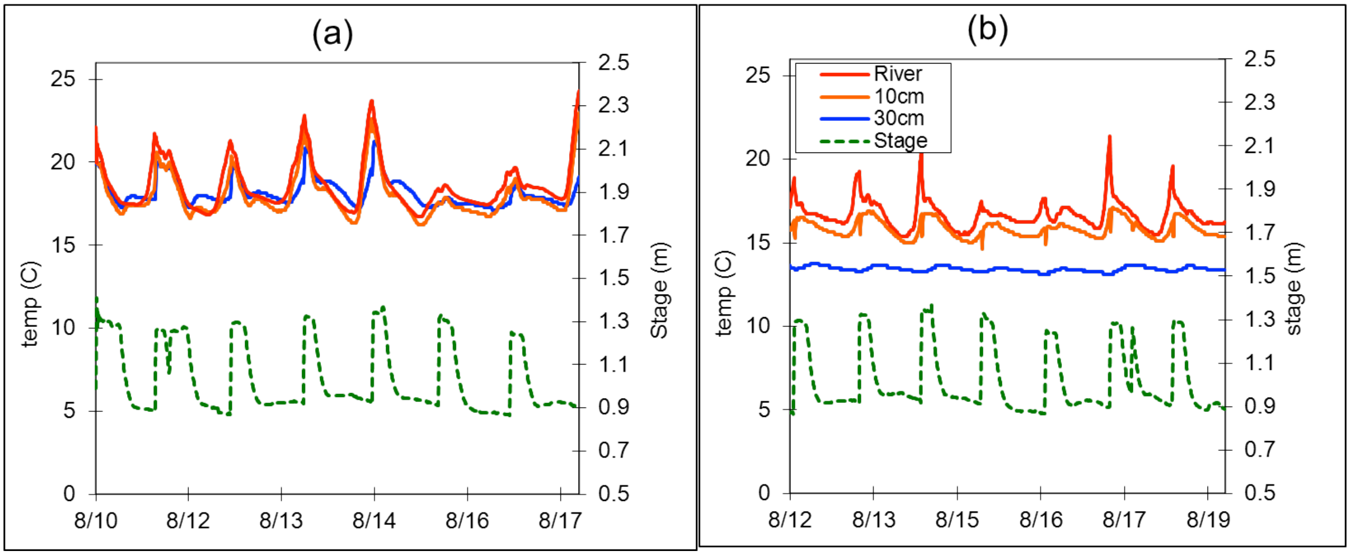


Figure 3 - Vertical temperature profiles at two sites in the Deerfield River study reach. Slug tests showed that the riverbed at site (a) had much higher hydraulic conductivity than that at site (b). The temperature of the subsurface therefore varies a lot at site (a) due to the close hydraulic connection between surface water and groundwater there.

Field data and two-dimensional simulations suggest that the width and conductivity of the valley aquifer have the greatest control on losses. We have identified a strong correlation between water loss during a given event and evaporative forcing, suggesting that water driven into the bank during high stage is made available to riparian vegetation and lost permanently due to transpiration. This finding has two major implications: 1) Hydroelectric operators with multiple facilities on a single river face a tradeoff when generating electricity on days with high evaporative forcing; 2) Minimum flow requirements for in-stream fauna may need to be adjusted considering that the discharge at some distance downstream can be reduced below that which is released from the dam.

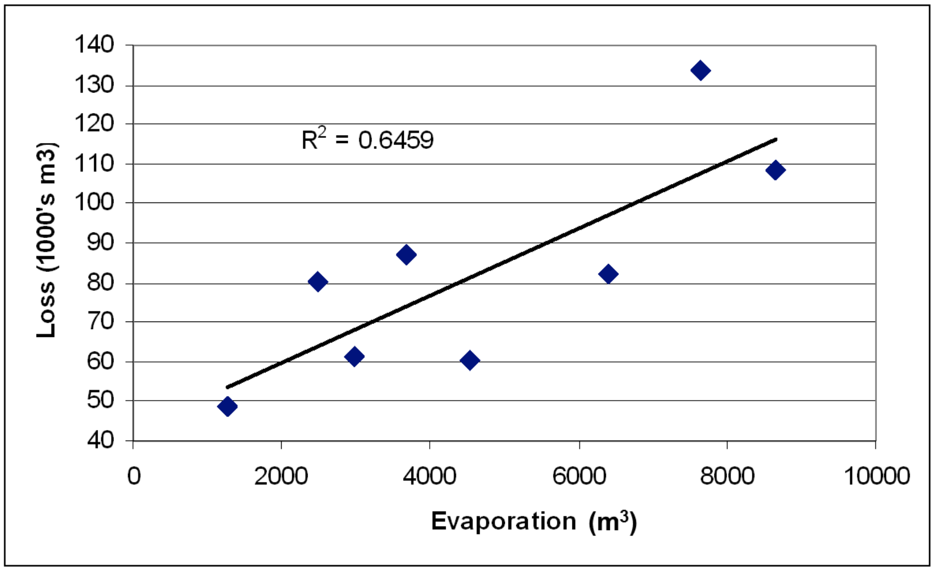


Figure 4 - Total direct evaporation from the river surface during an individual flood event is a good predictor of loss during that same event. Data from a nine-day period in July, 2010 are plotted here.

Publications and Conference Presentations:

a. Articles in Refereed Scientific Journals

b. Book Chapter

c. Dissertations

d. Water Resources Research Institute Reports

e. Conference Proceedings

Water Resources Conference, Amherst, MA. *Can hydroelectric dams cause a river system to lose water?* April 7, 2011.

American Geophysical Union Annual Fall Meeting, San Francisco, CA. *A reach-scale study of dam-induced hyporheic exchange: controlling mechanisms and effects, Deerfield River, Massachusetts.* December 13, 2010.

f. Other Publications

Student Support

One full-time graduate student.

Notable Achievements and Awards

Provide a brief description of any especially notable achievements and awards resulting from work supported by section 104 and required matching funds and by supplemental grants during the reporting period.

Citations

Arntzen, E.V. 2006, *Effects of fluctuating river flow on ground water/surface water mixing in the hyporheic zone of a regulated, large cobble bed river*, Wiley : Chichester, United Kingdom, United Kingdom.

Sawyer, A.H. 2009, *Impact of dam operations on hyporheic exchange in the riparian zone of a regulated river*, Wiley : New York, NY, United States, United States.

Impact of the hemlock woolly adelgid on the water cycle in New England: Differences in hydrologic fluxes between hemlock and deciduous forest stands

Basic Information

Title:	Impact of the hemlock woolly adelgid on the water cycle in New England: Differences in hydrologic fluxes between hemlock and deciduous forest stands
Project Number:	2010MA241B
Start Date:	4/1/2010
End Date:	12/31/2010
Funding Source:	104B
Congressional District:	2nd (Northampton)
Research Category:	Climate and Hydrologic Processes
Focus Category:	Hydrology, Invasive Species, Ecology
Descriptors:	None
Principal Investigators:	Andrew J Guswa

Publications

1. Guswa, Andrew J. and C. M. Spence ‘11, in review. Effect of vegetation change on throughfall patterns and recharge: Application to hemlock and deciduous forests in western Massachusetts, submitted to Ecohydrology.
2. Spence, C. M., 2011. Biotic and abiotic factors affecting throughfall volume and spatial variability in a New England forest, undergraduate Honors Theses, Smith College.
3. Spence, C. and A. J. Guswa, 2011. Biotic and abiotic factors affecting throughfall volume and spatial variability in a New England forest, MA Water Resources Research Center, 8th Annual Conference, University of Massachusetts, Amherst, MA, 7 April 2011.
4. Guswa, Andrew J., M. Mussehl ‘12, A. Pecht ‘12, and C. Spence ‘11, 2010. Spatial pulses of water inputs in deciduous and hemlock forest stands, Eos Trans. AGU, Fall Meeting Suppl., Abstract B24B-08.

Problem and Research Objectives:

Invasive pests, especially in conjunction with climate change, have the potential to transform the species composition of many forests. In the northeastern United States, the hemlock woolly adelgid (HWA) poses a significant threat to eastern hemlock (Tsuga canadensis). This pest arrived in western Massachusetts over the last ten to fifteen years, is steadily making its way into southern Vermont and New Hampshire, and often kills infested hemlocks within a few years (National Forest Service, 2009).

Replacement of hemlock forests by other species, such as birch, maple, and oak, may alter the hydrologic cycle and impact water resources. Changes to hydrologic fluxes include both the input of water, which is affected by canopy interception, and the uptake of water for transpiration. Canopy interception affects not only the mean input of water but also its distribution in space, which has implications for hydrologic and geochemical processes that are non-linear functions of soil moisture, such as drainage below the root zone and nitrification/denitrification. To better understand the impact of HWA invasion on the hydrologic cycle, we must understand how hemlock and deciduous forest stands differ with respect to canopy interception and water uptake.

This proposal seeks to build upon and complement these early findings to better understand differences in hydrologic fluxes between hemlock and deciduous forests. The objectives of this research project were to

- Quantify the difference in average interception between hemlock and deciduous stands
- Quantify the spatial variability of throughfall in hemlock and deciduous stands
- Quantify differences in summertime water use for hemlock and deciduous stands

Methodology:

During the summer of 2010, a cohort of three undergraduates engaged in a two-month field campaign to measure and characterize hydrologic fluxes in hemlock and deciduous forest stands. The field work was carried out at the Ada and Archibald MacLeish Field Station, a 240-acre site maintained by Smith College and located adjacent to the primary reservoir that supplies drinking water to the City of Northampton. Ongoing monitoring at this site includes continuous measurements of precipitation, air temperature and pressure, relative humidity, solar radiation, and wind speed and direction.

We established two hemlock sites and two deciduous sites and instrumented them to measure throughfall. Additionally, we characterized the trees and vegetation in each plot (e.g., stem location, species). Our hemlock sites are situated within permanent vegetation plots that were established on the property in 2009. Within these 20m x 50m plots, all trees and saplings > 1.4 m have been tagged and measured for diameter at breast height (DBH); individual trees are being tracked for growth and survival in coming years. Tree coring is being used to reconstruct stand histories and to assess hemlock population dynamics over time.

At both the hemlock and deciduous sites, we deployed thirty stationary throughfall collectors over a ten-meter by ten-meter plot. In two sites (one hemlock and one deciduous), these collectors were arranged in a regular grid with 1.5-meter spacing; in the two other sites, the collectors were placed using a stratified random design. Throughfall volumes were measured following each precipitation event.

To assess water use, the students developed instrumentation to measure sapflux using the heat-pulse method. These sensors use the transport of a heat pulse to infer water velocity in the xylem. Due to unforeseen challenges in the development and use of these sensors, we only got as far as field testing the sensors in a single tree, and we obtained no useable sapflux data during the summer of 2010. Three undergraduates will continue this work during the summer of 2011.

Principal Findings and Significance:

The measurement and analysis of throughfall during the summer of 2010 complements a prior throughfall study carried out in 2009. From 3 June through 25 July 2009, fourteen precipitation events generated 311 mm of rain; 2010 was much drier with eight rain events generating 148 mm of precipitation over the campaign.

In 2009, stand-average throughfall amounted to 276 mm (89% of precipitation) in the deciduous plot and 242 mm (78%) in the west hemlock stand; in 2010, stand-average throughfall totals were 129 mm (87%), 123 mm (83%), and 126 mm (85%) in the deciduous, west hemlock, and north hemlock stands, respectively. On an event-by-event basis, the throughfall fraction increases with precipitation amount, and representing interception as a threshold depth, D, provides a good fit. These threshold depths are lower (2.4 mm and 2.5 mm) for the deciduous stand than for the hemlock stands (5.0 mm, 3.5 mm, and 3.2 mm).

With the exception of very light events (i.e., less than 5 mm of precipitation), the spatial variability among collectors, as measured by the coefficient of variation or the ratio of the interquartile range to the median, is insensitive to precipitation amount. Additionally, wet and dry spots tend to persist over the season and even from year to year, and analysis of variance confirms that collector position is highly significant as a predictor of normalized throughfall amount. While persistent through time, the spatial patterns exhibited no discernable spatial correlation structure. Moment-based statistics of spatial variability are strongly influenced by extremes, and the deciduous stands show positive skewness among collector totals (i.e., some very wet spots), while the hemlock stands exhibit negative skewness (i.e., dry spots).

An assessment methodology for differential impact on environmental justice populations of releases of industrial toxics to water in Massachusetts

Basic Information

Title:	An assessment methodology for differential impact on environmental justice populations of releases of industrial toxics to water in Massachusetts
Project Number:	2010MA248B
Start Date:	4/1/2010
End Date:	7/30/2010
Funding Source:	104B
Congressional District:	Massachusetts 1st Congressional District
Research Category:	Not Applicable
Focus Category:	Water Quality, Toxic Substances, Methods
Descriptors:	
Principal Investigators:	Michael Ash

Publications

There are no publications.

Annual Report (from March 1, 2010 to February 28, 2011)

Title: An assessment methodology for differential impact on environmental justice populations of releases of industrial toxics to water in Massachusetts

Project Number: USGS 06HQGR0091

Start Date: 03/01/2010

End Date: 02/28/2011

Funding Source: GEOLOGICAL SURVEY USGS

Research Category:

Focus Categories: ECON, LIP, MET, TS, WQL

Descriptors: environmental justice, water toxics, drinking water

Primary PI: Michael Ash, mash@econs.umass.edu

Problem and Research Objectives:

Release of toxic chemicals into surface water by industrial facilities in Massachusetts puts the water of communities at risk and potentially impacts human health. Previous research on environmental justice (EJ) has found that low income and minority populations are disproportionately impacted by industrial air pollution and by the siting of hazardous waste facilities, but no methodology exists for assessing the differential impact of surface-water releases of industrial toxics on EJ populations. We will examine social and economic characteristics of communities in proximity to facilities releasing toxics to surface water in Massachusetts. The methods and metrics developed for assessing community exposure and environmental justice in industrial water pollution will be applicable to the entire United States.

Methodology:

U.S. EPA's Risk Screening Environmental Indicators (RSEI) generates annual toxicity-weighted hazard measures for every industrial facility based on the mass and toxicity of releases into different media (air, water, ground) reported to U.S. EPA's Toxics Release Inventory (TRI). This study focuses on releases to surface water. We geo code U.S. Census data to determine the proximity of each census tract to facilities identified as high hazard based on the surface-water releases. The data permit examination of both how sources of toxic releases differ in their disproportionate impacts and also how communities differ in their cumulative exposure to toxic releases. We generate facility-based estimates of toxic water pollution weighted by relative toxicity and examine the demographics of communities in proximity to each facility. By comparing the demographics of nearby communities to the demographics of the entire Commonwealth, we measure disparities in exposure to water toxic releases. Additionally, we model the probability that a census tract is near a toxic releasing facility based on its demographic characteristics.

This project draws on the Toxic 100 Water Polluters, an index developed by the Political Economy Research Institute at the University of Massachusetts Amherst in partnership with Food & Water Watch, a non-profit consumer organization.

U.S. EPA has recently revised the portion of the RSEI methodology that models the fate and transport of toxics released into surface water. The RSEI water geographic micro data, which reports the estimated concentration of every toxic release to surface water in every downstream flowline, has not previously been used in research. These data, which became available in September 2010, permit more specific measurement of community exposure to industrial toxic releases to surface water than previous methods based on simple proximity to releasing facilities.

Results Expected

The primary objective of this research is to develop measurements of the disproportionate impacts on EJ communities of toxics released into nearby surface water. We developed measurements based both on community proximity to toxic releasing facilities and of exposure to drinking water supplies. The results will provide a baseline assessment which will allow us to compare releases and impacts annually in order to determine if policies, enforcement efforts and community action are effective in reducing the impact of toxic releases on exposed populations.

Principal Findings and Significance:

The project successfully developed a methodology for assessing disproportionate impacts on EJ communities of toxics released into nearby surface water and applied the methodology to Massachusetts. First we present some results and then discuss the methodology. These results have not yet been peer reviewed and should be considered provisional.

There are roughly 88,500 Census Blocks in Massachusetts. Census Blocks correspond to city blocks in urban areas and can be somewhat larger in rural areas; the average Census Block in Massachusetts has 72 residents, but there is variation. Using our method to classify Census Blocks by proximity to waterways that were highly polluted by industrial releases in 2007, we identified 204 “extremely polluted Census Blocks” associated with the top five industrially polluted flowlines; 146 of these Census Blocks are in Suffolk County, 45 are in Middlesex County, and 13 are in Franklin County. We then analyzed the racial and ethnic composition and average household income in the extremely polluted Census Blocks and the rest of Massachusetts. The Census Blocks with extremely high impact from industrial toxic water releases are substantially less (non-Hispanic) white (56 percent versus 84 percent) and more Hispanic (35 percent versus 6 percent) than is the rest of the state. The percentage black is similar in highly impacted and other Census Blocks. The highly impacted Census Blocks are also very much poorer: the median household income is almost $20,000 less in the most impacted Census Blocks.

	All MA blocks (N=88,307)		Extremely polluted blocks (N=204)	
	Mean	Std. Dev.	Mean	Std. Dev.
percent white	83.9%		56.0%	
percent black	4.6%		2.6%	
percent hispanic	5.6%		35.4%	
median income	$55,090	$24,200	$38,450	$16,660

The table focuses on the most impacted Blocks and implies that there is significant disparity in water pollution exposure by race, ethnicity, and class.

Regression analysis can test whether the EJ relationship appears pervasively in the state and whether the race/ethnic and income relationships appear independently from each other. Specifically, we modeled how the racial, ethnic, and income composition of a Census Block is associated the likelihood of a Census Block appearing among the most impacted 10 percent, 5 percent, or 1 percent of Massachusetts Blocks that abut a waterway. (We also examined the effect of socio-economic composition on the likelihood of any exposure and on a continuous measure of exposure, as well as other specifications. We present illustrative results here.)

	top 10%	top 5%	top 1%
Percent Black	-0.257***	-0.205***	-0.0590***
	(0.0245)	(0.0151)	(0.00883)
Percent Hispanic	0.265***	0.319***	0.153***
	(0.0319)	(0.0291)	(0.0208)
Percent Asian	0.0756*	0.0840**	-0.0318***
	(0.0453)	(0.0365)	(0.0103)
Percent Native American	-0.462***	-0.291**	-0.0825
	(0.160)	(0.116)	(0.0679)
Median income ($10K)	-0.0296***	-0.000134	0.000739
	(0.00408)	(0.00277)	(0.00158)
Median income squared	0.00111***	-0.000388**	-0.000108
	(0.000241)	(0.000153)	(8.81e-05)
Constant	0.241***	0.0687***	0.0113*
	(0.0155)	(0.0109)	(0.00650)
Observations	58,108	58,108	58,108
R-squared	0.027	0.029	0.020

These results indicate that compared to non-Hispanic whites, African Americans are substantially less likely to live in the most polluted Census Blocks near water. Hispanics are much more likely to live in the most polluted Census Blocks that abut water. A Block that is 100 percent Hispanic rather than 0 percent Hispanic is more than 30 percentage points more likely to appear in the top 5 percent of Census Blocks and 15 percentage points more likely to appear in the top 1 percent. (A randomly selected Block is by definition 5 percent likely to appear in the top 5 percent of Census Blocks; so the Hispanic effect is quite strong.) A high percent Asian increases the probability that a Block is among the top 10 percent or top 5 percent but not in the most polluted 1 percent. Because of the relatively small population in Massachusetts, the Native American results are estimated with limited precision but nonetheless indicate that percent Native American is associated with lower likelihood of high pollution exposure.

The income results imply a significant protective effect of income with respect to the likelihood of a Block being in the most polluted 10 percent of Blocks abutting water. Furthermore, it should be noted that the race and ethnicity results control for income. So the effect of high percentage Hispanic is not explained by income differences between Hispanics and non-Hispanics.

The results described above required the development of new methods for matching polluted stream reaches to Census geography and operationalization of the concepts exposure and differential exposure for industrial toxic releases to water. The key dataset is the U.S. EPA's Risk Screening Environmental Indicators, Geographic Microdata, for Water (RSEI-GMW) for 2007. This dataset provides the estimated concentration in every downstream flowline of every 2007 TRI release to surface water. The RSEI model also provides oral toxicity weights for each of the approximately 600 TRI chemicals that make it possible to add toxicity-weighted concentrations of different chemicals. Details of the RSEI model are available from U.S. EPA http://www.epa.gov/oppt/rsei/pubs/index.html and there is a summary of the RSEI model on the Corporate Toxics Information Project website at http://www.peri.umass.edu/ctip.

Because the aim is to provide inter-area assessment for very large areas (the Commonwealth of Massachusetts in this project, but the methods are scalable), GIS is not feasible for matching polluted reaches to Census geography. There are approximately 8 million Census Blocks in the United States, 3 million flowlines in the NHDPlus (EPA's augmented version of the National Hydrography Dataset), 187,000 impacted flowlines and 123 million distinct release-flowline impacts, or estimated concentrations in flowlines from industrial toxic releases, in RSEI-GMW. Although GIS data for the components exist, spatial joins are impractical or impossible on this scale. Instead, we extracted longitude and latitude data for every point on every flowline from the NHDPlus, converted each point to a corresponding 1-kilometer cell in the RSEI geography, and employed the existing crosswalk between RSEI geography and the Census. In this way, we were able to assign pollution concentration estimates, by specific source industrial facility and chemical, to each U.S. Census Block. (Blocks that do not abut water have zero concentration as do Blocks that abut water but are not affected by any upstream industrial releases.)

The extraction and matching methodology was carried out with perl and MySQL, and scripts are available on request. The scripts, in particular, those that extract and process point data from the NHDPlus shapefiles, may prove useful for a variety of applications. Other scripts are more specifically focused on RSEI applications, and these too are available for researchers.

Student Support (Number of students supported by grant or matching funds, the degree they are pursuing, and their major.)

Four (4) students received support as research assistants from matching funds associated with this project:
Grace Chang, Ph.D. student in Economics, UMass Amherst
Robin Kemkes, Ph.D. student in Economics, UMass Amherst
Helen Scharber, Ph.D. student in Economics, UMass Amherst
Owen Thompson, Ph.D. student in Economics, UMass Amherst

Notable Achievements and Awards

Provide a brief description of any especially notable achievements and awards resulting from work supported by section 104 and required matching funds and by supplemental grants during the reporting period.

1. The Corporate Toxics Information Project received the second year of a two-year grant from Food and Water Watch, a Washington, D.C.-based non-profit organization to further develop

methodology for the assessment of population risk and environmental-justice disparities from toxic industrial water pollution reported in the U.S. EPA's Toxics Release Inventory and the Risk Screening Environmental Indicators model.

2. James K. Boyce (PI) and Michael Ash (co-PI) submitted a successful proposal to the National Science Foundation in collaboration with researchers at the University of Southern California and the University of Michigan to continue research on environmental justice with the Risk Screening Environmental Indicators geographic microdata.
3. Michael Ash (PI) submitted a proposal and received an allocation on TeraGrid, National Science Foundation's effort to build and deploy the world's largest distributed infrastructure for open scientific research, to use TeraGrid resources to manage the databases for the project.

Developing a physically-based and policy-relevant river classification scheme for sustainable water and ecosystem management decisions.

Basic Information

Title:	Developing a physically-based and policy-relevant river classification scheme for sustainable water and ecosystem management decisions.
Project Number:	2010MA253B
Start Date:	3/1/2010
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	MA-009
Research Category:	Climate and Hydrologic Processes
Focus Category:	Hydrology, Ecology, Economics
Descriptors:	dam removal, river restoration, ecosystem management
Principal Investigators:	Ellen Marie Douglas, Bob Bowen

Publications

1. Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert, Does current policy help or hinder dam removal decision-making in Massachusetts: a survey of stakeholders, J. Environmental Management, anticipated submission, Jul 2011.
2. Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert. A framework for valuing the environmental costs and benefits of dam removal, Water Resources Management, anticipated submission Sep 2011.
3. O'Brion, K., E. Douglas and A. Christian, Biomonitoring the Eel River headwaters restoration project, Plymouth, Massachusetts, Freshwater Biology, anticipated submission, Jan 2012.

Annual Report (from March 1, 2010 to February 28, 2011)

Title: Developing a physically-based and policy-relevant dam removal decision framework for sustainable water and ecosystem management decisions.

Project Number: 2010MA253B

Start Date: March 1, 2010

End Date: February 28, 2011

Funding Source: NIWR

Research Category: Ecological impacts

Focus Categories: Hydrology, Ecology, Economics

Descriptors: ecosystem integrity, ecosystem services, aquatic habitat, streamflow, dam removal

Primary PI: Douglas, Ellen Marie
Assistant Professor, Hydrology, University of Massachusetts Boston
email:ellen.douglas@umb.edu; phone: 617-287-7437

Problem and Research Objectives:

One of the biggest human impacts on rivers has resulted from the building of dams. The number of dams worldwide has been estimated at 40,000 large (>15 meters in height) and more than 800,000 smaller ones (Petts, 1984; McCully, 1996). Dam operations have caused ecological changes in riparian ecosystems at all scales; how to balance the needs of aquatic and riparian ecosystems and humans remains one of the most important questions of our time (Nilsson and Berggren, 2000). There are 2,964 dams in the Massachusetts dam inventory database. Over half of these dams are privately owned and nearly a third is municipally owned. Most of the dams in Massachusetts are low head, "run-of-the-river" dams that no longer serve the purpose for which they were built. The presence of these dams has fragmented aquatic and riparian ecosystems, impeded fish passage and generally impacted the natural ecological and hydrological functioning of the streams in which they reside. Dam removal should be considered when a dam no longer serves its function. Facilitating dam removal is a major focus of the Massachusetts Division of Ecological Restoration (DER; see http://www.mass.gov/dfwele/river/programs/priorityprojects/projectlist.htm). The removal of a dam incurs many environmental benefits (i.e., enhanced fish passage, restoration of aquatic and riparian habitats, return to a more natural flow regime), but sometimes it can incur environmental costs as well (i.e., release of contaminants that were sequestered behind the dam). In many cases, dam removal is less costly than dam maintenance or upgrade, hence dam removal decisions tend to be based on purely monetary considerations, and the environmental costs or benefits associated with the dam are not fully considered. Furthermore, dam removal projects can be delayed or completely derailed by the *perception* that doing so will result in the loss of aesthetic, recreational and property values associated with the impoundment behind the dam. While dam removal is a high priority in Massachusetts as well as across New England, the true cost of these efforts, which include direct (economic), indirect (environmental) and cultural (recreation, aesthetic) costs, are not well understood and hence are usually not well quantified in dam removal decisions.

The main challenge of water resource management is to find a balance between the use of resources as a basis for human livelihood and the protection and conservation of the resource to sustain its ecosystem functions and benefits. In assessing ecological condition, whether it be in terms of ecosystem health, service value or integrity, it is necessary to consider economic, social and political influences, since these are often the driving factors in environmental protection and restoration decisions. One of the ultimate goals of this proposed research is to provide a framework for estimating the total ecosystem service value that any particular riverine habitat has to the humans and ecosystems that benefit from it. For this one year project, we focused on the dam removal. We developed a decision support framework that is both physically-based and policy-relevant, and useful to environmental managers and policy-makers in dam removal decisions. The specific objectives of this research were to:

- estimate the economic costs and benefits of dam removal, including estimates of value of ecosystems that are impacted.
- develop a comprehensive framework that incorporates both physical (construction-related) and non-physical (market-related and environmental) costs and benefits
- test this framework on one or more upcoming dam removal projects in Massachusetts that are being facilitated by the DER.

Methodology

This one year project was comprised of three tasks: 1) a survey to assess the current status of the dam removal decision process; 2) development of a conceptual framework for dam removal decisions that accounts for environmental costs and benefits and 3) applying this framework to a case study, namely the removal of two dams on the Ipswich River in northeastern Massachusetts.

Task 1: Stakeholder Survey. Before we could develop an improved dam removal decision-making framework, we first needed to assess how well the current decision process does or does not work and what stakeholders believe are the most important aspects of this process. We developed set of nine questions that were sent via SurveyMonkey (www.surveymonkey.com), a free on-line survey tool, to eleven stakeholders. The stakeholders, including dam owners, town managers, consultants, state and federal agencies and watershed associations, were chosen because they were involved in some aspect of the dam removal process in Massachusetts. Table 1 presents the survey questions and the range of response choices.

Task 2: Conceptual Framework an ideal dam removal process. The results of the stakeholder survey was summarized and analyzed. A conceptual framework for an ideal dam removal process and a comprehensive accounting of the costs and benefits of dam removal was developed. This was done using the information gained from the survey and by adapting a framework for preliminary economic assessment of dam removal on the Klamath River in California, presented by Kruse & Scholz (2006*;* with the authors approval). In addition, existing feasibility studies for dam removals in Massachusetts and Maine were perused for environmental benefits, which are typically qualitatively considered, but not quantitatively assessed.

Task 3: Cost-Benefit Analysis for dam removal on the Ipswich River. A cost-benefit analysis of removing two dams located within Ipswich River Watershed was performed. In this analysis, environmental and social costs and benefits are included, but are not yet fully quantified. The benefit transfer method was used, which takes an estimated value from another study and adopts it as the transfer value for the new analysis. Benefit Transfers is a less expensive and less time-consuming method. However, there will always be some error in the transfer application. Benefit transfer's limitations are related to a lack of accessible, unbiased information, difference in methods, and differences in geographic and socioeconomic conditions.

Table 1. Survey Questions

1.You represent:
Private sector
Public sector (choose one below)
- State agency
- Federal agency
Please indicate your affiliation:
2. Do you think that dam removal is a necessary process? Why is that?
3. In your opinion, the decision of removing a dam is based on:
Funding
Improving fish habitat
River restoration
Liability
Others (please, specify)
1-Very significant; 2-Significant; 3- Somewhat significant; 4- Not significant
4. In your opinion, what is the most important outcome of the dam removal?
5. If you were the only decision maker, how would you decide which dam to be removed first?
6. Do you think that the existing legislation regarding dam removal makes this process an efficient one? If this is not the case, what other regulations would you add or/and would you change if you had the power to do so?
7. Why do you think the dam removal process has grown so much over the last decade?
8. Who do you think should decide when and what dam to be removed? Why is that?
9. Would you consider being part of a future survey?

Principal Findings and Significance

Survey Results. Only seven of the eleven stakeholders have responded to date. The majority of the responders (six out of seven) ranked funding as a "very significant" factor in the decision making process. Improving fish habitat is the second most important factor, and it is also listed as the most important outcome of the dam removal by five out seven respondents. Three of the responders ranked all of the factors as very significant factors in the decision making process. Two responded that liability and river restoration are "significant" or "somewhat significant". Only one respondent ranked river restoration as “not-significant”. While funding appears to be the most significant factor in the decision making process, river restoration and improving the fish habitat are the main outcomes of the removal. The biggest difference in opinion is reflected in the fact that all the public agencies agreed that dam removal is a necessary process, while the private sector does not always find it necessary. The lack of legislation is not seen as a challenge directly, it is more reflected in the permitting process and the lack of funding. All the responders agreed that the removals should be prioritized based on the highest potential hazard and the highest environmental outcomes (fish habitat and river restoration). One responder specified, however, that currently the removal is prioritized among the dams that already have been chosen for removal by the owners. All the answers indicated that the final decision should belong to the owners or to whoever pays for the removal. Three of the responders added that the community should be part of the decision making process too. To summarize, the general perception of the survey appears to be that dam removal is a good alternative for improving the fish habitat and river restoration. However, the decision process itself as well as which dams to remove are topics that require improvement.

<u>Conceptual Framework.</u> Based on the survey results summarized above, an improved decision making process is presented as a five-step conceptual framework. This conceptual framework is a statement of an ideal dam removal process, which would create ways in which the dam removal could become a more controlled, efficient and transparent process. The steps of this conceptual framework are as follows:

1. **Institute a dam removal regulation**: This would include a general rule in which a high hazard dam or dams in disrepair would not be allowed to remain in place. Owners who could not afford to remove the dam could transfer the property rights to the state. The state, through a well-trained entity, would become the only decision-maker. This way the owner transfers the liability, and does not have to pay for the removal. Furthermore, this entity could now prioritize the removals throughout risk assessment and/or comprehensive cost-benefit analysis.
2. **Create a streamlined mechanism for ownership transfer:** a straight-forward mechanism for the transfer of ownership of a dam to the state.
3. **Establish a single agency responsible for dam removal with responsibility for reviewing and permitting projects.** Currently the permitting process alone can take more than a year, because there are numerous agencies involved. Some of these agencies (local, state and federal) have overlapping authority and reviewers assess essentially the same factors, making for a time consuming and confusing process.
4. **Prioritize the dam removal based on potential hazard and/or the potential environmental and social benefits**. The dam removal decision should be based on studies which indicate what dam has the highest hazard in case of failure and/or the greatest potential environmental and social benefits. For instance, if the main goal of the removal is to restore fish habitat, it would be efficient to start with the most-downstream dam and proceed sequentially in an upstream direction.
5. **Comprehensive valuation of all the costs and benefits, including the environmental and social externalities as part of the dam removal decision**. Neglecting to include externalities and common goods in an economic analysis can result in a failure to efficiently allocate all the resources related to dam removal. Externalities arise when the impact of an economic activity on outsiders is not considered. For example, in the case of a dam removal, the sediments from behind the dam released by the removal could settle downstream. The long term impact that the sediment deposits could have on downstream areas is rarely included in the cost-benefit analysis of the dam removal. These costs and benefits are listed in Table 2.

Table 2: Conceptual framework for a complete economic assessment of dam removal (adapted from Kruse & Scholz, 2006 with permission)

Costs	**Benefits**
Immediate costs: Final design; Sediment disposal Staging of materials; Constructions on site; Feasibility Study; Disposal of waste material; Permits;	Market goods: These are not applicable as they would be transferred from another location; hence including them would result in double counting.
Loss of Direct Services: Hydro-power Recreational value (use of the lake for fishing, boating) Flood control Irrigation abilities Water supply	Non-market goods / Environmental Benefits: Improving fish habitat; Fish passage Returning the river to the free-flow conditions; Recreational opportunities (fishing, boating, hiking, etc) Environmental aesthetics; Reduce flooding upstream of the dam; Cultural values (if this is the case)
External (indirect) impacts: - wetland change; - change in wildlife from the lake; - loss of "lake view"; - temporary sediment transport Infrastructure Cultural	

Case Study: Cost-Benefit Analysis. The conceptual cost-benefit analysis was applied to the removal of two dams located within Ipswich River Watershed: the South Middleton Dam and the Ipswich Mills Dam. The South Middleton Dam is currently under consideration for removal, but it is not necessarily the most viable decision because it is upstream of the Ipswich Mills Dam. One of the biggest challenges in this analysis is the lack of data. Dam removal in a new process in Massachusetts; hence the availability of records pre or post removal is minimal. Another problem is the lack of literature. Although the studies concerning dam removal are increasing dramatically, studies including environmental valuations with respect to the dam removal are sparse. In this analysis we focus on identifying all the environmental benefits associated with the removal of the Ipswich Mills and South Middleton Dams. Not all of these benefits have been quantified as yet, but they must be included in the cost-benefit analysis and will be quantified in the future. These benefits include: improving the fish habitats, returning the river to the natural flow, aesthetics, recreation (boating, freshwater fishing, walking), reducing the upstream flooding, liability, and cultural values.

Using benefits transfer method, the benefits of returning to a free-flow river were quantified. The willingness to pay method (WTP) performed by Sanders, Walsh, and Loomis (1990) was used to quantify the benefits from returning the river to a natural flow regime in Colorado. This entailed a mail survey of WTP to preserve free-flowing rivers sent to Colorado households statewide. The mail survey had a 51% response rate of deliverable surveys. The annual WTP per household for option, existence and bequest value was $77 in 1983 dollars. In order to obtain the value per mile the total amount was divided by the length of river being valued (555 miles). This resulted in a WTP of $0.21 per mile in 1983 dollars. If we account for inflation by using the ratio median incomes, a value of $77 in 1983 would be equivalent $163.70 in 2009 in Colorado (this is the most recent year that had median income information) or $0.29 per mile. Transferring this value to Massachusetts using the ratio of median incomes results in 0.305 per mile in 2009. Removal of the South Middleton and Ipswich Mill Dams would open 56 miles and 17 miles of river, respectively. In Middleton there are 2305 households and in Ipswich there are 5290 households. Multiplying $0.305 per mile by the potential free flowing river length then by the number of households in each community results in a benefit of $39,369 for removing the South Middleton Dam and , and $27,429 for removing the Ipswich Mill dam.

Dam deconstruction and removal costs were obtained from a report entitled "A Preliminary Site Reconnaissance and Cost Estimates for Ipswich River, Ox Pasture Brook, and Skug River Dams" by Woodlot Alternatives, Inc. determined a total value of deconstructing and removing the dams. In addition to these costs, the cost of reconsolidation of the two bridges located downstream of Ipswich Mill dam need to be added because there is a concern that the removal of this dam would affect the structure of these bridges. In January 2010, the California Department of Transportation developed a general guideline for structure type selections and their costs[7]. They determine the "bridge cost" based on the bridge type and the span range. They estimate that for reinforced concrete slab bridges with the span range between 16 and 19 feet, the cost range is $90-200 dollars per square foot. The bridges downstream of the dams are concrete structures measuring approximately 40 feet long (personal observations, October 2010) by 12 feet wide (http://www.ckollars.org/ipswich.html). With these considerations a rough estimation of the cost of repairing the affected bridges would be somewhere between $43,000 and 96,000. There is also major concern about the impact that removing the Ipswich Mill dam would have on the structure of the EBSCO Building located just upstream the dam (Ipswich River Watershed Association, personal communication, April 2010). However, this cost could not be estimated. With these assumptions, the estimated cost of the removal of the Ipswich Mill dam to be within a range of $608,000- $661,000. The removal cost of the South Middleton dam was estimated to be $1,310,000 using a similar method. Table 3 is a summary of preliminary cost and benefit estimates and a list of environmental benefits that still need to be quantified.

The ultimate goal of this research is to provide a useful tool that identifies ecosystem service value associated with a multitude of indicators of human influence. For the research proposed herein, we focused our approach on dam removal, which is important to the restoration of aquatic and riparian ecosystems and fish passage in New England. The conceptual framework developed in this research will be available for decision makers, to address the complexities involved in ecosystem restoration, multi-objective decision making and the optimal allocation of limited state and local resources.

Table 3: Summary of the costs and benefits associated with the two dam removals

South Middleton Dam	**Ipswich Mills Dam**
Cost of Removal	**Cost of Removal**
$1,310,000	$608,000- $661,000
Benefits of Removal	**Benefits of Removal**
Returning to free –flow river: $39,369 **Other benefits to be valued.** Improve Fish Habitat Support from public and private sector	Returning to free –flow river: $27,429 **Other benefits to be valued.** Improve Fish Habitat Reduces flooding upstream of the dam Reduce risk (The lack of exclusionary fencing and ease of public access)

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Publications and Conference Presentations:

a. <u>Articles in Refereed Scientific Journals in preparation</u>

Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert, Does current policy help or hinder dam removal decision-making in Massachusetts: a survey of stakeholders, *J. Environmental Management*, anticipated submission, Jul 2011.

Oglavie, D.R., E. M. Douglas, D. G. Terkla and B. Lambert. A framework for valuing the environmental costs and benefits of dam removal, *Water Resources Manag*ement, anticipated submission Sep 2011.

O'Brion, K., E. Douglas and A. Christian, Biomonitoring the Eel River headwaters restoration project, Plymouth, Massachusetts, *Freshwater Biology*, anticipated submission, Jan 2012.

b. <u>Conference Presentations</u>

A conceptual framework for estimating the environmental costs and benefits of dam removal (oral), National Conference on Engineering and Ecohydrology for Fish Passage, Amherst, MA June 27-29, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (oral), National Conference on Engineering and Ecohydrology for Fish Passage, Amherst, MA June 27-29, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (poster), North American Benthological Society Annual Meeting 2011, Providence RI, May 22-26, 2011.

Freshwater fish and aquatic macroinvertebrate biomonitoring of the Eel River Headwaters Restoration sites in Plymouth, Massachusetts (oral), 67th Annual Northeast Fish and Wildlife Conference, Manchester, NH, April 17-19, 2011.

A conceptual framework for estimating the environmental costs and benefits of dam removal (oral), 67th Annual Northeast Fish and Wildlife Conference, Manchester, NH, April 17-19, 2011.

Student Support

Two graduate students were supported with this grant:

Kevin O'Brion, MS Env. Sci (summer 2010 support).

Doina Oglavie, MS Env. Sci (summer 2010 through spring 2011).

Notable Achievements and Awards

n/a

Acid Rain Monitoring Project

Basic Information

Title:	Acid Rain Monitoring Project
Project Number:	2010MA262B
Start Date:	3/1/2010
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	1st
Research Category:	Not Applicable
Focus Category:	Acid Deposition, Water Quality, Surface Water
Descriptors:	None
Principal Investigators:	Marie-Francoise Hatte

Publications

1. Finn, E., Hatte, M.F., 2010, Acid Rain Monitoring Project Brochure. University of Massachusetts Water Resources Research Center, University of Massachusetts, Amherst, MA. http://www.umass.edu/tei/wrrc/arm/ARM_Brochure.pdf
2. Hatte, M.F. and Finn, E. 2010, Acid Rain Monitoring Project FY10 End of Fiscal Year Report, University of Massachusetts Water Resources Research Center, University of Massachusetts, Amherst, MA. http://www.umass.edu/tei/wrrc/arm/ARM%20FY10%20Annual%20Report.pdf

Introduction
This report covers the period July 1, 2009 to June 30, 2010, the ninth year of Phase IV of the Acid Rain Monitoring Project. Phase I began in 1983 when about one thousand citizen volunteers were recruited to collect and help samples from nearly half the state's surface waters. In 1985, Phase II aimed to do the same for the rest of the streams and ponds in Massachusetts. The third phase spanned the years 1986-1993 and concentrated on a subsample of streams and ponds to document the effects of acid deposition to surface waters in the state. Over 800 sites were followed in Phase III, with 300 citizen volunteers collecting samples and doing pH and ANC analyses. In 2001, the project was resumed on a smaller scale: about 50 volunteers are involved to collect samples from approximately 150 sites, 26 of which are long-term sites with ion and color data dating back to Phase I. In Phase IV (2001-2003), 161 lakes were monitored for 3 years. Since fall 2003 (Phase V), the project has been monitoring 151 sites, mostly streams, except for the 26 long-terms sites which are predominantly lakes.

Goals
The goals of this project are to determine the overall trend of sensitivity to acidification in Massachusetts surface waters and whether the 1990 Clean Air Act Amendment has resulted in improved water quality.

Methods
Methods were mostly unchanged from previous years: Volunteer collectors are contacted a month before the collection to confirm participation. Clean sample bottles are sent to them in the mail or via UPS, along with sampling directions, a field sheet/chain of custody form, and directions to the sampling sites if necessary. Collectors visit their site(s) twice a year, in April and October, when they collect a surface water sample from the bank or wading a short distance into the water body. They collect upstream of their body after rinsing their sample bottle 3 times with sample lake or stream water. If collecting by a bridge, they collect upstream of the bridge unless safety and access do not allow it. They fill in their field data sheet with date, time, site code information, place their samples on ice in a cooler and deliver the samples to their local laboratory right away. They are instructed to collect their samples as close to the lab analysis time as possible. In a few cases, samples are collected the day prior to analysis because the lab is not open on traditional “ARM Sunday.” Previous studies by our research team has established that pH does not change significantly when the samples are refrigerated and stored in the dark.

Volunteer labs are sent any needed supplies (sulfuric acid titrating cartridge, electrode, buffers), 2 quality control (QC) samples, aliquot containers for long-term site samples, and a lab sheet one week to ten days before the collection. They analyze the first QC sample in the week prior to the collection and call in their results to the Statewide Coordinator. If QC results are not acceptable, the volunteer analyst discusses possible reasons with the Statewide Coordinator and the Lab Director and makes modifications until the QC sample gives acceptable results. On Collection day, volunteer labs analyze the second QC sample before and after the regular samples, and report the results on their lab sheet along with the regular samples. Analyses are done on their pH-meters with KCl-filled combination pH electrodes. Acid neutralizing capacity (ANC) is measured with a double end-point titration to pH 4.5 and 4.2. Most labs use a Hach digital titrator for the ANC determination, but some use traditional pipette titration equipment. Aliquots are taken from the 26 long-term sites to fill one 60mL bottle and one 50mL tube for later analysis of ions and color. These aliquots are kept refrigerated until pick-up from UMass staff.

Aliquots, empty bottles, and results are collected by ARM staff a day or two after the collection. The Cape Cod lab mails those in, with aliquot samples refrigerated in a cooler with dry ice.

The Statewide Coordinator reviews the QC results for all labs and flags data for any lab results that do not pass Data Quality Objectives (within 0.3 units for pH and within 3mg/L for ANC). pH and ANC data are entered by one ARM staff and proofed by another. Data are uploaded into the web-based database at http://umatei.tei.umass.edu/ColdFusionProjects/AcidRainMonitoring/ and posted on the ARM web page at http://www.umass.edu/tei/wrrc/arm/.

Aliquots for 26 long-term sites are analyzed for color on a spectrophotometer within one day; anions within one month on an Ion Chromatograph; and cations within 6 months (but usually 2 months) on an ICP at the Environmental Analysis Lab (EAL) on the UMass Amherst campus. The data is sent via MS Excel spreadsheet to the Statewide Coordinator who uploads it into the web-based database.

UMass Chemistry Department's Dr. Julian Tyson and his laboratory team of graduate students run the Environmental Analysis Lab (EAL) and provide the QC samples for pH and ANC to all of the volunteer labs. EAL also provides analysis for pH and ANC for selected sampling sites.

Accomplishments

1. Monitoring was completed for 23 and 25 of our long-term group of 26 lakes and streams for pH, ANC, color and ions for the October 25, 2009 and the April 11, 2010 collections, respectively. Analysis results are presented in Tables 6 and 7 (see Appendix).

2. An additional 127 statistically representative streams were sampled to measure statewide trends in acidification (pH and ANC only). Analysis results are presented in Table 8 (see Appendix).

3. The network of volunteers was maintained and kept well informed on the condition of Massachusetts surface waters so that they would be able to participate effectively in the public debate. This was accomplished by e-mail and telephone communication, as well as through updates via an internet list-serv.

   There were 11 volunteer labs across the state, in addition to the EAL at UMass Amherst, in charge of pH and ANC analyses (Table 4).

   Table 1: Volunteer Laboratories

Analyst Name	Affiliation	Town
Joseph Ciccotelli	Ipswich Water Treatment Dept	Ipswich
Alan Christian	UMass Boston Environmental Studies Program	Boston
Cathy Wilkins	Greenfield High School	Greenfield
Sherrie Sunter	MDC Quabbin Lab	Belchertown
Dave Bennett	Cushing Academy	Ashburnham
Holly Bailey	Cape Cod National Seashore	South Wellfleet
Robert Caron	Bristol Community College	Fall River
Bob Bentley	Analytical Balance Labs	Carver
David Doe	Biology Dept. Wilson Hall WSC	Westfield
Jim Bonofiglio	City of Worcester Water Lab	Holden
Carmen DeFillippo	Pepperell Waste Water Treatment Plant	Pepperell
Chengbei Li	University of Massachusetts Environmental Analysis Lab	Amherst

   Several volunteer collectors were also recruited to replace retiring or ill collectors. As in the past, our volunteers take their responsibilities very seriously and take great pride in doing the job in full, revisiting a site if necessary. Some of our volunteers have been with the project since 1982 and are now quite advanced in age but are extremely dedicated and their experience is valuable to the project.

   A total of 72 volunteers participated in this year's program, 49 of them participating in both collections. Sixty-three of the volunteers were collectors, 12 were lab analysts, and 3 were both.

4. The ARM web site and searchable database were maintained and updated, adding new data as it became available. pH, ANC, ions and color data were added to the web database via the uploading tool created in previous years. The database was evaluated for quality control and uploading errors were corrected. The web-based program was updated to include recent years.

5. The data collected was analyzed for trends in pH and ANC for 151 sites and for color and ions for 26 sites, using the JMP® Statistical Discovery Software (http://www.jmp.com/software/). Bivariate

analyses (scatter plots, regression, and correlation) were run on pH, ANC, each ion, and color separately, predicting concentration vs. time. We looked at the data set for all seasons and for April and October separately to see if trends were dependent on season. Standard t-tests were also run on the same groups of data, comparing the current 10 years of data (2001-2010) to the older 10 years of data (1983-1993). We should note that the historical data includes collections from all months of the year rather than just April and October which are the only months we sampled in the latest phase of the project. This explains why statistics for the whole set of data are sometimes somewhat different from the results shown in separating the data into two seasons ("April" vs "October").

6. The Acid Rain Monitoring Project Brochure 2010 was created, published, and distributed to all program volunteers, past and present. Additional copies were distributed to the volunteer laboratories upon request.

Data Analysis Results

pH and ANC

Bivariate analysis for pH and ANC

Table 2 displays the number of sites out of 151 that show a significant change over time for pH or ANC. If the difference was not statistically significant ($p>0.05$), the sites are tabulated in the 'No Change' (not significantly different) category.

Table 2: Bivariate analysis results for pH and ANC

	All seasons		**April**		**October**	
	pH	**ANC**	**pH**	**ANC**	**pH**	**ANC**
No Change	96	120	92	105	116	123
Increased	48	22	58	44	18	15
Decreased	7	9	1	2	17	13

Table 3 displays the results of the t-test analysis, showing how many sites have a significant change in the current period compared to historical data.

Table 3: Standard t-test results for pH and ANC

	All Seasons		**April**		**October**	
	pH	ANC	pH	ANC	pH	ANC
No Change	101	113	105	107	124	125
Increased	42	21	44	44	15	15
Decreased	8	17	2	0	12	11

Those results are also graphed in figures 1 and 2.

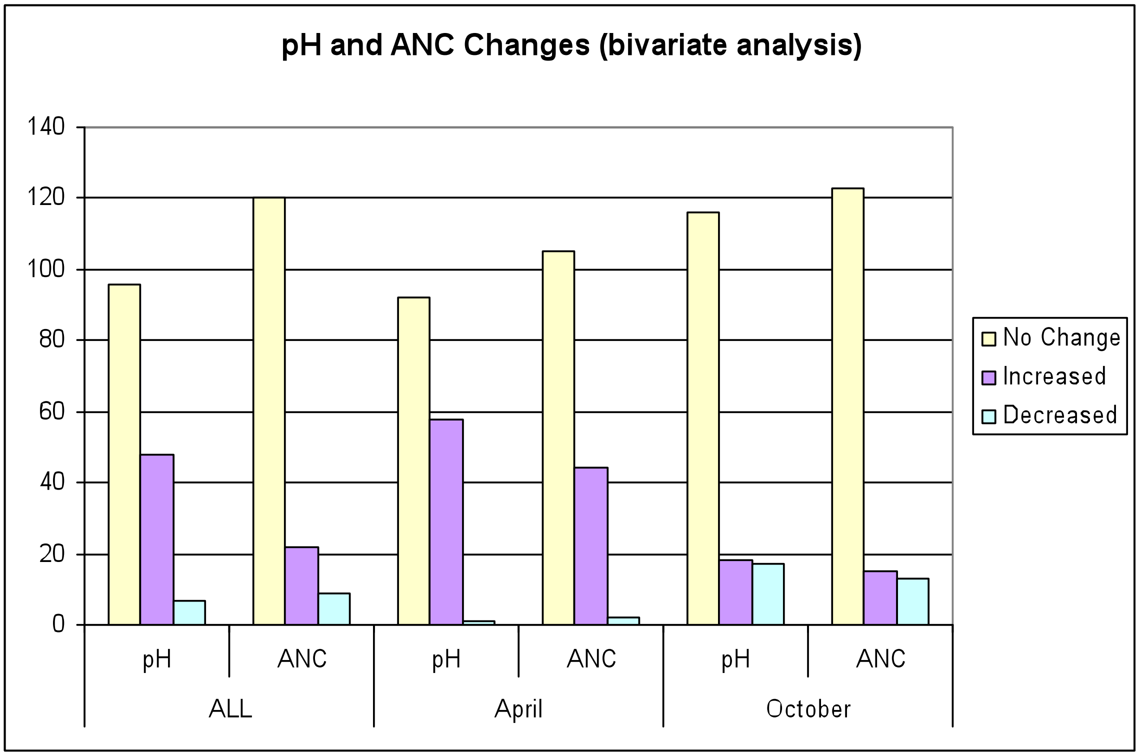


Figure 1. Changes in pH and ANC, from bivariate analysis

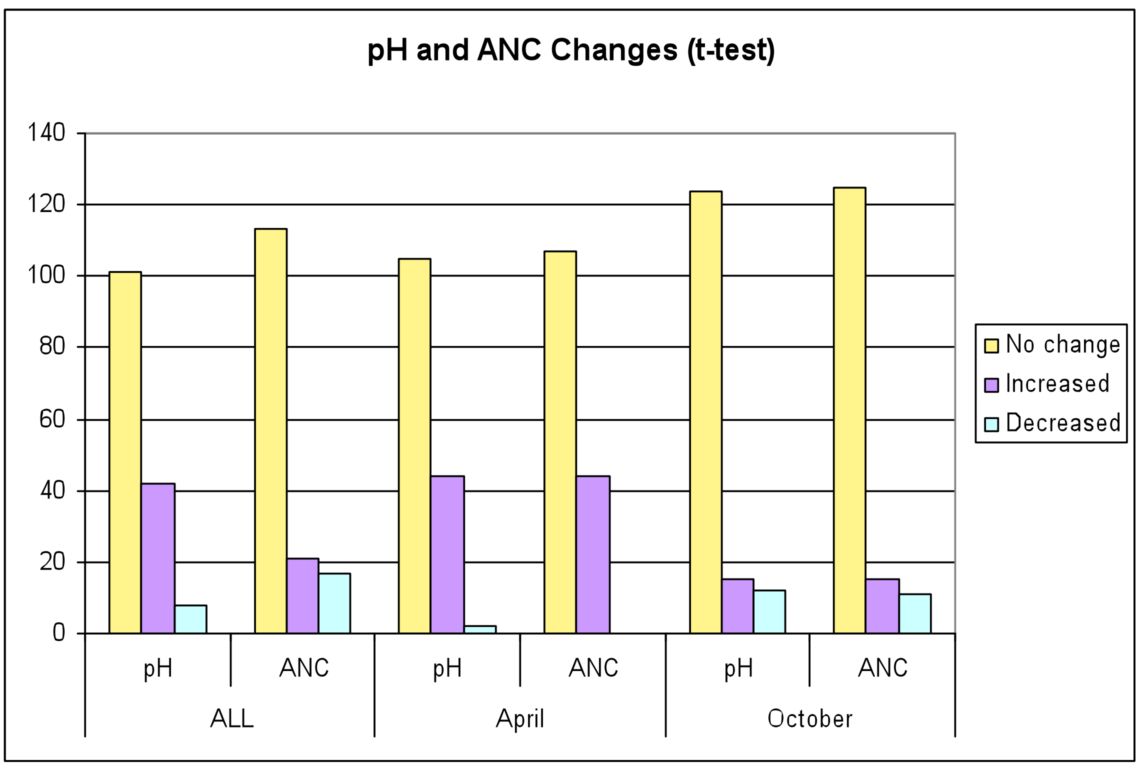


Figure 2. Changes in pH and ANC, from t-test analysis

While both types of statistical analysis give somewhat different results, they both show a similar tendency that for most sites, neither pH nor ANC has changed significantly over time. However, for those sites that show a significant change, more show an increase than a decrease in value. That is especially true for pH, with almost one third of the sites showing a statistically significant increase. ANC shows a less clear trend, except when spring and fall seasons are analyzed separately. In that case, many more sites show an increase in ANC in April than in October.

Ions and Color
Bivariate and standard t-test analyses were run on the 26 long-term sites that are analyzed for 10 ions and color. (In Phase V we analyze 11 ions, but Cu was not part of the cation suite in Phases I through III so no comparison can be made for that ion).

Table 4 and figure 3 show the results of the bivariate analysis for all parameters, while table 5 and figure 4 show the results of the standard t-test analysis.

Table 4: Bivariate analysis results for ions and color

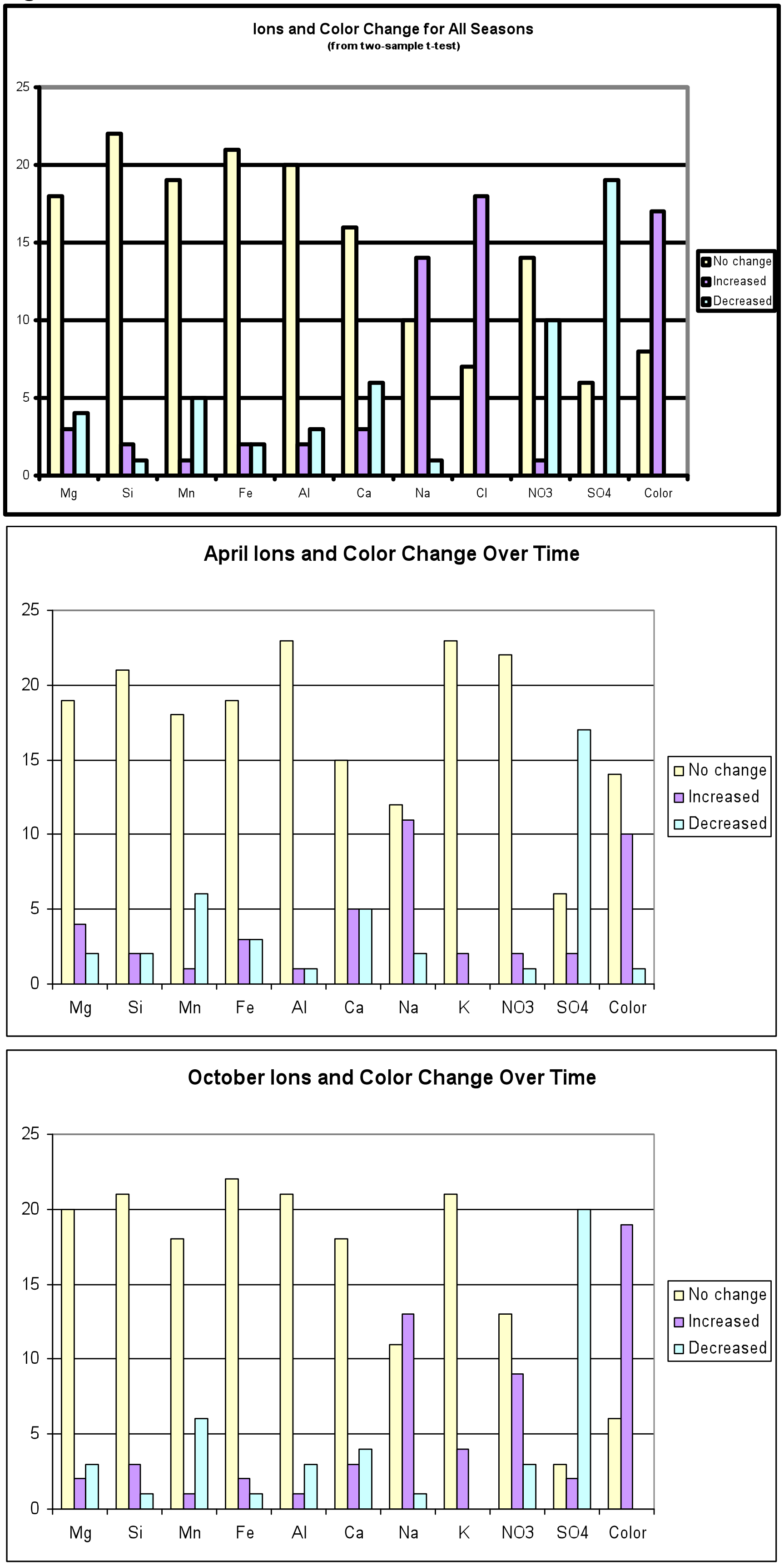
	April			October			All Seasons		
	No Change	Increased	Decreased	No Change	Increased	Decreased	No Change	Increased	Decreased
Mg	18	3	4	19	5	1	19	3	3
Si	22	0	3	21	3	1	23	1	1
Mn	20	0	5	19	1	5	19	1	5
Fe	19	1	5	19	3	3	18	4	3
Al	19	1	5	19	5	1	19	4	2
Ca	19	2	4	15	3	7	14	4	7
Na	11	13	1	12	9	3	12	12	1
K	22	3	0	21	4	0	22	3	0
Cl	10	15	0	8	17	0	6	19	0
NO3	15	9	1	19	5	1	14	10	1
SO4	3	0	22	5	0	20	8	0	17
Color	5	20	0	11	14	0	8	17	0

Table 5: T-test analysis results for ions and color

	April			October			All		
	No change	Increased	Decreased	No change	Increased	Decreased	No change	Increased	Decreased
Mg	19	4	2	20	2	3	18	3	4
Si	21	2	2	21	3	1	22	2	1
Mn	18	1	6	18	1	6	19	1	5
Fe	19	3	3	22	2	1	21	2	2
Al	23	1	1	21	1	3	20	2	3
Ca	15	5	5	18	3	4	16	3	6
Na	12	11	2	11	13	1	10	14	1
K	23	2	0	21	4	0	7	18	0
NO3	22	2	1	13	9	3	14	1	10
SO4	6	2	17	3	2	20	6	0	19
Color	14	10	1	6	19	0	8	17	0

Figure 3: Results of bivariate analysis for ions and color, all seasons, April, and October.

Figure 4: Results of standard t-test for ions and color, all seasons



Most cations show no significant change over time for the 26 sites we are following. A notable difference, however, can be seen for sodium, which increased for almost half of the sites no matter what season and with both types of statistics used.

All anions show significant changes as well. This change is seen more clearly with the bivariate analysis, which tracks concentrations continuously over time, while the t-test compares only the set of data from the first 10 years with the last 10 years of the project. Chloride never decreases with time, and increases for two-thirds of the sites. Nitrate's change is less definite, but it clearly increases for about a third of the sites and decreases for a couple of sites on average. Sulfate shows the most dramatic change, a strong decrease for over two-thirds of the sites.

Color also shows a consistent increase over time, for over two-thirds of the sites in all seasons.

Discussion
These results are mostly consistent with what we found for an earlier analysis we performed on lakes in Phase IV of this project, and with results from other research studies in the northeast. The main difference is that we now see more of an increase in pH, and we are seeing a clearer increase in nitrate in Phase V.

It is interesting to note that for both pH and ANC, more sites show an increase in April than in October, and this trend holds with both statistical tests performed. April is the time of year when we typically see the lowest pH and ANC values, most probably due to snowmelt waters that carry an important amount of nitrates into surface waters. Yet we do not see a clear corresponding trend with nitrate. On the contrary, the bivariate analysis shows more of an increase in nitrate in April than in October, though the reverse is true with the t-test analysis. An explanation might be that the reduction in sulfate is more important than the increase in nitrate, but further research or literature review should be done in order to draw a confident conclusion.

Base cations calcium and magnesium still show no sign of recovery, if anything calcium actually seems to be still declining.

Sulfate continues to show a strong and significant decline, in line with decreases in emissions of sulfur dioxide that followed the 1990 Clean Air Amendment. The increase in nitrate is also not surprising, despite a similar decrease in NOx emissions from power plants, because nitrogen emissions from vehicle sources have increased over time. Since roads in Massachusetts are often located along streams, and because roads are designed to channel water off and away from their surface, increased NOx emissions have a direct path into surface waters.

At this time we cannot confidently assess an increase or a decline in aluminum.

However, we continue to document a significant increase in sodium and especially chloride. This results very likely from road salting practices in the northeast.

The obvious increase in color is less intuitive to explain, but in New England, there is a buffer other than ANC that is rarely considered - organic acids. These natural acids make waters somewhat acidic and tea brown in color, and they act as buffers against further lowering of the pH by mineral acids. So naturally colored waters have been titrating acid deposition and becoming less colored. The increase in color we are observing would point to an increase in buffering capacity that is not measured by ANC.

In conclusion, to answer our question whether the 1990 Clean Air Act Amendment has resulted in improved water quality in Massachusetts surface waters, the answer is a cautious "somewhat." More water bodies seem to have improved than worsened, but the increase in nitrates, coupled with the lack of increase in calcium and magnesium cause concern that the improvement may not last and may even reverse in the future if NOx emissions are not curbed.

It is our recommendation to pursue this long-term monitoring of surface waters in the Commonwealth. We propose to change our sampling scheme to drop half of the streams we are currently following, and replacing them with an equal number of lakes that were monitored in Phase IV.

Acknowledgements

Thank you to all of the project's volunteers who make this project possible by collecting samples all over the state under any weather conditions, and who spend many hours in the lab analyzing samples.

Appendix

Table 6: October 2009 ARM Color and Ion Data

Name	Palsite	Color	Cl	NO3_N	SO4	Mg	Si	Mn	Fe2	Cu	Al	Ca	Na	K
Shingle Island Brook	188	686	12.649	0	2.919	1.07	5.87	0.01	1.01	0.09	0.68	2.18	7.06	1.90
Belmont Reservoir	21010	196	1.397	0	3.968	1.14	2.22	0.0025	0.01	0.15	0.33	3.27	0.60	1.29
Cobble Mt. Reservoir	32018	45	16.353	0.009	3.537	1.12	2.12	0.0025	0.01	0.11	0.20	2.73	9.58	1.11
Hawley Reservoir	34031	827	8.213	0	4.625	0.61	4.77	0.0025	0.17	0.10	0.41	2.30	4.81	1.58
Wyola Dam	34103	NS	NS	NS	NS	0.46	1.04	0.0025	0.07	0.09	0.21	1.65	4.71	1.02
Upper Naukeag Lake	35090	16	12.996	0	3.154	0.37	0.12	0.0025	0.04	0.09	0.31	0.89	8.41	0.94
Crystal Lake	36043	49	1.29	0	1.592	0.27	0.01	0.0025	0.01	0.10	0.22	0.38	0.26	1.34
Lake Lorraine	36084	4	31.121	0	4.029	0.80	0.01	0.0025	0.01	0.09	0.19	3.66	19.28	1.69
Quabbin Station	36129	21	8.186	0	4.432	0.59	0.48	0.0025	0.01	0.09	0.24	2.06	4.78	1.05
Nipmuck Pond	42039	69	19.793	0	4.668	0.52	2.53	0.0025	0.01	0.09	0.36	1.83	11.63	0.88
N. Watuppa Lake	61004	270	18.856	0	2.953	0.75	2.55	0.0025	0.32	0.09	0.40	1.79	10.60	1.11
Ashby Reservoir	81001	95	19.329	0	3.888	0.75	1.78	0.0025	0.49	0.09	0.31	2.35	12.24	1.68
Wright Pond	81160	167	10.187	0	2.646	0.49	1.05	0.0025	1.19	0.09	0.35	1.51	6.44	1.35
Whitehall Reservoir	82120	57	22.634	0.007	3.946	1.00	0.01	0.0025	0.01	0.09	0.27	2.73	12.58	1.28
Hedges Pond	94065	33	11.628	0	4.152	1.09	0.01	0.0025	0.01	0.09	0.23	0.70	6.78	1.17
College Pond	95030	35	6.244	0	3.632	0.73	0.08	0.0025	0.01	0.11	0.13	0.92	3.82	1.01
Ezekiel Pond	95051	42	26.018	0.015	4.958	1.21	0.01	0.0025	0.01	0.10	0.22	1.76	14.93	1.62
Little Sandy Pond	95092	305	19.625	0	3.247	1.01	0.98	0.0025	0.22	0.10	0.62	1.28	11.16	1.93
Great Pond	96117	7	28.617	0	6.661	1.97	0.01	0.0025	0.01	0.10	0.20	1.24	15.67	1.51
Kinnacum Pond	96163	115	17.759	0	3.287	1.15	0.01	0.0025	0.01	0.09	0.23	0.30	8.95	1.27
Caldwell Creek	3626575	99	7.541	0	4.456	0.61	4.45	0.0025	0.10	0.12	0.37	1.82	4.14	1.30
W. Branch Swift River	3626800	279	13.419	0.007	3.586	0.79	2.91	0.0025	0.37	0.09	0.32	2.76	7.42	1.97
E. Branch Swift River	3627200	438	7.1	0	3.543	0.69	5.41	0.0025	0.61	0.09	0.66	1.28	4.01	1.40
Rattlesnake Brook	6235125	577	14.09	0	2.869	1.14	5.43	0.0025	0.49	0.20	0.75	1.78	7.20	1.83
Angeline Brook	9560000	753	39.981	0	3.897	1.59	5.50	0.0025	0.94	0.14	0.60	4.34	23.23	3.40
Bread & Cheese Brook	9560150	686	12.649	0	2.919	1.07	5.87	0.01	1.01	0.09	0.68	2.18	7.06	1.90

NS = No Sample

Table 7: April 2010 ARM Color and Ion Data

Name	Palsite	Color	Cl	NO3_N	SO4	Mg	Si	Mn	Fe2	Cu	Al	Ca	Na	K
Shingle Island Brook	188	385.00	11.49	0.06	5.60	0.86	2.35	0.09	0.52	0.01	0.23	2.32	6.99	1.36
Belmont Reservoir	21010													
Cobble Mt. Reservoir	32018	75.50	14.74	0.06	3.72	0.90	2.69	0.02	0.11	0.01	0.02	2.28	8.87	0.58
Hawley Reservoir	34031	42.00	10.49	0.03	5.54	0.52	4.45	0.03	0.08	0.01	0.08	2.27	6.32	0.49
Wyola Dam	34103		6.40	0.01	4.38	0.36	2.46	0.02	0.05	0.00	0.06	1.55	4.29	0.48
Upper Naukeag Lake	35090	46.50	14.09	0.01	3.11	0.27	0.83	0.01	0.04	0.01	0.06	0.87	9.07	0.37
Crystal Lake	36043	30.00	1.14	0.01	2.30	0.23	0.02	0.03	0.05	0.01	0.05	0.61	0.59	0.48
Lake Lorraine	36084	7.50	33.85	0.03	4.40	0.72	0.02	0.01	0.03	0.01	0.01	3.70	21.28	1.10
Quabbin Station	36129													
Nipmuck Pond	42039	27.00	16.72	0.01	5.13	0.36	2.72	0.02	0.03	0.01	0.18	1.61	10.45	0.39
N. Watuppa Lake	61004													
Ashby Reservoir	81001	86.00	12.62	0.01	4.00	0.50	1.42	0.03	0.27	0.02	0.01	1.81	8.71	0.69
Wright Pond	81160	101.50	7.64	0.01	2.87	0.31	1.23	0.03	0.33	0.00	0.10	1.14	5.28	0.59
Whitehall Reservoir	82120	56.00	19.42	0.01	4.69	0.78	0.36	0.01	0.08	0.01	< DL	2.71	11.56	0.81
Hedges Pond	94065	26.00	12.22	0.01	4.21	0.97	0.28	0.00	0.01	0.01	0.01	0.83	6.87	0.55
College Pond	95030	30.00	6.21	0.01	3.53	0.67	0.47	0.01	0.03	0.03	< DL	0.85	4.01	0.46
Ezekiel Pond	95051	44.00	24.20	0.01	4.51	1.06	0.11	0.00	0.05	0.01	0.01	1.78	13.91	0.87
Little Sandy Pond	95092	47.50	22.16	0.07	3.78	0.93	0.05	0.01	0.04	0.01	0.01	1.29	12.47	1.27
Great Pond	96117	11.00	27.39	0.01	6.62	1.83	0.05	0.03	0.03	0.01	< DL	0.97	15.10	0.79
Kinnacum Pond	96163	89.00	18.39	0.01	2.93	1.21	0.06	0.03	0.04	0.01	0.06	0.50	10.10	0.75
Caldwell Creek	3626575	38.00	6.85	0.01	5.30	0.47	4.15	0.02	0.04	0.01	0.09	1.64	4.67	0.26
W. Branch Swift River	3626800	101.00	11.24	0.05	4.68	0.55	2.32	0.05	0.26	0.01	0.07	2.27	7.20	0.83
E. Branch Swift River	3627200	203.50	6.76	0.01	5.49	0.43	3.26	0.03	0.21	0.01	0.17	1.05	4.77	0.49
Rattlesnake Brook	6235125	358.00	10.06	0.03	3.52	0.93	1.82	0.01	0.31	0.02	0.32	1.88	6.03	1.16
Angeline Brook	9560000	428.50	38.73	0.42	5.33	1.32	3.27	0.04	0.50	0.05	0.24	3.88	23.27	1.61
Bread & Cheese Brook	9560150	385.00	11.49	0.06	5.60	0.86	2.35	0.09	0.52	0.01	0.23	2.32	6.99	1.36

NS = No Sample
< DL = Below Detection Limit

Table 8: pH and ANC, all sampling sites.

PALSITE	NAME	October 2009		April 2010	
		PH	ALK	PH	ALK
5131425	Aldrich Brook	6.22	5.2	6.31	4.22
9560000	Angeline Brook	4.57	-2.8	5.48	0.7
2105425	Anthony Brook	6.70	5.9	6.76	5.1
81001	Ashby Reservoir	6.55	4.2	6.64	3.2
3626700	Atherton Brook	5.24	0.3	6.1	0.7
3107625	Babcock Brook	6.99	9.2	6.16	4.3
3417750	Bagg Brook	NS	NS	8.1	79.5
3523925	Bailey Brook	7.85	-0.3	6.08	3.7
3524050	Baker Brook	5.43	5.4	5.64	1.6
8146000	Bartlett Pond Brook	5.71	1.5	5.61	0.5
2105350	Barton Brook	7.18	22.2	7.72	44.9
6236100	Bassett Brook	5.51	1.3	6.05	3.4
371.0001	Beagle Club Pond	5.91	7.1	6.6	4.2
3523825	Beaman Brook	6.11	1.8	5.75	0.9
3627475	Beaver Brook	6.90	10.44	6.84	10.1
6235800	Beaver Brook	6.11	8.8	6.62	10.7
9458025	Beaver Dam Brook	5.90	5.3	6.35	8.3
6236250	Beaverdam Brook	4.46	-3	4.71	0
21010	Belmont Reservoir	6.79	7.6	6.98	12.8
3107375	Benton Brook	6.22	4.8	6.41	6.6
2105750	Bilodeau Brook	7.02	20.3	7.16	22.5
8144075	Bixby Brook	6.45	6.35	6.55	4.1
3522675	Black Brook	6.47	3.1	6.19	1.9
6237625	Black Brook	5.43	2.1	6.03	0.2
9253700	Black Brook		20.9	6.78	15.1
6134700	Blossom Brook	4.45	-1.9	4.52	-1.8
3524375	Bluefield Brook	4.87	-0.2	4.67	-1.5
9253925	Boston Brook		19.5	6.91	16.6
3523400	Boyce Brook	5.69	0.8	5.99	0.6
3315325	Bozrah Brook	7.45	17.4	7.16	13.1
9560150	Bread And Cheese Brook	6.04	4.5	6.13	3
9153000	Bull Brook		16.6	6.59	12
5233750	Bungay River	NS	NS	6.72	15.6
3628075	Burnshirt River	NS	NS	5.98	7.4
3627850	Burrow Brook	NS	NS	6.15	1.28
3626575	Cadwell Creek	5.69	0.65	5.99	8.6
2105725	Cady Brook	6.82	10.2	6.8	14
5334150	Clear Run Brook	5.69	3.5	7.58	75.9
7239175	Clematis Brook	NS	NS	6.7	30.2
5132550	Coal Mine Brook	7.59	36.5	NS	NS
32018	Cobble Mountain Reservoir	6.77	5.5	6.58	3.8
6134550	Cole River	6.69	7.1	6.72	11.3
95030	College Pond	6.31	1.2	6.47	2.6

7240050	Cress Brook		32.1	6.44	13.5
5132625	Cronin Brook	6.42	9	6.89	8.1
36043	Crystal Lake	5.45	0.2	5.61	4.2
6235925	Dam Lot Brook	5.48	1.5	6.24	4.6
3419600	Dean Brook	4.99	-0.5	6.05	0.8
7240225	Dix Brook		14.1	6.07	14
5132700	Dorothy Brook	7.01	30.5	NS	NS
2103800	Dry Brook	7.15	46.1	7.54	110.5
3627200	East Branch Swift River	6.53	4.26	6.46	28
3314925	East Oxbow Brook	7.02	5.6	6.76	5.5
3420100	Esther Brook	7.34	31.6	7.44	20.2
95051	Ezekiel Pond	6.14	2.3	6.47	0.6
6235375	Fall Brook	5.49	2	5.72	1.7
3627500	Flat Brook	6.41	9.51	6.51	80.4
3106825	Fox Brook	6.72	6.3	6.22	2
4230075	French River	6.94	13.5	6.43	7.98
7240375	Godfrey Brook	6.93	33.5	NS	NS
96117	Great Pond	5.20	-0.4	5.105	-0.2
8143775	Greens Brook	6.5	50.9	7.22	36.2
3420000	Ground Brook	7.23	24.1	7.67	30.3
8143675	Gulf Brook	6.94	30.2	7.13	9.35
3210425	Hamilton Brook	6.56	4.8	6.51	3
3315075	Hartwell Brook	7.48	23.2	7.54	21.3
9661525	Hatches Creek	6.21	9.2	6.16	6.6
34031	Hawley Reservoir	5.91	1.89	6.02	11.8
94065	Hedges Pond	5.81	1.4	5.99	1.2
3313175	Hinsdale Brook	7.71	52.4	7.92	52.7
3627000	Hop Brook	6.39	3.2	6.7	4.4
9253500	Ipswich River		24.5	6.85	19.4
8143925	James Brook	6.74	42.8	7.03	50.8
3523750	Kenny Brook	6.00	1.7	6.11	1.3
6134500	Kickamuit River	5.80	4.6	6.49	6
3421725	Kidder Brook	6.63	3.3	6.67	2.5
2105700	Kilburn Brook	6.78	5.2	6.94	7.9
9253625	Kimball Brook		18.6	7	19.8
6134725	King Phillip Brook	4.36	-2.4	4.5	-1.8
96163	Kinnacum Pond	4.95	-0.5	4.82	-0.8
3314450	Kinsman Brook	7.12	11.6	7.26	12.5
36084	Lake Lorraine	6.94	9	6.88	7.7
34103	Lake Wyola	6.62	3.2	6.15	1.2
5131775	Laurel Brook	6.61	6.8	5.91	3.1
3208725	Little River		0	6.79	4.7
95092	Little Sandy Pond	6.23	1.6	6.08	2.4
3316550	Lord Brook	6.71	2.9	7.08	4.6
3524075	Mahoney Brook	5.87	2.8	5.6	0.5
8451825	Martins Pond Brook	6.79	40.1	6.98	30.25
3626475	Maynard Brook	NS	NS	5.72	1.27

8144725	McGovern Brook	7.20	15.9	7.12	9.9
2105100	Mill Brook	7.46	29.3	7.37	20.6
3419825	Mill River	7.50	54.6	7.5	37.8
7240075	Miller Brook		36	6.7	14.7
8247475	Millham Brook	6.98	22.5	NS	NS
8144825	Monoosnuc Brook	5.56	18.2	6.81	6.7
3107075	Moody Brook	6.19	8.4	6.03	4.5
3627050	Moosehorn Brook	6.41	2.7	6.46	1.9
6235775	Mulberry Meadow	5.90	6.2	6.65	7
42039	Nipmuck Pond	5.85	1	5.66	0.57
7239550	Noanet Brook		15	6.22	7.8
3314100	North River	7.02	9.5	7.49	23.7
61004	North Watuppa Lake	5.28	-0.2	NS	NS
5131350	Ohio Brook	5.11	0.4	5.33	0.88
3107575	Pond Brook	6.34	2.9	6.66	3.1
6235825	Poquanticut	5.98	4.3	6.33	3.8
7239525	Powissett Brook		10	5.93	2.5
36129	Quabbin Station 202	6.66	3.01	NS	NS
6235125	Rattlesnake Brook	4.48	-2.1	4.64	-1.8
3209275	Ripley Brook	6.09	1.8	5.59	1
3524250	Robbins Brook	5.46	-0.3	5.37	0.7
8143825	Robinson Brook	6.73	12.8	7.48	22
5334100	Rocky Run		0	7.5	74.1
5131275	Round Meadow Brook	6.26	4.3	6.16	3
3524175	Scott Brook	5.37	0.4	5.91	1.5
5133125	Scott Brook	NS	NS	NS	NS
5132600	Sewall Brook	7.17	23	NS	NS
3313850	Shingle Brook	7.78	14.3	7.87	44.9
188.0001	Shingle Island Brook	4.87	-0.5	5.35	0.8
5132750	Singletary Brook	7.21	11.2	7.28	10.4
2104200	Sleepy Hollow Brook	7.72	96	8.05	158.4
6235750	Snake River	6.18	10.2	6.1	4.8
2103725	Soda Creek	7.31	34.6	7.65	44.1
3313650	South River	7.39	39.2	7.66	34.7
3524275	Spud Brook	5.59	1.5	5.7	0.5
3313375	Stafford Brook	7.43	67.8	7.87	54.8
5132850	Stone Brook	6.48	13.1	7.19	20.5
7239925	Stop River		40	6.32	13.4
3625975	Sucker Brook	6.40	6.4	6.43	5.48
6235150	Terry Brook	4.95	-0.8	5	-0.2
3316050	Todd Brook	6.34	0.5	7	2.7
5334075	Torrey Creek	NS	NS	6.33	11.4
3524200	Towne Brook	5.08	0.8	5.62	0.2
8144250	Trapfall Brook	7.02	5.55	6.95	3.8
3314650	Underwood Brook	6.84	5.3	7.23	6.3
35090	Upper Naukeag Lake	6.04	0.9	5.51	0.3
3107700	Valley Brook	6.61	9.7	6.41	2.6

3314550	Vincent Brook	7.3	14.5	7.39	14.4
5133150	Wadsworth Brook	5.37	3.5	NS	NS
3210300	Walker Brook	6.94	9.8	6.94	10.8
4230325	Wellington Brook	6.41	25.5	6.61	19.4
3628175	West Branch Ware River	6.49	3	6.26	1.9
9558900	Weweantic River	5.62	2.1	6.01	3.6
8248425	Whitehall Brook		16	6.19	8.6
82120	Whitehall Reservoir		10.2	6.21	1.7
8145075	Whitman River	6.57	5.7	6.32	2.3
3523950	Wilder Brook	5.52	1.6	5.47	1
8144175	Willard Brook	6.91	35.3	6.85	3.45
2104100	Williams River	7.83	145.4	8	128.2
81160	Wright Pond	6.26	1.7	6.17	1.5

NS = No Sample

USGS Award No. G10AP00091 Evaluation of Adaptive Management of Lake Superior amid Climate Variability and Change

Basic Information

Title:	USGS Award No. G10AP00091 Evaluation of Adaptive Management of Lake Superior amid Climate Variability and Change
Project Number:	2010MA284S
Start Date:	4/30/2010
End Date:	3/31/2012
Funding Source:	Supplemental
Congressional District:	
Research Category:	Climate and Hydrologic Processes
Focus Category:	Management and Planning, Climatological Processes, Surface Water
Descriptors:	
Principal Investigators:	Casey Brown, Casey Brown

Publications

1. Brown, C., Werick, W., Fay, D., and Leger, W. (2011) "A Decision Analytic Approach to Managing Climate Risks - Application to the Upper Great Lakes" Journal of the American Water Resources Association (in press).
2. Brown, C., Moody, P., and Werick, W. (2010) Abstract H23M-04, Decision Scaling to Aid the development of a dynamic regulation plan for the Upper Great Lakes, presented at 2010 Fall Meeting, AGU, San Francisco, Calif., 13-17 Dec.

Background

The International Upper Great Lakes Study (IUGLS) began in 2007, as the International Joint Commission (IJC) established an independent study board composed of U.S. and Canadian members to review the operation of structures controlling Lake Superior outflows and to evaluate improvements to the operating rules and criteria governing the system. The Board is expected to publish recommendations in spring 2012 for near term changes to the regulation plan of Lake Superior, the largest managed freshwater body in the world (Clites and Quinn, 2003). The regulation of Lake Superior affects lake level, navigation and hydroelectricity production on Lakes Michigan, Huron and Erie, comprising an immense water resources system.

As a result of the considerable uncertainty associated with future climate and lake levels, as well as other sources of uncertainty such as ecosystem responses and the state of the navigation industry, a process of selecting the optimal plan based on a most probable future scenario was rejected. Instead, a bottom-up process for identifying vulnerabilities and assessing risk from climate change has been adopted

Lake levels in the Upper Great Lakes levels exhibit a significant degree of natural variability in the historical record (Clites and Quinn, 2003). This variability has caused considerable challenges in the design of regulation plans for Lake Superior in the past with changes being implemented several times in the 20th century. Given the lack of success in designing regulation plans that were robust to natural variability in the past and the additional uncertainty associated with climate change in the future, a change to the traditional regulation plan design is warranted. Underlying the process is the premise that we are limited in our ability to anticipate the future and therefore any recommended plan must perform well over a very broad range of possible futures. Of additional concern are surprises, low probability events that could have very large impacts. While incorporating unknown surprises into a regulation plan appeared infeasible, a strategy for managing their occurrence was prioritized. Finally, it was recognized that the identification of vulnerabilities must be led by those who understand the specific aspects of the lakes best, the stakeholders.

With these considerations, the Lake Superior regulation strategy incorporates an investigation of regulations plans that are robust to a wide range of climate conditions, definition of impacts in terms of lake levels by stakeholders, and an adaptive management process to manage uncertainty and to facilitate adaptation to changing climate, and other unanticipated changes. The uncertainty is due not only to changing climate but also due to the relatively poorly understood lake dynamics in response to changing conditions, the uncertainties associated with the definition of coping zones (described below) and other performance metrics and also the possibility of changing objectives for lake management.

The strategy is consists of three primary parts:

> 1) Identification of vulnerabilities by stakeholders and definition of acceptable and unacceptable lake levels for each impact area, called coping zones
>
> 2) The development and assessment of a dynamic regulation plan for Lake Superior that could incorporate short range and long ranges forecasts

3) Assessment of plausible climate risk for the evaluation of regulation plans and to assess climate risk beyond what the regulation of Lake Superior outflows is able to manage

In recognitions of the short timeframe for the study and Study Board guidance regarding the extent to which adaptive management will be investigated, our recent work has focused on the identification and estimation of plausible risk posed by climate change.

The work in this report reflects an effort that is conducted in collaboration with a variety of U.S. and Canadian partners. The UMass Hydrosystems Research Group is engaged in various aspects of these tasks. This report is an attempt to describe our efforts in that regard but does not report on the considerable effort being made by others. The effort reported here is the result of collaboration of many partners and it is not intended to be interpreted as entirely our effort, although the results reported here have all been produced by our group.

Part 1. Vulnerability Identification and Definition of Coping Zones

In order to prioritize concerns for the regulation of Lake Superior, stakeholder experts were tasked with identifying the vulnerabilities of the system to climate changes and other changing conditions. Termed technical working groups, stakeholders and technical experts convened in the following impact areas: ecosystems, hydropower, commercial shipping, municipal and industrial water and wastewater systems, coastal systems and recreational boating and tourism. A primary challenge was the quantification of vulnerabilities in commensurate units. To address this issue, our stakeholder groups represented by the Technical Working Groups (TWGs) were asked to define vulnerabilities in terms of lake levels, including the duration of the lake level. Lake levels were defined in three categories called "coping zones": A (acceptable), B (significant negative impacts, but survivable) and C (intolerable without policy changes). The TWGs are defining what combination of lake level and duration lead to the kind of impacts consistent with the coping zone descriptions. The definition of coping zones allows the evaluation of regulation plan performance to be conducted in terms that are comparable across impact sector and defined by the stakeholders.

At present we have evaluated the coping zones reported by all the Technical Working Groups. Table 1 shows the most conservative range of zones from those reported by Coastal TWG. The low coping zone C occurred 7 times during the 1918 – 2008 period, which seems frequent for a condition deemed intolerable. Figure 1 shows graphically the levels for each lake that define coping zones B and C. The Coastal TWG zones continue to be the most conservative. The occurrence rate for coping zones under historic and the 50,000 year

Table 1. Coastal TWG coping zones and the number of occurrences based on monthly Lake Superior levels 1918 – 2008. Note that Coast TWG coping zones vary by season on the low levels.

Coping Ranges for Superior			Count
Upper C	183.95	183.95	0
Upper B	183.56	183.56	244
Lower B	183.06	182.89	10
Lower C	182.99	182.79	7
Valid Months	Jun-Nov	Dec-May	

stochastic NBS time series are shown in Figures 2 through 5. Figures for Michigan-Huron, Erie and Lake St. Clair are in the Appendix. The results indicate that low C occurrences are more frequent than high C, largely as a result of a more conservative definition on the low side.

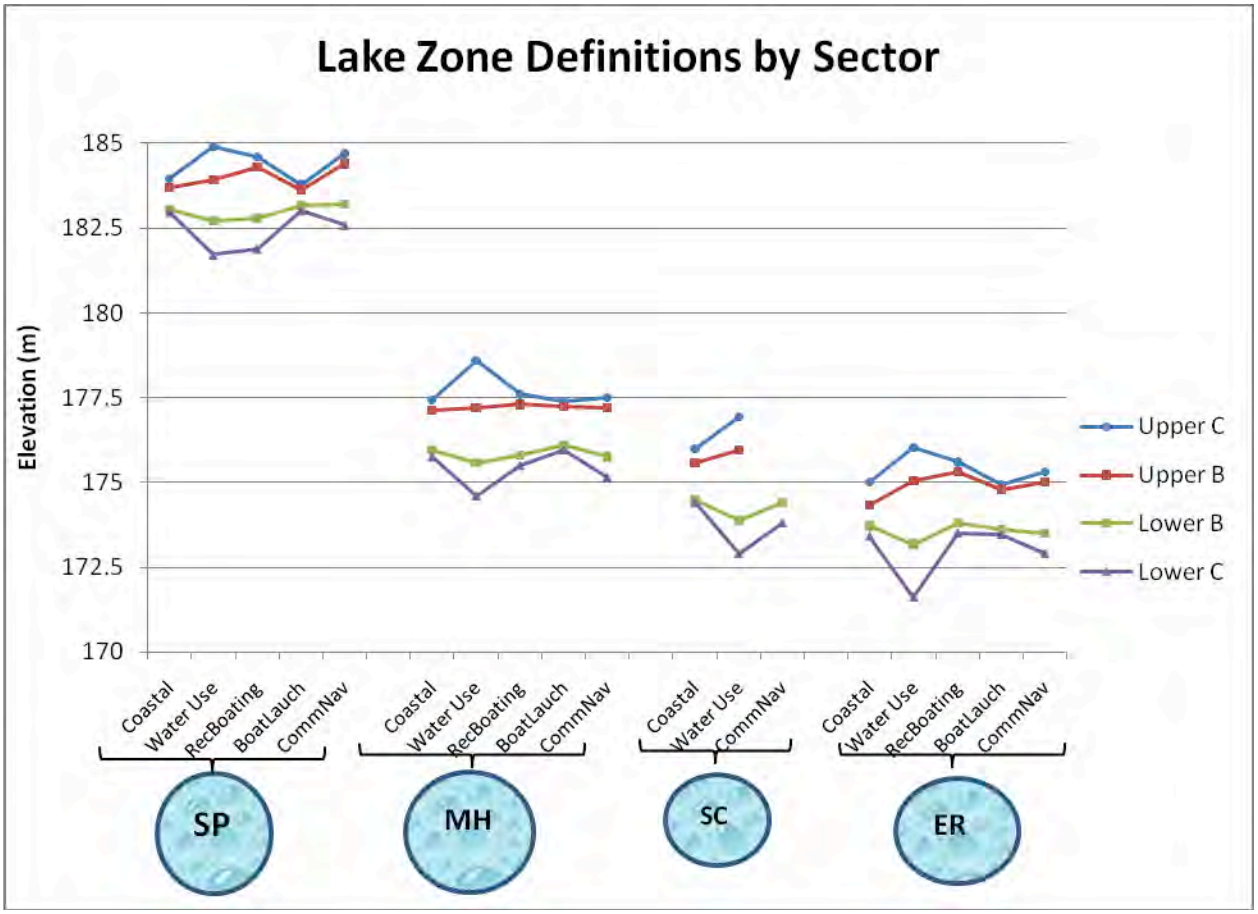


Figure 1: Zone Definitions by Lake and Sector

In a technical sense, the definition of the coping zones constitutes the definition of the "hazard," that is, the negative impacts that result from, say, a climate change. It does not take into account the probability of those impacts. Risk is defined as the product of the hazard and the probability of that hazard occurring. Our approach and progress in estimating risk is described in Part 3.

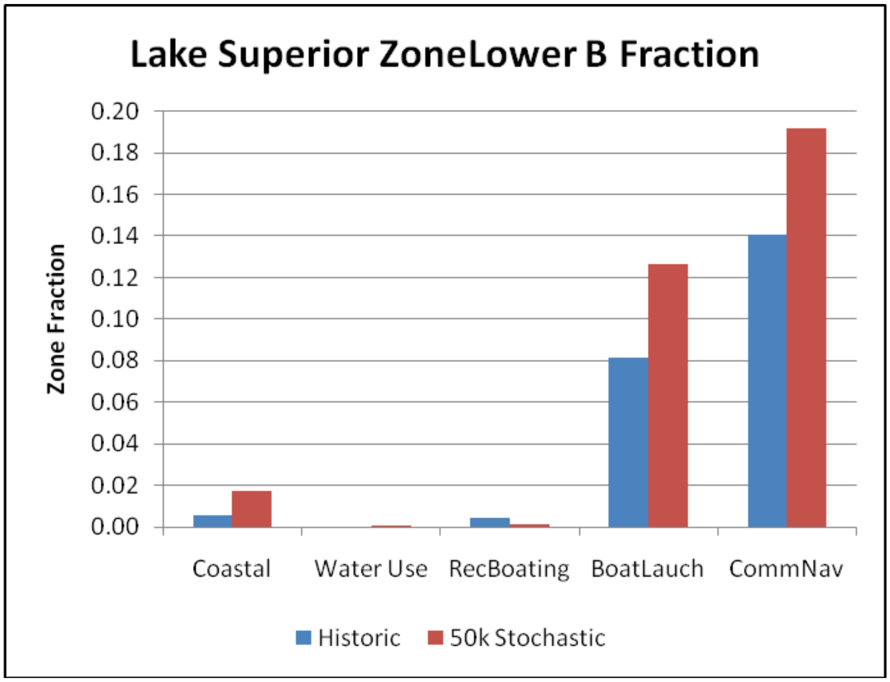


Figure 2. Lower Zone B Occurrences on Lake Superior

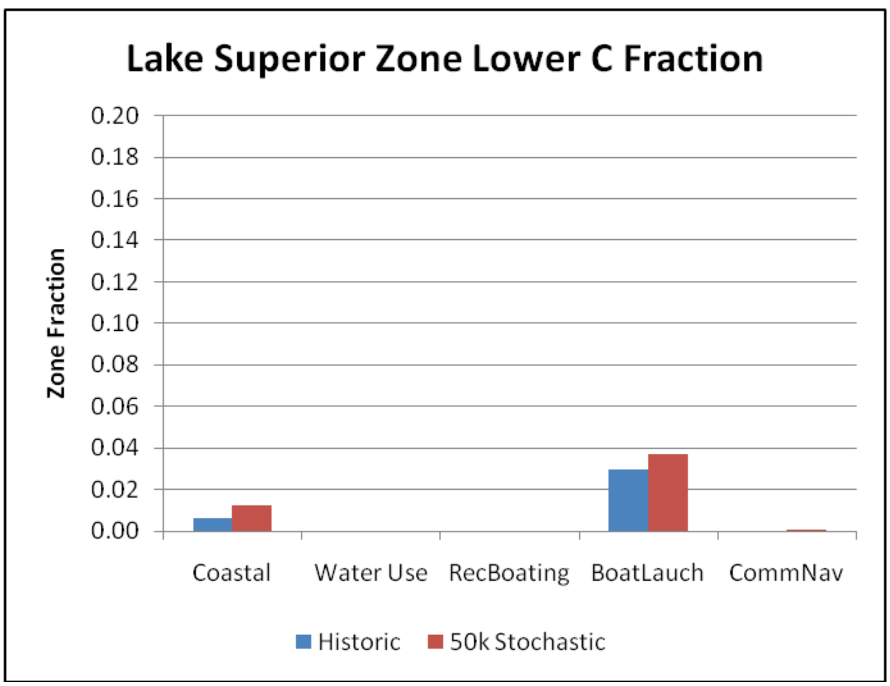


Figure 3. Lower Zone C Occurrences on Lake Superior

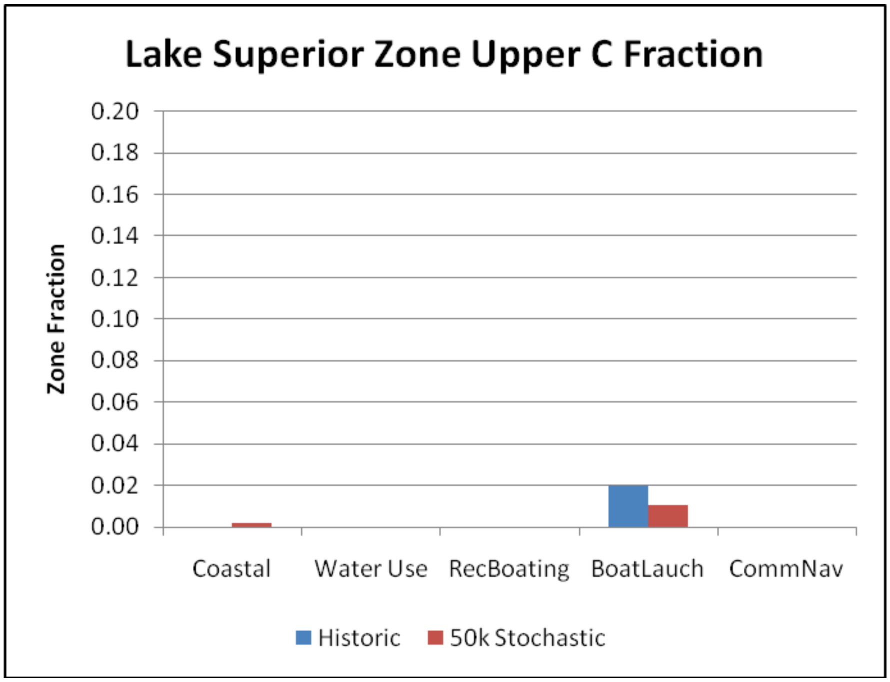


Figure 4. Upper Zone C Occurrences on Lake Superior

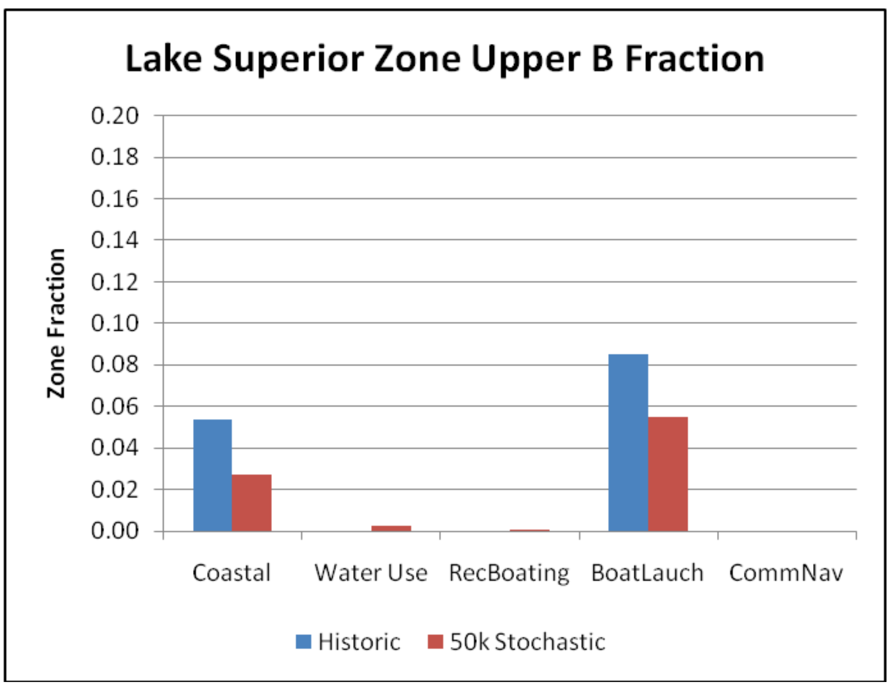


Figure 5. Upper Zone B Occurrences on Lake Superior

Part 2. Development and Assessment of a Lake Superior Regulation plan

Traditional water resources planning often focuses on formulating an optimal design based on performance evaluated with a best estimate of future hydrologic conditions. This is an apt description for previous approaches to developing regulation plans for Lake Superior (see Clites and Quinn, 2003). Regulation plans for Lake Superior were modified approximately seven times during the regulated period of 1914 to the present. Changes to the regulation plans resulted not only due to hydrologic conditions but also due to evolving societal priorities for regulation, including increasing interest in hydroelectricity production and new emphasis on including impacts on downstream lakes.

The relatively frequent rate of adjustment of past regulations plans reinforced the emphasis of this study on robustness, defined as a regulation strategy that could perform acceptably over a wide range of climate conditions. In order to increase the range over which a regulation plan can perform acceptable, the concept of a dynamic regulation plan has been proposed. The theory is that a dynamic plan would have rules that vary based on the prevailing climate conditions and possibly a forecast of future climate conditions. Our conceptualization of a dynamic regulation plan has two aspects: 1) a short term change in releases based on a short term forecast (< 1 year ahead) and 2) a major change in regulation strategy based on evidence of a major climate shift. Because new regulation strategies are still in development, most are very similar to the current 77A plan, and there has been development of a potentially skillful forecast of Lake Superior NBS, we have focused efforts on the first aspect of dynamic regulation.

Specifically, an alteration to plan 77A has been developed and evaluated based on a decision rule that relies on logic related to lake level and a forecast of Lake Superior NBS developed by Vincent Fortin and others at Environment Canada. The decision rule, simply stated, adjusts the plan 77A release either higher (or lower) when the Lake Superior level is in the upper tercile (or lower tercile) and the forecast is for above average NBS (or below average NBS). The decision rule is shown in Table 2 along with the results of the analysis.

In an initial analysis, the release was adjusted by 5 to 10% in accordance with the decision rule and the resulting lake levels were assessed using Werick's Excel version of the 77A and the lakes routing model. These results showed that for certain years, a correct forecast could lower high lake levels by 3 – 6 cm and increase low lake levels by 2 – 5 cm on Lake Superior while having minimal impact on downstream lakes. Because the Werick Excel version of 77A does not react dynamically to changing lake levels, only single year results are obtained. These are shown in the figures B1 – B4 in the Appendix.

The decision rule used for evaluation of the forecasts is purposely designed to be conservative, only using forecasts when a "wrong forecast" will have little impact. That is, it is presumed that when at a high tercile the impact of increasing releases will be small even if the forecast for high NBS does not come true. To test this hypothesis we explicitly evaluated lake levels in the actual years the Fortin forecast was not correct. The results are shown in Figure B5 and B6.

The results proved promising enough to warrant evaluation with a full 77A and lakes routing model, which the UMass Hydrosystems Research Group has recently developed in Matlab. A comparison of the Matlab model results with the existing Fortan model results are shown in Figure B7. The results are very positive with a correlation of 0.9987. The model will also be useful in the evaluation of plausible risk of plan 77A, modified 77A, and new regulation plans as they are developed. Using Matlab 77A, the dynamic response of 77A to the cumulative effects of the modified releases could be evaluated. The results showed that the use of perfect forecasts has a compressing effect on Lake Superior levels and minimal effect on Lakes Michigan, Huron and Erie. The distribution of Lake Superior levels with and without the perfect forecast is shown in Figure B8. More detailed results, including the effects of operation forecasts are available but have not been included here in the interest of conciseness.

Table 2. Decision Rule for modified 77A releases from Lake Superior based on lake level and NBS forecast. The table shows the changes in the releases and also the effects of those changes based on our evaluation. Of particular interest are the effects of "big misses" and false alarms, as discussed in the text. The results show that these do not have large impacts due to the conservative decision rule.

Lake Level (Tercile)	Forecast (Tercile)	Proposed AM Action	Observed NBS (Tercile)	Comments	Impact
High	High	Increase Outflow by X%	High	Correct Forecast and Action	Reduce Highs by 3 - 6 cm (5-10% increase)
		Increase Outflow by X%	Medium	False Alarm	Reduce Highs by 3 - 6 cm (5-10% increase)
		Increase Outflow by X%	Low	False Alarm	Reduce Highs by 3 - 6 cm (5-10% increase)
	Medium	No Action	High	Miss	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Miss	
	Low	No Action	High	Big Miss	Highs 3-6 cm higher than with AM
		No Action	Medium	Correct Action	
		No Action	Low	Correct Forecast and Action	
Medium	High	No Action	High	Correct Forecast and Action	
		No Action	Medium	Correct Action	
		No Action	Low	Correct Action	
	Medium	No Action	High	Correct Action	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Correct Action	
	Low	No Action	High	Correct Action	
		No Action	Medium	Correct Action	
		No Action	Low	Correct Forecast and Action	
Low	High	No Action	High	Correct Forecast and Action	
		No Action	Medium	Correct Action	
		No Action	Low	Big Miss	Lows 2-5 cm lower than with AM
	Medium	No Action	High	Miss	
		No Action	Medium	Correct Forecast and Action	
		No Action	Low	Miss	
	Low	Decrease Outflow by X%	High	False Alarm	Increase lows by 2-5 cm (5-10% decrease)
		Decrease Outflow by X%	Medium	False Alarm	Increase lows by 2-5 cm (5-10% decrease)
		Decrease Outflow by X%	Low	Correct Forecast and Action	Increase lows by 2-5 cm (5-10% decrease)

Next Steps: The use of forecasts with plan 77A shows some promise for improving the range of acceptable performance of the plan and may also aid other candidate regulation plans. Our next analyses would focus on the use of a mock forecast developed by Vincent Fortin and applied to long records, stochastic NBS series and GCM based-series. To improve plan performance, the decision rule could be experimented with in terms of the % alteration of release and the use of lake level in the decision rule. At present, the focus of our work is on climate risk estimation. However, if time and interest permit, we could pursue investigation of the incorporation of forecasts into Lake regulation.

Part 3. Assessment of Plausible Climate Risk

The assessment of plausible climate risks consists of three steps. The first is the identification of climate hazards, which is conducted without regard to how plausible a hazard may be. The identification of hazards is described in Part 1. The second step consists of relating climate hazards to the climate conditions that cause them. We do so by "data mining" the 50K stochastic NBS time series and summarize the findings through the development of the climate response function. This is described below in section 3.1. The final step is the estimation of plausible climate risk, which is completed by using the climate response function to estimate the impacts of climate conditions derived from a variety of sources of climate information, including output from GCM and stochastic series, including paleodata-based. This step is described below in section 3.2.

3.1 Development of the Climate Response Function

Climate conditions on the Great Lakes can be summarized in terms of Net Basin Supplies (NBS), where NBS is the sum of precipitation, runoff, releases, inflows and diversions and evaporation (negative). The identification of risks first depends on the description of risk. In this process, risks are quantified in terms of coping zones as described above. The coping zones are used to quantify risk level by counting occurrences of coping zones B and C. Next, in order to relate coping zones to the estimation of plausible climate risk, the coping zones occurrences must be related to the climate conditions that cause them. The identification of climate risks for a particular regulation plan is conducted by summarizing the climate conditions in terms of the statistics of NBS and tracking the occurrence of zone B and C occurrences for each climate scenario. Note that because plausibility must be assessed in terms of climate conditions, i.e., the long term statistics of weather, the risks must be quantified in terms of climate conditions (e.g., 30 year mean annual NBS instead of a particular annual value of NBS).

Figures 6 through 9 display the relationship between climate conditions (in terms of NBS statistics) and the occurrences of coping zone levels on Lakes Superior and Michigan-Huron, as indicated by the symbols and their size. These figures are generated by "data mining" the 50K stochastic time series. Climate conditions for 30 year windows are calculated and the zone occurrences tracked. The figures show indicate that zone occurrences are much more common under conditions of reductions in mean NBS and increases in variability. High zone levels occur only infrequently and under fairly extreme conditions.

Relationships between the explanatory climate conditions and the number of occurrences are then developed. The results indicate that no single statistic completely explains the occurrence of the zones. However, the mean, standard deviation and serial correlation of NBS over the 30 year climate window have significant influence on them. These results form the basis for the climate response function.

The development of the climate response function is the key step that will relate climate hazards to their plausibility in order to estimate risk. The function is used to predict the number of zone occurrences that will occur under a given set of climate conditions or climate change. It is designed to be able to directly link the climate change projections to the information encapsulated in the climate response function. A major benefit of this approach is that since the climate response function was developed on 50,000 years of NBS, the results yield a much more robust estimate of risk than would be attained by simply running the 30 year GCM time series through the lake routing model.

The underlying hypothesis is that statistics of climate variables can explain or predict zone occurrences on the Upper Great Lakes. Three climate statistics based on annual NBS values were analyzed. During the study, 100 year, 50 year, and 30 year windows were considered. 30 year analysis windows were used because it allows direct comparison with GCM models output. The three climate statistics are the normalized mean annual NBS, the normalized standard deviation of the annual NBS and the serial correlation of the annual NBS. The mean annual NBS and the standard deviation of the annual NBS were normalized based on the historic NBS series mean and standard deviation.

$$\overline{NBS} = \frac{1}{n}\sum_{i=1}^{n} NBS_i \tag{1}$$

$$X_1 = \overline{NBS}^* = \frac{\overline{NBS} - \overline{NBS}_{hist}}{\overline{NBS}_{hist}} \tag{2}$$

$$s_{NBS} = \left(\frac{1}{n-1}\sum_{i=1}^{n}\left(NBS_i - \overline{NBS}\right)^2\right)^{\frac{1}{2}} \tag{3}$$

$$X_2 = s_{NBS}^* = \frac{s_{NBS} - s_{NBS,hist}}{s_{NBS,hist}} \tag{4}$$

$$X_3 = r = \frac{1}{n-1}\sum_{i=1}^{n-1}\frac{\left(NBS_i - \overline{NBS}\right)^2\left(NBS_{i+1} - \overline{NBS}\right)^2}{S_{NBS}^2} \tag{5}$$

The relationship between the percent change in average annual NBS, the standard deviation of the annual NBS, and the relative occurrence of zone C is shown in Figure 2 for Lake Superior and in Figure 3 for Lakes Michigan-Huron. Figure 4 shows the relationship between percent change in NBS, serial correlation and zone occurrences for Lakes Superior and Figure 5 for Michigan-Huron. The number of zone occurrences is proportional to the size of the marker.

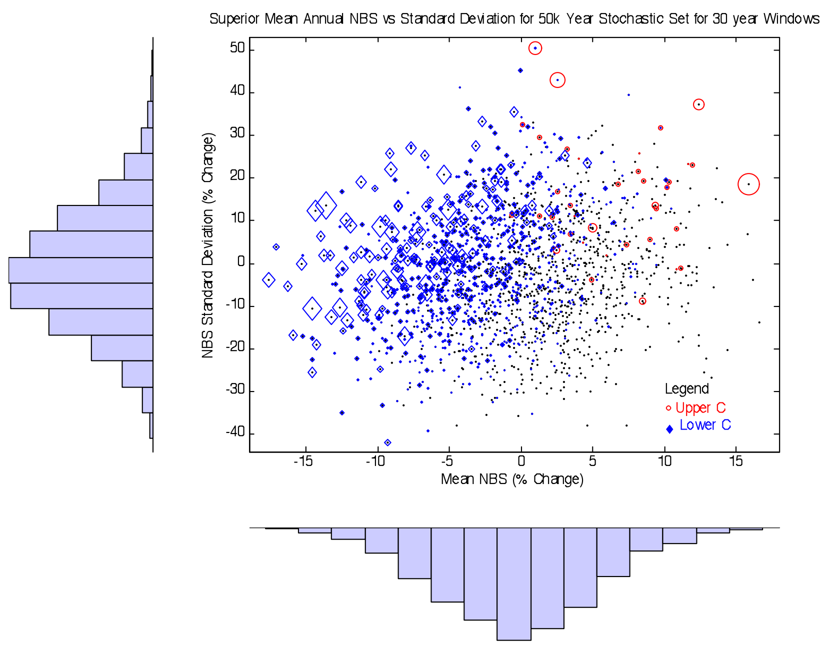


Figure 6: Lake Superior Mean NBS versus NBS Standard Deviation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

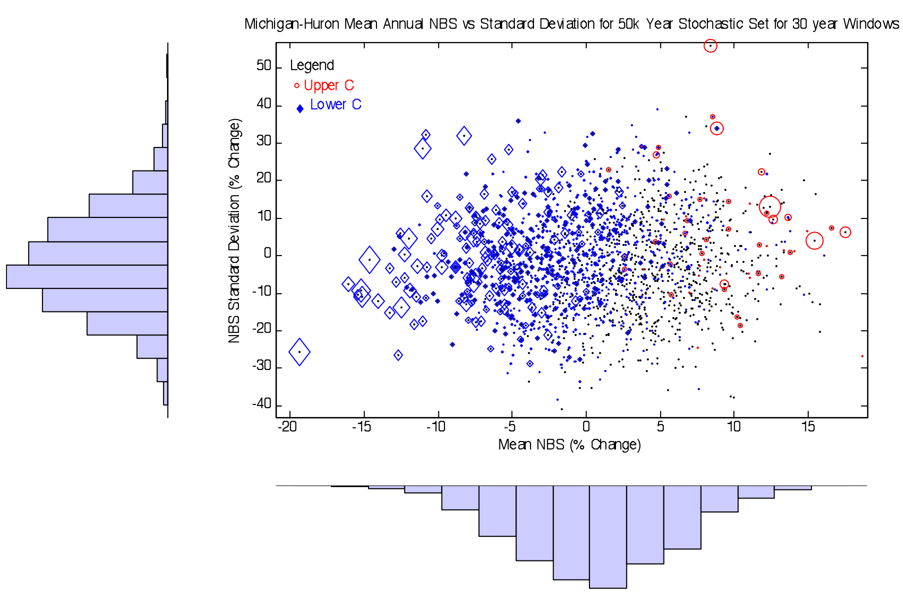


Figure 7: Lakes Michigan-Huron Mean NBS versus NBS Standard Deviation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

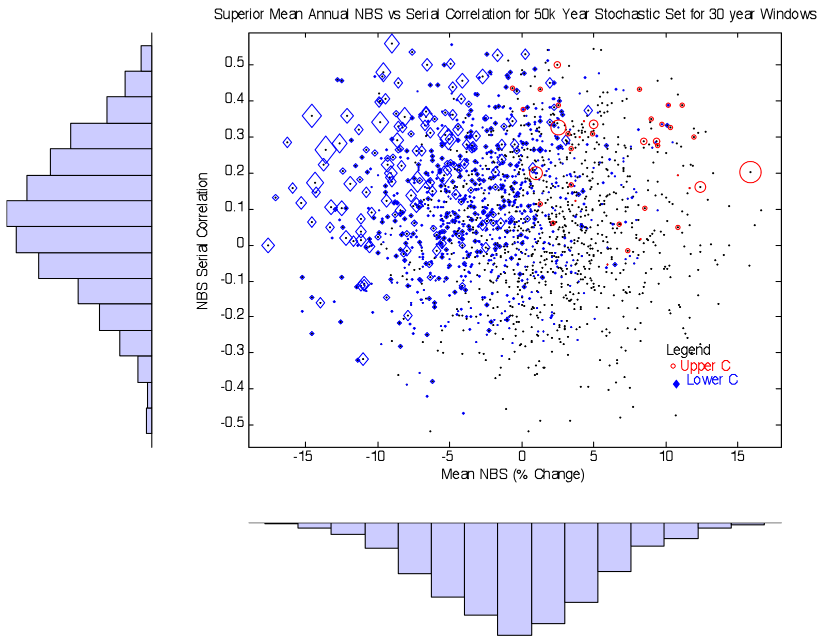


Figure 8: Lake Superior Mean NBS versus NBS Serial Correlation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

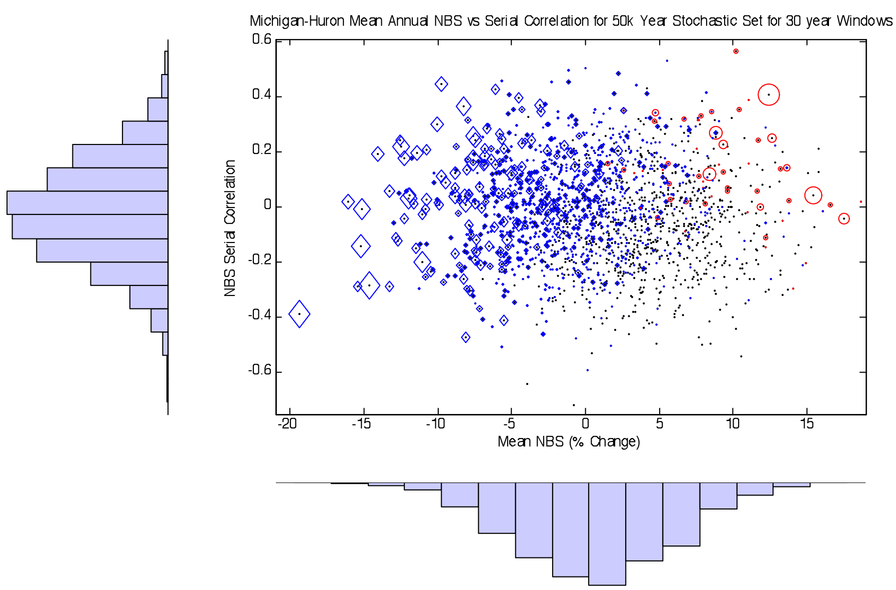


Figure 9: Lakes Michigan-Huron Mean NBS versus NBS Serial Correlation with Upper and Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set.

The figures demonstrate the relation between the climate variables and the zone occurrences. This led to the formulation and testing of statistical models of the relationship between the climate statistics and the zone occurrences. The number of zone occurrences can be treated as fractional data or as non-negative discrete data. This analysis treats the number of zone occurrences in the Lower C, Lower B, Upper B, and Upper C as independent. The occurrences are not, strictly speaking, independent and the total number of zone occurrences in Lower C, Lower B, A, Upper B and Upper C is equal to the number of months in the analysis period. In this statistical analysis, the model's ability to estimate the number of zone occurrences per zone was assessed. This model did not constrain the total number of predicted months.

Based on data screening, an exponential model was selected for the deterministic function. The data screening demonstrated a correlation between all three predictor climate statistics and the zones, with X_1 = percent change in mean NBS showing the strongest correlation. The candidate exponential models in Equations 6a through 6d were considered.

$$E(Y) = A\ exp(B\ X_1) \tag{6a}$$

$$E(Y) = A\ exp(B\ X_1 + C\ X_2) \tag{6b}$$

$$E(Y) = A\ exp(B\ X_1 + D\ X_3) \tag{6c}$$

$$E(Y) = A\ exp(B\ X_1 + C\ X_2 + D\ X_3) \tag{6d}$$

where Y is the expected number of Lower C, Lower B, Upper B or Upper C zone occurrences. The Maximum Likelihood Estimate method was used to determine the model parameters. Unlike the ordinary least squares method, this method requires an explicit assumption of the stochastic error component of the model. There are several stochastic error models appropriate for a non-negative, integer predictand. The two primary models that considered were the Poisson distribution and the negative binomial distribution. The Poisson distribution has a single parameter, λ, and has the following properties:

$$P_P(y = Y) = \frac{e^{-\lambda}\lambda^y}{y!} \tag{7a}$$

$$E[P_p] = \lambda \tag{7b}$$

$$s_{P_p} = \lambda \tag{7c}$$

The parameterization for the Poisson model allows us to substitute the deterministic model component in for λ. Substituting equation 6a into 7a yields:

$$Y \propto \frac{e^{A\ exp(B\ X_1)}\left(A\ exp(B\ X_1)\right)^y}{y!} \tag{8}$$

The advantages of using a Poisson stochastic model include that it is bounded to non-negative integers and that it does not add additional model parameters. The model also allows for a standard deviation that changes with the expected value.

The negative binomial model has two parameterizations. The most common is the mechanistic parameterization where p represents the per trial probability of success and n represents the number of successes awaited. In this parameterization, y is the number of failures until n successes. The alternative parameterization is μ which is the mean and k which is an overdispersion parameter. In this case, y does not have a mechanistic interpretation.

$$P_{NB}(y = Y) = \frac{\Gamma(\mathbf{k}+\mathbf{y})}{\Gamma(\mathbf{k})\mathbf{y}!}\left(\frac{\mathbf{k}}{\mathbf{k}+\boldsymbol{\mu}}\right)^{\mathbf{k}}\left(\frac{\boldsymbol{\mu}}{\mathbf{k}+\boldsymbol{\mu}}\right)^{\mathbf{y}} \tag{9a}$$

$$E[P_{NB}] = \mu \tag{9b}$$

$$S_{P_{NB}} = \mu + \frac{\mu^{2}}{k} \tag{9c}$$

The advantage of the negative binomial model is that the parameter k can capture the greater standard deviation, or dispersion, of the predictand relative to the Poisson model.

The Maximum Likelihood Estimate method was used to estimate parameters. This method seeks the parameters that provide the greatest likelihood, given the data. For a data point, y_i, the probability of y_i given the input variables $\boldsymbol{X}_i$ and the parameters $\boldsymbol{C}$ can be expresses as the likelihood of the data and is given in Equation 10. Note that $\boldsymbol{X}$ and $\boldsymbol{C}$ can be vectors from X_1 to X_n and C_1 to C_m.

$$P(y_i \mid \boldsymbol{X}_i, \boldsymbol{C}) = L\{data_i \mid \boldsymbol{X}_i, \boldsymbol{C}\} \tag{10}$$

Assuming that the data points are independent, or that each 30 year sequence is independent of the next, then the likelihood of the entire data series is the product the likelihood of each data point, as shown in Equation 11.

$$L\{data \mid \boldsymbol{X}, \boldsymbol{C}\} = \prod_{i=1}^{N} \left(P(y_i \mid \boldsymbol{X}_i, \boldsymbol{C})\right) \tag{11}$$

The product of small numbers gets small very rapidly, so it is common to take sum logarithm of the likelihoods. The maximum value of log likelihood occurs at the MLE estimates of the parameters, $\boldsymbol{C}$. Since many search algorithms are written to find minimum values, it is normal to search for the parameters, $\boldsymbol{C}$, that produce the smallest negative log likelihood, NLL.

$$NLL\{data \mid \boldsymbol{X}, \boldsymbol{C}\} = \sum_{i=1}^{N} \left(P(y_i \mid \boldsymbol{X}_i, \boldsymbol{C})\right) \tag{12}$$

The absolute value of the NLL does not have a probabilistic or physical meaning, but the relative value does convey information about the relative support for the model given the data. Competing models can be compared using their NLL values based on the best fit parameterization using information criteria. The most common of these is the Akaike Information Criteria (AIC), which is calculated using Equation 13. The principle of parsimony implies that simple models are better. Nearly any model can have an improved fit with more parameters or input variables, but the AIC penalizes the more complex model based on the number of parameters used, m.

$$AIC = 2\, NLL + 2\, m \tag{13}$$

The relative AIC value is then used to distinguish between competing models. Models with a Δ AIC < 2 are indistinguishable; with 4 < Δ AIC < 7 they are distinguishable; and models with Δ AIC > 10 are definitely different.

Table 3: Model comparison for five competing models to predict Lake Superior Lower C zone occurrences using a 30 year analysis window over the 50k stochastic data set.

Deterministic Model	Stochastic Model	NLL	Parameters	AIC	Δ AIC
A exp(B X_1)	Poisson	7,050	2	14,104	6,554
A exp(B X_1 + C X_2)	Poisson	6,304	3	12,614	5,064
A exp(B X_1 + D X_3)	Poisson	6,379	3	12,764	5,214
A exp(B X_1 + C X_2 + D X_3)	Poisson	5,858	4	11,724	4,174
A exp(B X_1 + C X_2 + D X_3)	Negative Binomial	3,770	5	7,550	0

As shown in Table 1, the negative binomial stochastic model with an exponential model including three predictor variables has the lowest NLL value and the lowest AIC value. The Δ AIC values indicate that this model has significantly more support, given the data. Similar analysis of Lower B, Upper B and Upper C zones, all show that the exponential model with a negative binomial stochastic distribution is the consistent best fit model. Figures 10 through 13 show the expected number of zone occurrences with a 95% confidence interval for Lower C, Lower B, Upper B and Upper C on Lake Superior. Figures 14 through 17 show the same for Lakes Michigan-Huron. In each case, the only predictor variable shown is X_1, the percentage change in mean NBS, since it has the highest correlation.

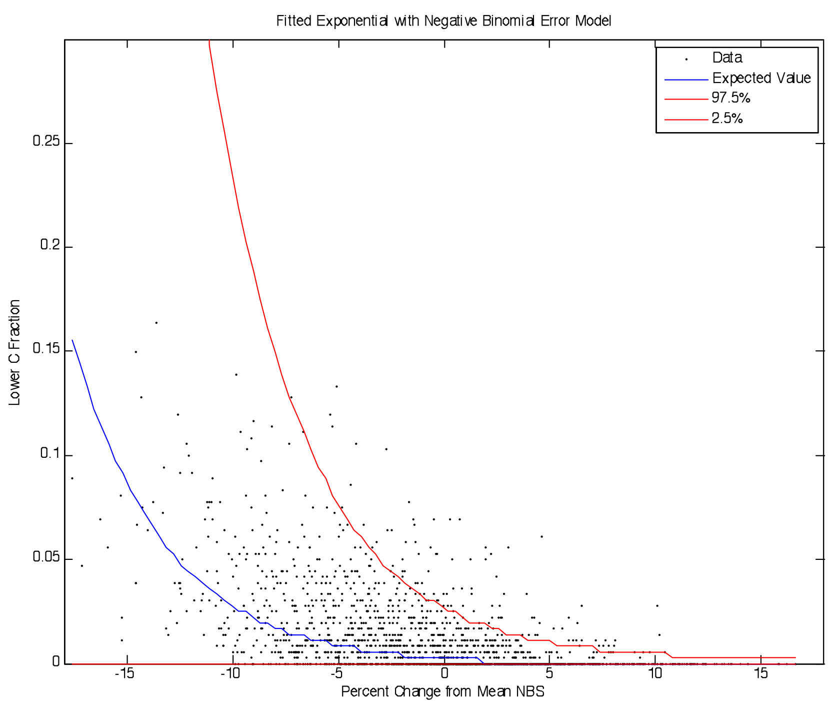


Figure 10: Lake Superior Mean NBS versus Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

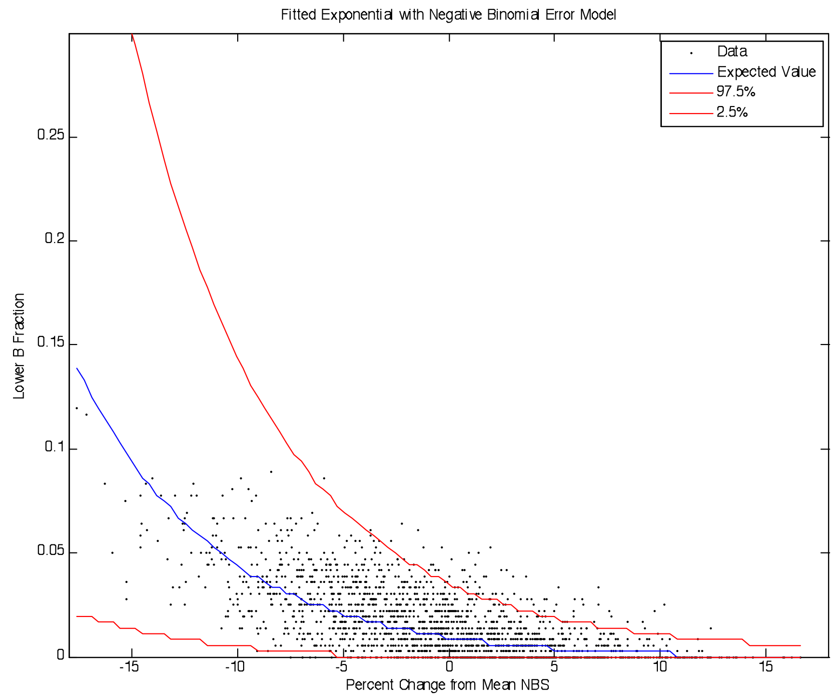


Figure 11: Lake Superior Mean NBS versus Lower Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

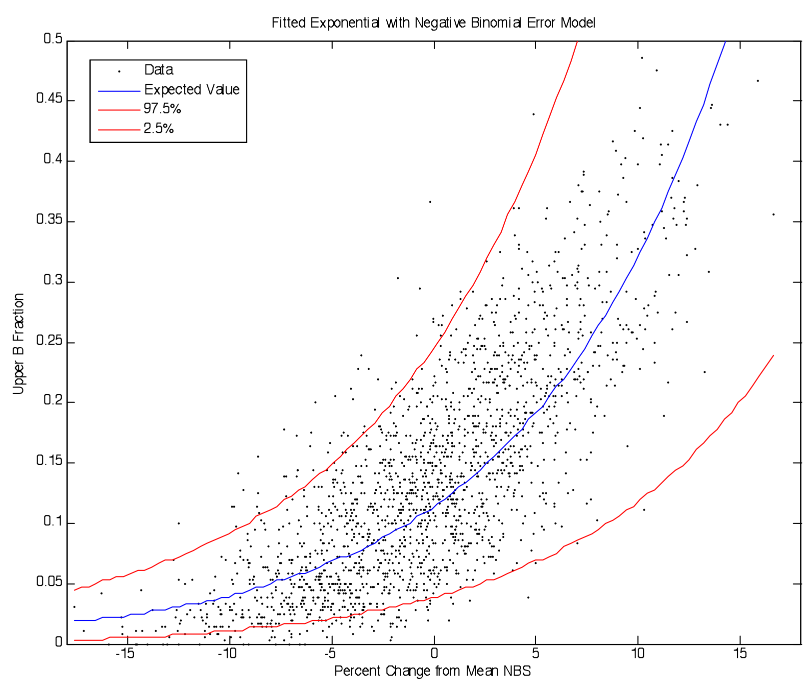


Figure 12: Lake Superior Mean NBS versus Upper Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

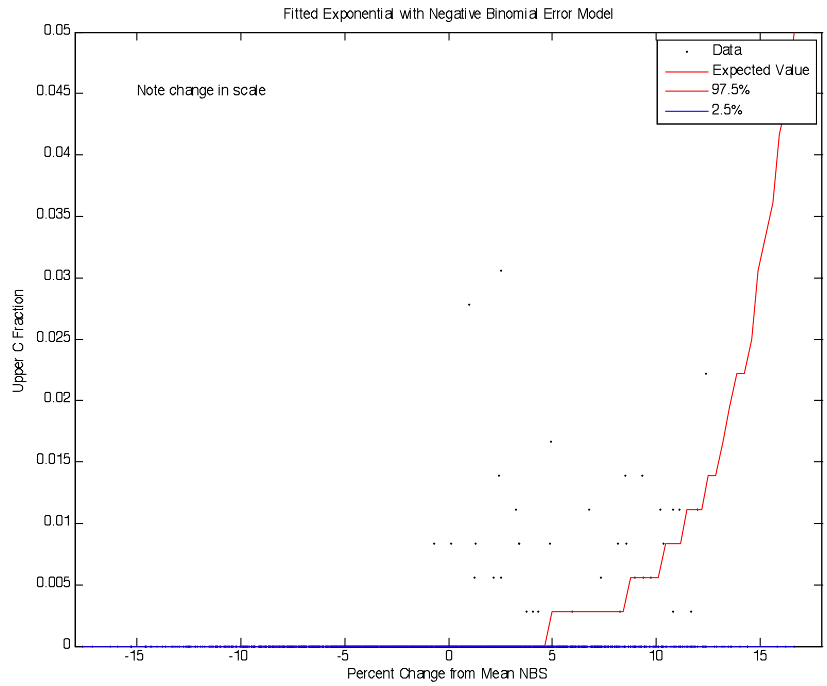


Figure 13: Lake Superior Mean NBS versus Upper Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

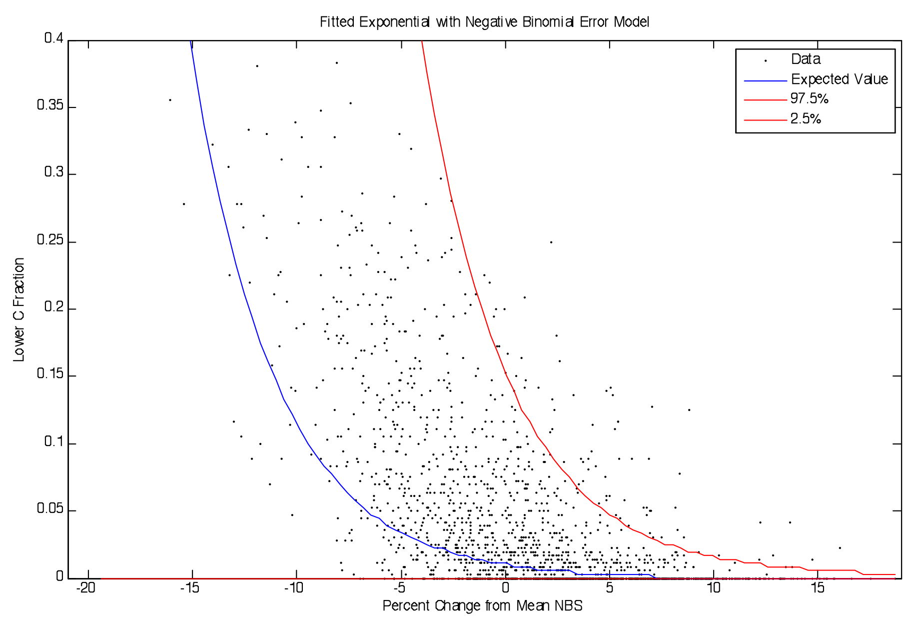


Figure 14: Lakes Michigan-Huron Mean NBS versus Lower Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

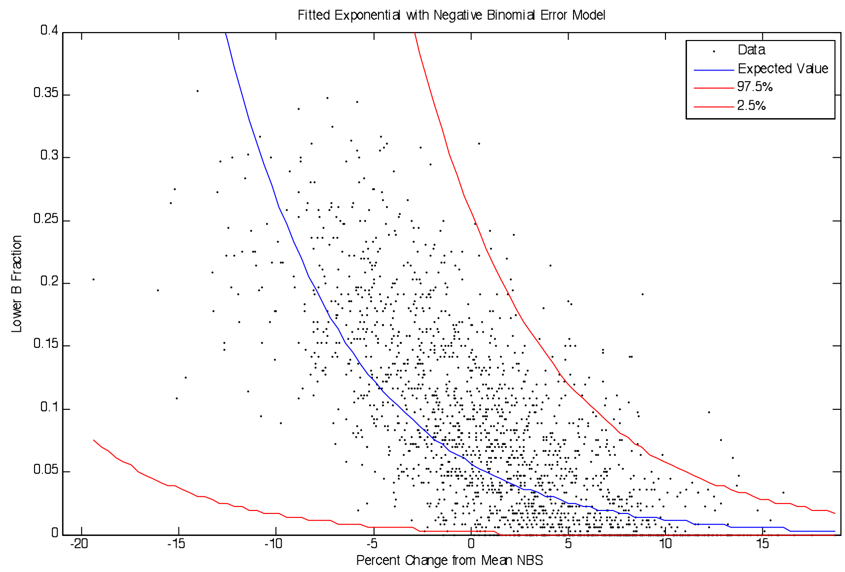


Figure 15: Lakes Michigan-Huron Mean NBS versus Lower Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

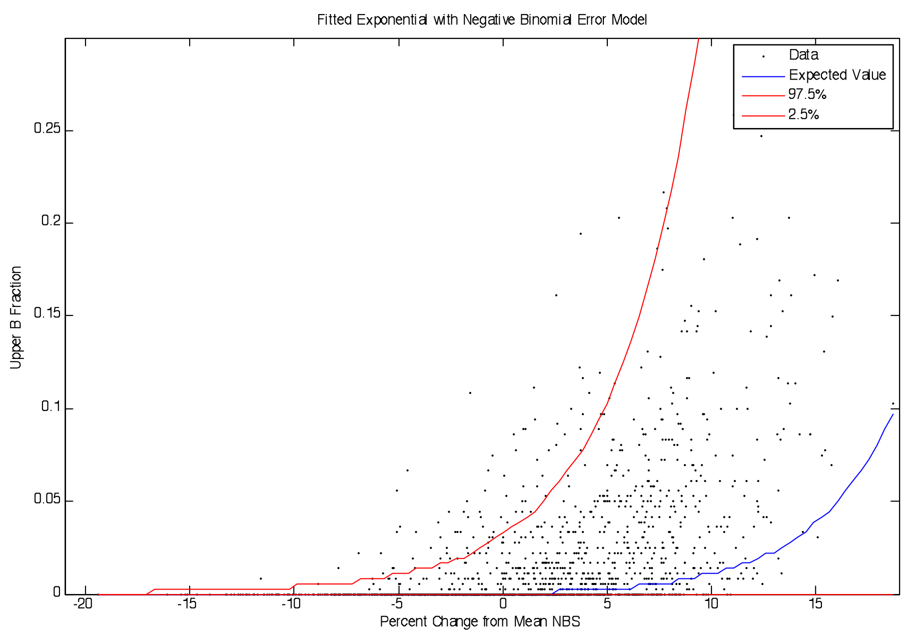


Figure 16: Lakes Michigan-Huron Mean NBS versus Upper Zone B Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

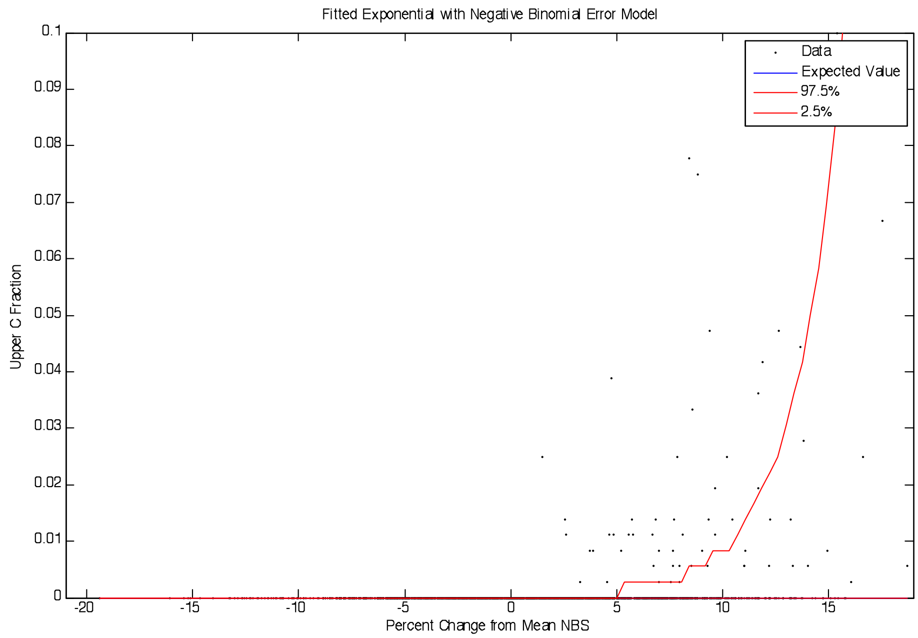


Figure 17: Lakes Michigan-Huron Mean NBS versus Upper Zone C Occurrences for 30 Year Windows from the 50k Year Stochastic Data Set with an exponential deterministic and negative binomial stochastic model.

The sum of the number of expected zone occurrences provides an indication of the climate conditions which would put the Upper Great Lakes at the highest risk given the current regulation plan. The impact is measured by an excessive number of zone occurrences. Figures 18 and 19 show a surface where the surface height, *z*, is equal to the fractional number of expected Lower C, Lower B, Upper B and Upper C occurrences on Lake Superior. Figure 18 shows the zone occurrences as a function of percent change mean NBS versus percent change NBS standard deviation while holding serial correlation constant. Figure 19 shows the relation between NBS percent change and serial correlation to zone occurrences while holding percent change standard deviation constant. Where the sum of the expected zone occurrences is greater than the number of months, the fraction is capped at 1. Figures 20 and 21 show the same relationships for Lakes Michigan-Huron.

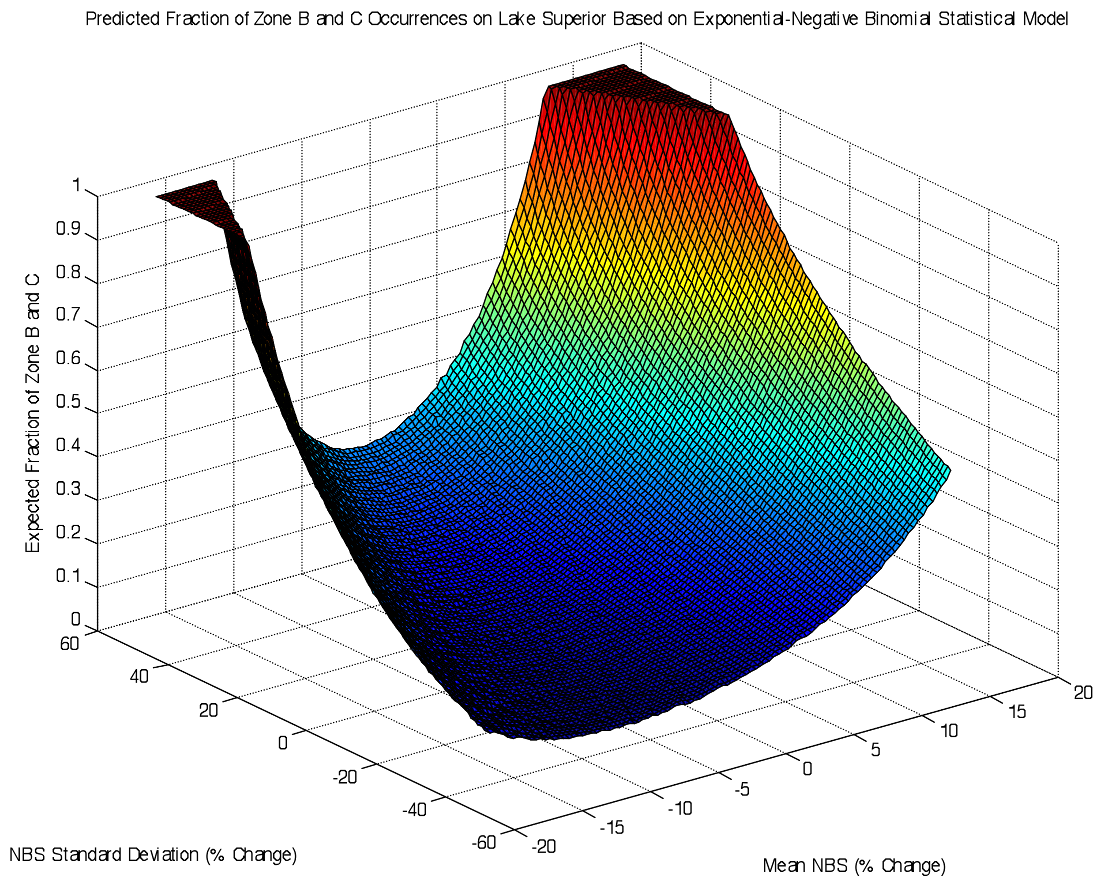


Figure 18: Lake Superior Expected Number of Zones as a Function of Mean NBS and NBS Standard Deviation Based on the Fitted Statistical Model.

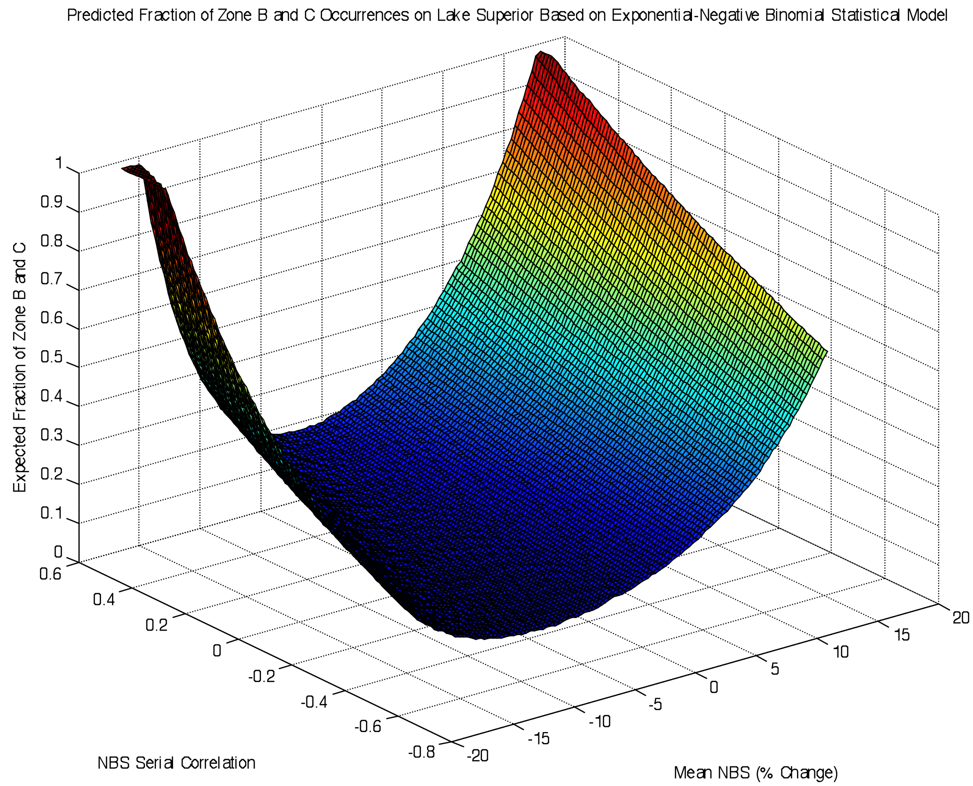


Figure 19: Lake Superior Expected Number of Zones as a Function of Mean NBS and NBS Serial Correlation Based on the Fitted Statistical Model.

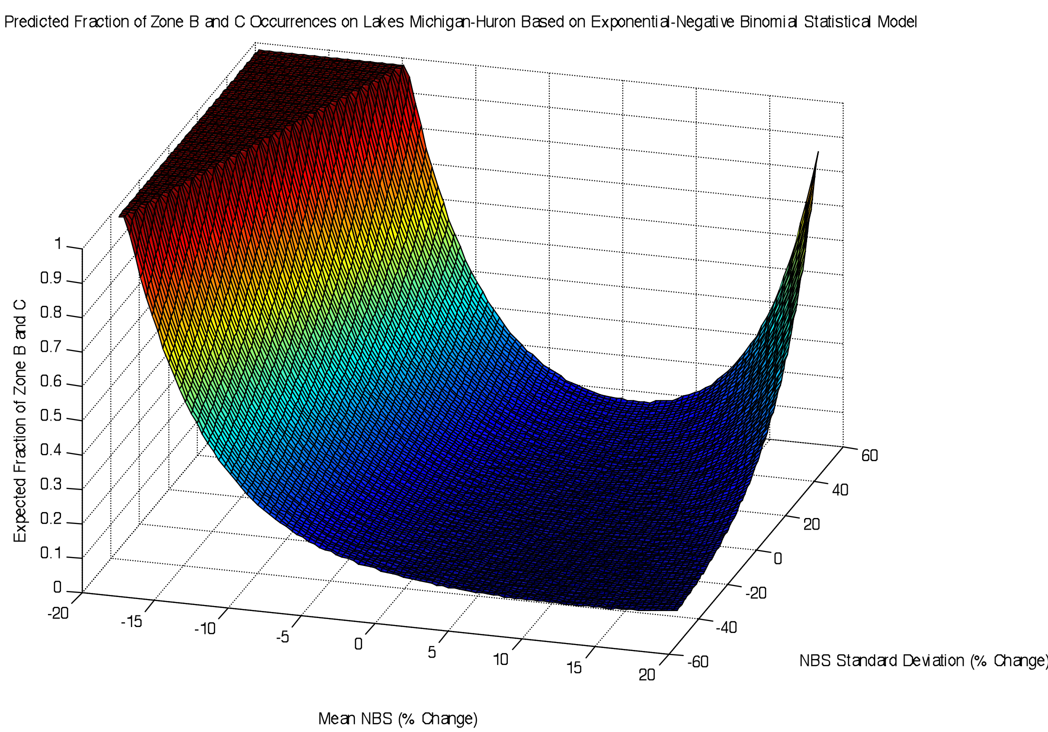


Figure 20: Lakes Michigan-Huron Expected Number of Zones as a Function of Mean NBS and NBS Standard Deviation Based on the Fitted Statistical Model.

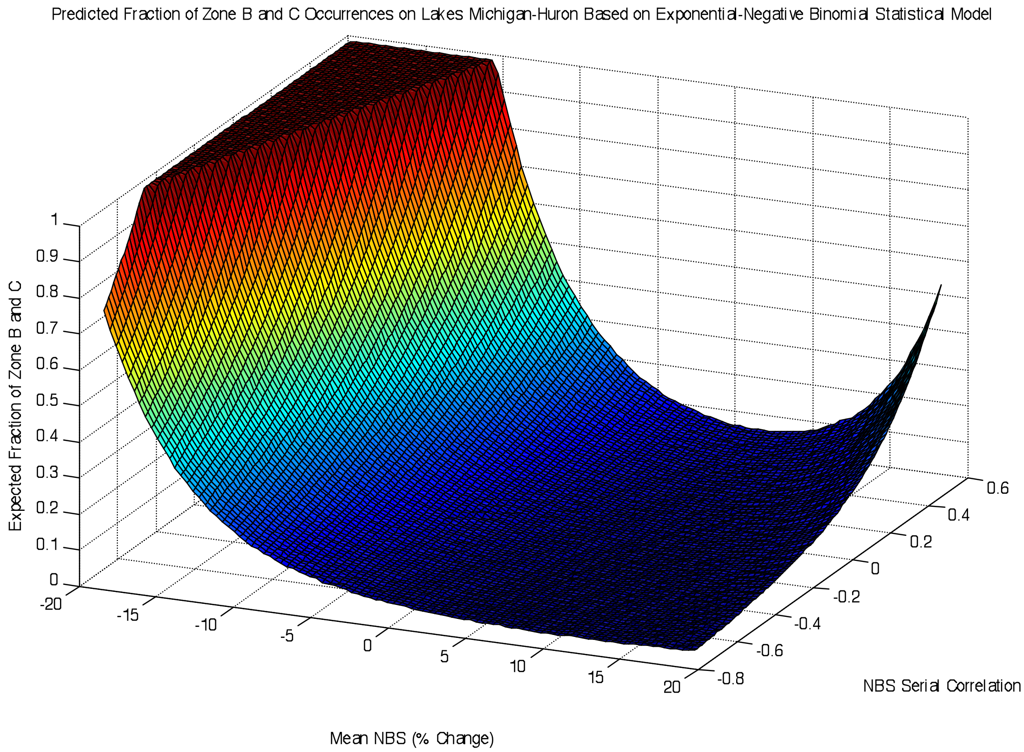


Figure 21: Lakes Michigan-Huron Expected Number of Zones as a Function of Mean NBS and NBS Serial Correlation Based on the Fitted Statistical Model.

Another way to consider the impact of climate on zone occurrences is to look at the histogram and predicted distribution of zone occurrences based on the percent change in mean NBS. Figure 22 shows the histogram and fitted distribution of Lower C Zone occurrences based on the range of mean NBS percent change. The top graph is for a -10% or greater mean NBS change and the bottom graph is for a +10% or greater mean NBS change. The shift in distribution and variability over the range of percent change of mean NBS is clearly evident. Figure 23 shows the same graph for Lower B Zone, Figure 24 shows the graph for Upper B Zone and Figure 25 shows the graph for Upper C Zone. Figure 21 also demonstrates the relatively low occurrence of Upper Zone C. Even at high values for percent change of mean NBS, there are still very few Upper Zone C occurrences.

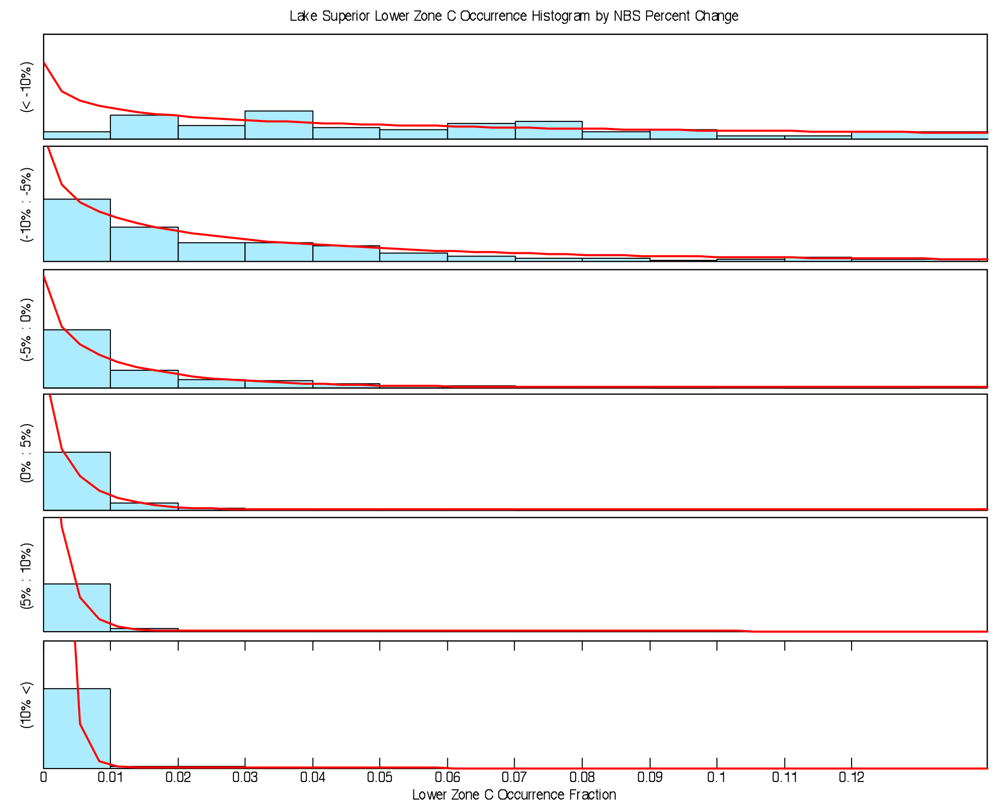


Figure 22: Lake Superior Lower C Zone Occurrence by NBS Percent Change

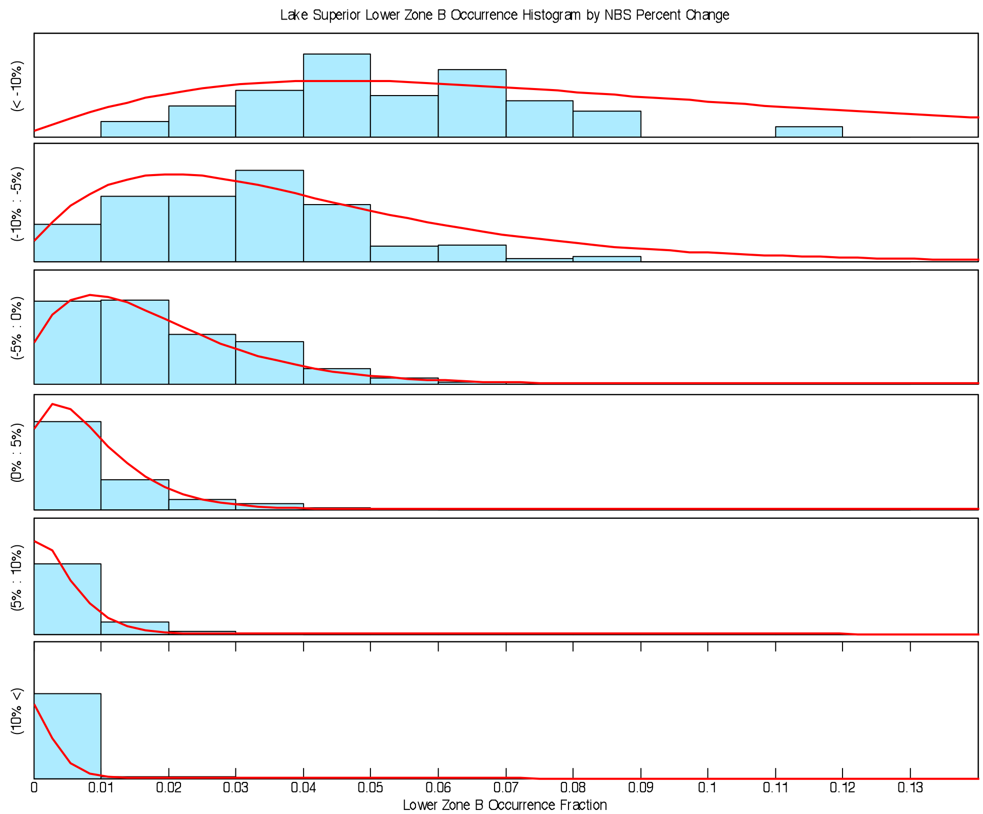


Figure 23: Lake Superior Lower B Zone Occurrence by NBS Percent Change

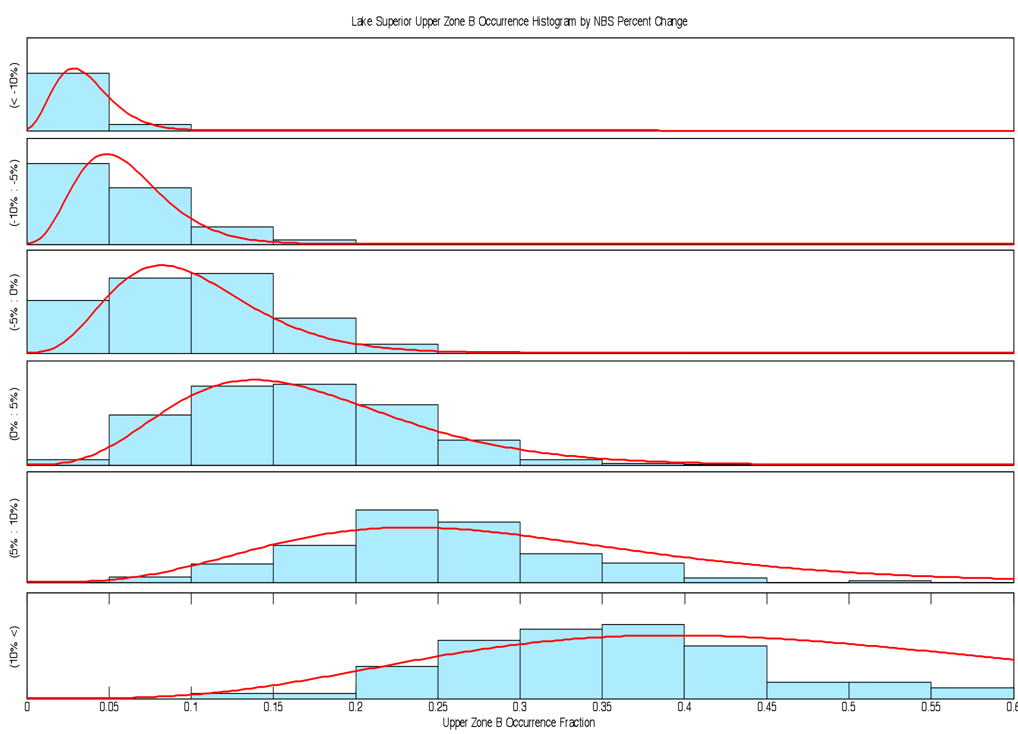


Figure 24: Lake Superior Upper B Zone Occurrence by NBS Percent Change

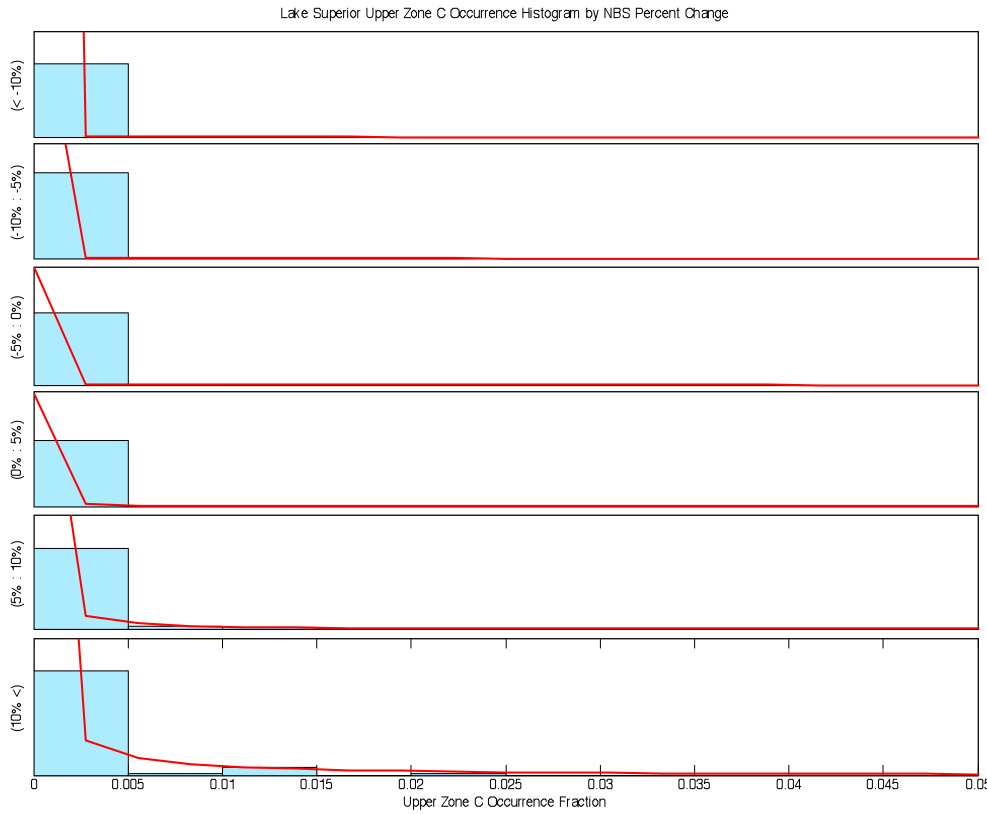


Figure 25: Lake Superior Upper C Zone Occurrence by NBS Percent Change

The relationships developed in the course of this analysis are helpful at identifying climate conditions that result in unfavorable lake conditions. Continued analysis of the GCM based climate projections and the paleo based climate will then help to identify plausible climate conditions. Plausible climate conditions that cause the current operations plan to fail can then be addressed to determine lake management decisions to mitigate the climate effects. The analysis should consider acceptable levels of failure based on zone occurrences.

There are several limitations to the current statistical model. The model treats the zone occurrences as independent and does not constrain the total number of zone occurrences to the number of months in the analysis period. The significance of the limitations increases as the percent change in mean NBS or the percent change in annual NBS standard deviation increases. Unfortunately, this corresponds to the areas in climate space that are the most interesting due to their impact. The UMass Hydrosystems Research Group is examining model alternatives that address these shortcomings.

3.2 Estimation of Climate-informed Plausible Risk

In the application to the Great Lakes, a multi-model, multi-run ensemble of GCM projections is used in combination with stochastically generated timeseries, including those informed by paleodata to describe probabilities. Given the uncertainties associated with the estimation of probabilities, the term "plausibility" has been adopted in its place. The concept of plausibility is best described as a stakeholder developed, subjective ranking of the probability of specific climate states. The concept borrows from the practice of shared vision modeling, in that the estimation of probabilities is not a black box process, but rather a tool for discussion and ranking relative uncertainties during the planning process.

The plausibility of a climate state is generally based on the frequency of occurrence of that state in the climate simulations. In addition, the source of the simulation is considered. For example, climate state that occurs in many runs from multiple GCMs and also occurs in the paleodata-based stochastic simulation is more plausible than a climate state that occurs rarely in a small number of sources. Where the relative plausibility is less clear cut, a discussion of the different sources of the occurrence of the climate states (e.g., specific GCMs) and relative merits of those sources can be discussed among the decision makers (Board members, other experts) facilitated by the AM and PFEG teams. The goal is to use a wide range of climate information in a transparent manner to facilitate comfort for the decision makers in the use of that information for decisions. The decision makers, in this case the Study Board, will be presented with plausibility estimates of climate states associated with each regulation plan and the sources of information that information that assigned probability to that state. The plausibility estimates may be adjusted based on different comfort levels of the Board members with the various climate information sources.

Access to 160 GCM runs, taken from 18 different models originated all across the globe, was made possible with the help of Dr. James Angel and Dr. Kenneth Kunkel. All 160 GCM runs, of 30yrs windows each (centered around 2050), were compared to a GCM Base Case (centered around 1985) to get the Mean NBS Percent Change suggested by these GCMs' models. All runs are established under the emission scenario A_2.

$$MeanNBS\%Change = \frac{\frac{1}{n}\sum_{i=1}^{n} X_{GCM_i} - \frac{1}{n}\sum_{i=1}^{n} X_{Base70-99_i}}{\frac{1}{n}\sum_{i=1}^{n} X_{Base70-99_i}} * 100\%$$

Where, X_{GCMi} = NBS monthly values for GCM runs centered around 2050 ; $X_{Base70\text{-}99i}$ = NBS monthly values for GCM Base Case (centered around 1985) ; n = the number of months in each timeseries; *MeanNBS%Change* = change in NBS, proposed by the GCM runs

A histogram is created to show the models' suggested changes. Figure 26 shows a range for NBS changes at Lake Superior of -28% to 19%, and at Lake Michigan-Huron from -28% to 24%. Distribution for mean NBS percent changes seem to follow a normal distribution, with the range falling outside of the range seen in the 50k stochastic set, meaning that GCMs are suggesting larger extreme scenarios than what the stochastic model is. Now, a study on model consistency and an examination of any discrepancy among the models is completed as followed.

When analyzing plausibility proposed by GCMs, a consideration of the models' resolution is warranted. Previous results had shown a fairly robust difference in the climate change signal for NBS based on models considered to be more or less coarse in resolution. To investigate the effect of resolution on plausible climate changes inferred from the GCMs, conditional histograms of the climate changes from fine resolution models (smaller than 2.5° grids) and from coarse resolution models (bigger than 2.5° grids) were created. Histograms in figure 27 and 28 show the distributions of mean NBS percent changes for both resolution categories. There is a clear disparity in the suggested mean NBS percent changes between the fine resolution models and the course resolution models. The reason for such inconsistency is still unknown. The processes by which these have been downscaled, or perhaps the particular source for each models should be considered suspects for such inconsistencies.

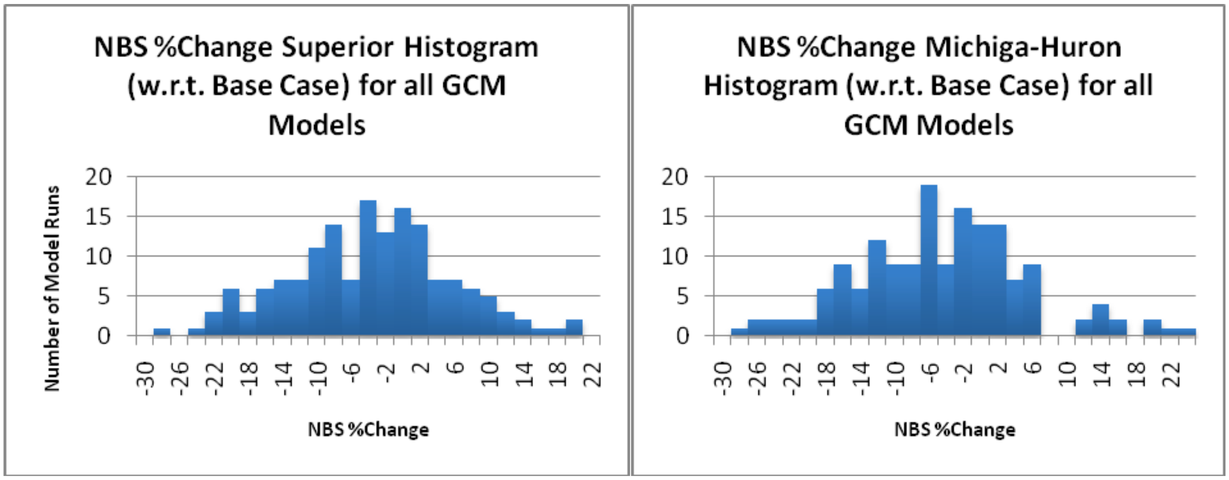


Figure 26: Histogram of NBS %Changes for all GCM Models

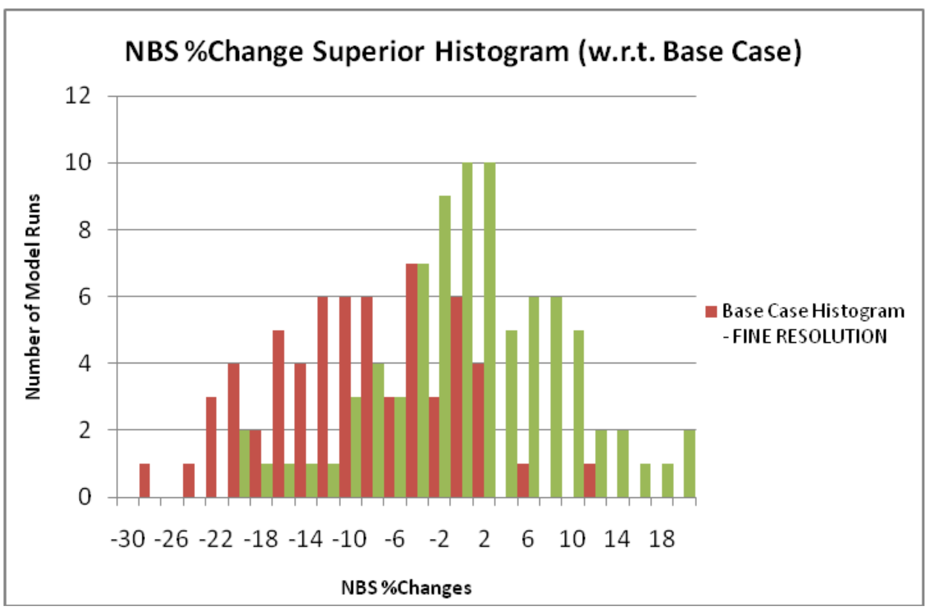


Figure 27: Lake Superior's Histogram of NBS %Changes for all GCM Models, split up into Fine and Coarse Resolution

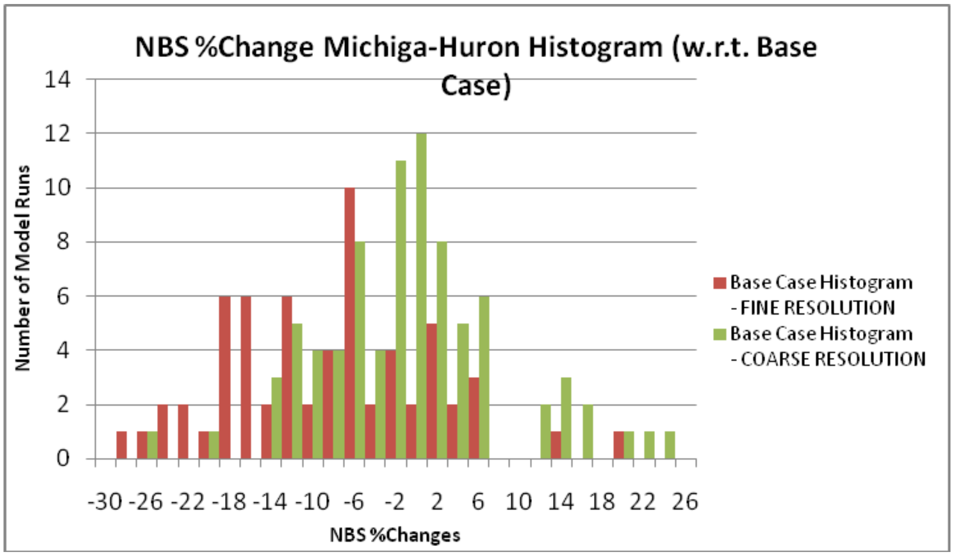


Figure 28: Lake Michigan-Huron's Histogram of NBS %Changes for all GCM Models, split up into Fine and Coarse Resolution

For a better understanding of what these models are actually proposing, the data is then fitted into a normal distribution. This fitting is done for three different datasets: All GCMs, Fine Resolution GCMs, and Coarse Resolution GCMs. The mean value and the standard deviation is calculated for each dataset, and based on these numbers a normal distribution is then created:

$$\mu_{dataset} = \frac{1}{n}\sum_{i=1}^{n} X_i$$

where, $\mu_{dataset}$ = mean value for dataset; X_{seti} = mean NBS percent change for each run

$$\sigma_{dataset} = \sqrt{\frac{1}{n}\sum_{i=1}^{n}(X_i - \mu_{dataset})^2}$$

where, $\sigma_{dataset}$ = standard deviation for dataset

Tables 3 and 4 present the normal distribution values for each of the datasets, highlighting four specific mean NBS percent changes (-10, -5, 5, 10), and their respective probabilities. The exceedence probability is the probability of a variable being exceeded given the statistical distribution, while the cumulative probability is the probability of a variable being greater than or equal to a random sample from the same distribution. The probability density function (PDF) is also calculated and placed into a column next to the exceedance probability. Figures 25 and 26 illustrate the PDF curve for each of the GCM datasets.

Table 3: Lake Superior's Exceedence probabilities for the normally distributed series focused in the -10%, -5%, 5%, 10% NBS Change

Base Case Normal Distribution - ALL MODELS				
Mean	-4.771	n = 160	95% Confidence Internval	
Stand Dev	9.332		0.025	0.975
			-23.062	13.5192
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.003	0.997	0.001	
-28	0.006	0.994	0.002	
-26	0.011	0.989	0.003	
-24	0.020	0.980	0.005	
-22	0.032	0.968	0.008	
-20	0.051	0.949	0.011	
-18	0.078	0.922	0.016	
-16	0.114	0.886	0.021	
-14	0.161	0.839	0.026	
-12	0.219	0.781	0.032	
-10	0.288	0.712	0.037	
-8	0.365	0.635	0.040	
-6	0.448	0.552	0.042	
-5	0.490	0.510	0.043	
-4	0.533	0.467	0.043	
-2	0.617	0.383	0.041	
0	0.695	0.305	0.038	
2	0.766	0.234	0.033	
4	0.826	0.174	0.027	
5	0.852	0.148	0.025	
6	0.876	0.124	0.022	
8	0.914	0.086	0.017	
10	0.943	0.057	0.012	
12	0.964	0.036	0.009	
14	0.978	0.022	0.006	
16	0.987	0.013	0.004	
18	0.993	0.007	0.002	
20	0.996	0.004	0.001	

Base Case Normal Distribution - FINE RESOLUTION				
Mean	-10.191	n = 63	95% Confidence Internval	
Stand Dev	8.135		0.025	0.975
			-26.1366	5.75392
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.007	0.993	0.003	
-28	0.014	0.986	0.004	
-26	0.026	0.974	0.007	
-24	0.045	0.955	0.012	
-22	0.073	0.927	0.017	
-20	0.114	0.886	0.024	
-18	0.169	0.831	0.031	
-16	0.238	0.762	0.038	
-14	0.320	0.680	0.044	
-12	0.412	0.588	0.048	
-10	0.509	0.491	0.049	
-8	0.606	0.394	0.047	
-6	0.697	0.303	0.043	
-5	0.738	0.262	0.040	
-4	0.777	0.223	0.037	
-2	0.843	0.157	0.030	
0	0.895	0.105	0.022	
2	0.933	0.067	0.016	
4	0.959	0.041	0.011	
5	0.969	0.031	0.009	
6	0.977	0.023	0.007	
8	0.987	0.013	0.004	
10	0.993	0.007	0.002	
12	0.997	0.003	0.001	

Base Case Normal Distribution - COARSE RESOLUTION				
Mean	-0.164	n = 82	95% Confidence Internval	
Stand Dev	8.366		0.025	0.975
			-16.559925	16.2327
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.000	1.000	0.000	
-28	0.000	1.000	0.000	
-26	0.001	0.999	0.000	
-24	0.002	0.998	0.001	
-22	0.005	0.995	0.002	
-20	0.009	0.991	0.003	
-18	0.016	0.984	0.005	
-16	0.029	0.971	0.008	
-14	0.049	0.951	0.012	
-12	0.079	0.921	0.018	
-10	0.120	0.880	0.024	
-8	0.174	0.826	0.031	
-6	0.243	0.757	0.037	
-5	0.282	0.718	0.040	
-4	0.323	0.677	0.043	
-2	0.413	0.587	0.047	
0	0.508	0.492	0.048	
2	0.602	0.398	0.046	
4	0.691	0.309	0.042	
5	0.731	0.269	0.039	
6	0.769	0.231	0.036	
8	0.835	0.165	0.030	
10	0.888	0.112	0.023	
12	0.927	0.073	0.017	
14	0.955	0.045	0.011	
16	0.973	0.027	0.007	
18	0.985	0.015	0.005	
20	0.992	0.008	0.003	

Table 4: Lake Michigan-Huron's Exceedence probabilities for the normally distributed series focused in the -10%, -5%, 5%, 10% NBS Change

Base Case Normal Distribution - ALL MODELS				
Mean	-5.508	n = 160	95% Confidence Internval	
Stand Dev	9.788		0.025	0.975
			-24.692	13.676
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.006	0.994	0.002	
-28	0.011	0.989	0.003	
-26	0.018	0.982	0.005	
-24	0.029	0.971	0.007	
-22	0.046	0.954	0.010	
-20	0.069	0.931	0.014	
-18	0.101	0.899	0.018	
-16	0.142	0.858	0.023	
-14	0.193	0.807	0.028	
-12	0.254	0.746	0.033	
-10	0.323	0.677	0.037	
-8	0.400	0.600	0.039	
-6	0.480	0.520	0.041	
-5	0.521	0.479	0.041	
-4	0.561	0.439	0.040	
-2	0.640	0.360	0.038	
0	0.713	0.287	0.035	
2	0.778	0.222	0.030	
4	0.834	0.166	0.025	
5	0.858	0.142	0.023	
6	0.880	0.120	0.020	
8	0.916	0.084	0.016	
10	0.943	0.057	0.012	
12	0.963	0.037	0.008	
14	0.977	0.023	0.006	
16	0.986	0.014	0.004	
18	0.992	0.008	0.002	
20	0.995	0.005	0.001	

Base Case Normal Distribution - FINE RESOLUTION				
Mean	-9.344	n = 63	95% Confidence Internval	
Stand Dev	9.721		0.025	0.975
			-28.397	9.709
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.017	0.983	0.004	
-28	0.027	0.973	0.007	
-26	0.043	0.957	0.009	
-24	0.066	0.934	0.013	
-22	0.096	0.904	0.018	
-20	0.136	0.864	0.023	
-18	0.187	0.813	0.028	
-16	0.247	0.753	0.032	
-14	0.316	0.684	0.037	
-12	0.392	0.608	0.040	
-10	0.473	0.527	0.041	
-8	0.555	0.445	0.041	
-6	0.635	0.365	0.039	
-5	0.673	0.327	0.037	
-4	0.709	0.291	0.035	
-2	0.775	0.225	0.031	
0	0.832	0.168	0.026	
2	0.878	0.122	0.021	
4	0.915	0.085	0.016	
5	0.930	0.070	0.014	
6	0.943	0.057	0.012	
8	0.963	0.037	0.008	
10	0.977	0.023	0.006	
12	0.986	0.014	0.004	
14	0.992	0.008	0.002	
16	0.995	0.005	0.001	

Base Case Normal Distribution - COARSE RESOLUTION				
Mean	-1.791	n = 82	95% Confidence Internval	
Stand Dev	9.013		0.025	0.975
			-19.457	15.874
% Base Change	Cumm. Probability	Excedence	PDF	
-30	0.001	0.999	0.000	
-28	0.002	0.998	0.001	
-26	0.004	0.996	0.001	
-24	0.007	0.993	0.002	
-22	0.012	0.988	0.004	
-20	0.022	0.978	0.006	
-18	0.036	0.964	0.009	
-16	0.057	0.943	0.013	
-14	0.088	0.912	0.018	
-12	0.129	0.871	0.023	
-10	0.181	0.819	0.029	
-8	0.245	0.755	0.035	
-6	0.320	0.680	0.040	
-5	0.361	0.639	0.042	
-4	0.403	0.597	0.043	
-2	0.491	0.509	0.044	
0	0.579	0.421	0.043	
2	0.663	0.337	0.041	
4	0.740	0.260	0.036	
5	0.774	0.226	0.033	
6	0.806	0.194	0.030	
8	0.861	0.139	0.025	
10	0.905	0.095	0.019	
12	0.937	0.063	0.014	
14	0.960	0.040	0.010	
16	0.976	0.024	0.006	
18	0.986	0.014	0.004	
20	0.992	0.008	0.002	
22	0.996	0.004	0.001	
24	0.998	0.002	0.001	
26	0.999	0.001	0.000	

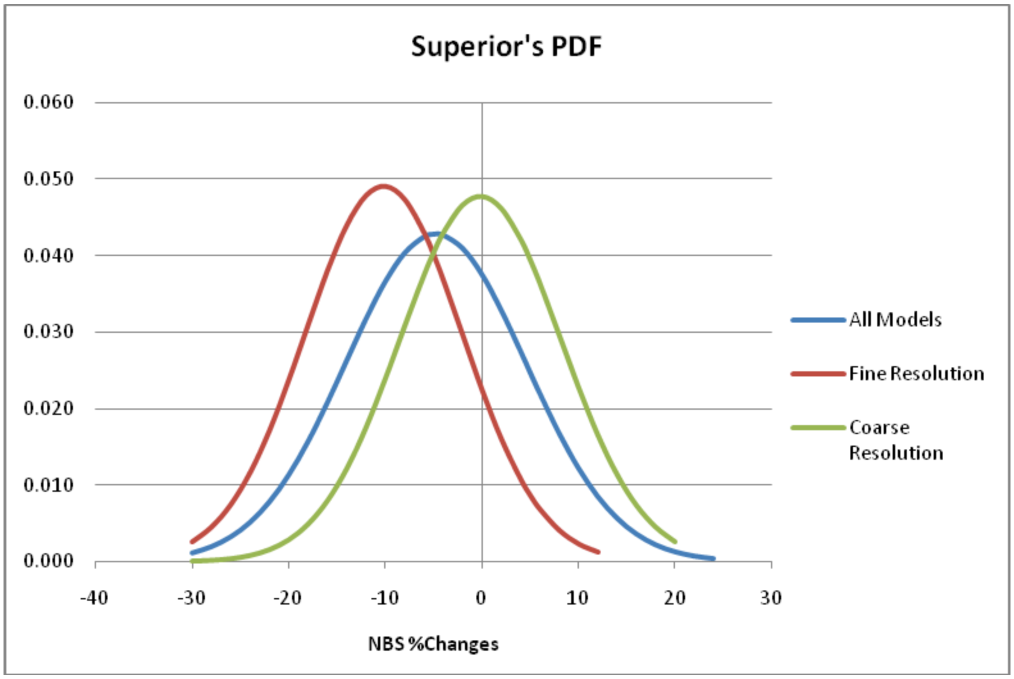


Figure 29: Lake Superior's Normal Distribution graph for all GCM models, for Fine Resolution models, and for Coarse Resolution models

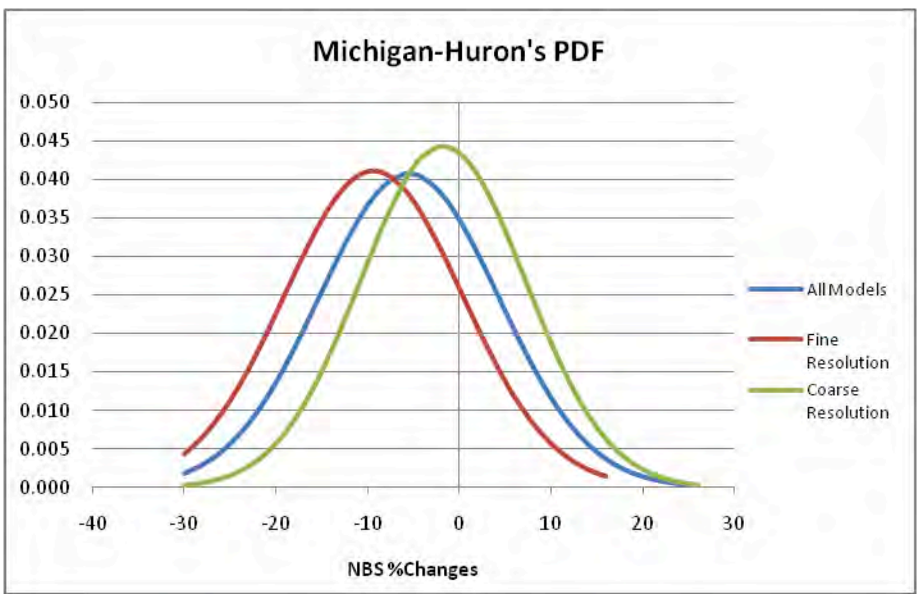


Figure 30: Lake Michigan-Huron's Normal Distribution graph for all GCM models, for Fine Resolution models, and for Coarse Resolution models

The T-test is a statistical hypothesis test appropriate for determining if two data sets are sampled from the same population. The null hypothesis is that the data sets are from the same population while the alternate is that they are from different populations with different means. With a P value of less than 0.05, the null hypothesis can be rejected with a 95% confidence. This means that with a 95% confidence it can be said that the two datasets in question are in fact not coming from the same population. Tables 5 and 6 exhibit some P values for different pairs of datasets, all coming from within the 160 GCM runs. The implications are that the coarse resolution and fine resolution results are statistically significantly different. However, it is unclear if the differences are due to the GCMs themselves, or due to differences in downscaling methods that were applied to the different models by Angel and Kunkel.

Table 5: Lake Superior's P-values for T-test

***P* values for Lake Superior's T-TEST**						
Models	All	Fine Res.	Coarse Res.	European	American	Aus/Asian
All		3.4E-05	1.4E-04	0.408	0.982	0.208
Fine Res.	3.4E-05		2.6E-11			
Coarse Res.	1.4E-04	2.6E-11				
European	0.408				0.442	0.131
American	0.982			0.442		0.256
Aus/Asian	0.208			0.131	0.256	

P < 0.05 'Can regret Null Hypothesis at 95% confidence
P > 0.05 'Can not regret Null Hypothesis at 95% confidence

Table 6: Lake Michigan-Huron's P-values for T-test

***P* values for Lake Mich-Huron's T-TEST**						
Models	All	Fine Res.	Coarse Res.	European	American	Aus/Asian
All		9.2E-03	3.6E-03	0.905	0.176	0.210
Fine Res.	9.2E-03		4.6E-06			
Coarse Res.	3.6E-03	4.6E-06				
European	0.905				0.503	0.564
American	0.176			0.503		0.040
Aus/Asian	0.210			0.564	0.040	

P < 0.05 'Can regret Null Hypothesis at 95% confidence
P > 0.05 'Can not regret Null Hypothesis at 95% confidence

It is useful to estimate climate impact from the GCM series to help establish risk. Using the Climate Response Function developed in our group, one can translate GCM climate statistics into lake impacts. The Climate Response Function requires three inputs: mean NBS percent change, annual standard deviation percent change, and annual serial correlation. The Climate response function was generated using percent change from the historic NBS series. To evaluate the GCMs using the Climate Response Function, a bias correction needs to be applied. This corrects the GCM percent change from the GCM base case to the historic NBS case using the historic NBS window corresponding to the base case. The following adjustment was applied to the mean NBS percent change, and to the standard deviation percent change:

$$\Delta GCM_H\% = \frac{X_{GCM} - [X_{Base70-99} - (X_{Hist70-99} - X_{Hist})]}{X_{Base70-99} - (X_{Hist70-99} - X_{Hist})} * 100\%$$

where, X_{GCM} = GCM statistic (centered around 2050) ; $X_{Base70\text{-}99}$ = GCM statistic for Base Case (centered around 1985) ; $X_{Hist70\text{-}99}$ = statistic for historic data (from 1970-99) ; X_{Hist} = statistic for historic data (from 1900-2006) ; $\Delta GCM_H\%$ = statistic percentage change suggested by GCM with respect to historic values. With the adjustment to the mean and standard deviation, the statistics from the 160 GCM runs can be used in the Climate Response Function. An initial attempt at this has been completed and the figures are shown in Figures 31 to 34 for Lakes Superior and Michigan-Huron.

Next Steps: The next step is to calculate probabilities associated with the risk levels through use of the climate response function with the various sources of climate information. We will also investigate the different approaches to weighting different sources. That work is currently being accomplished.

Task 4. Adaptive Management

The use of a dynamic regulation plan is envisioned to produce a robust regulation strategy for a broad range of future climates. However, it is well known that there are other uncertainties, including faulty assumptions and unforeseen surprises, which threaten the success of the regulation plan. For this reason, an adaptive management process is being incorporated into the regulation of Lake Superior. The process consists of long term monitoring of regulation plan performance and mechanisms for implementing changes when needed. Figure C1 in the Appendix illustrates the historical approach to management of Lake Superior regulation in comparison to the proposed adaptive management strategy (Figure C2).

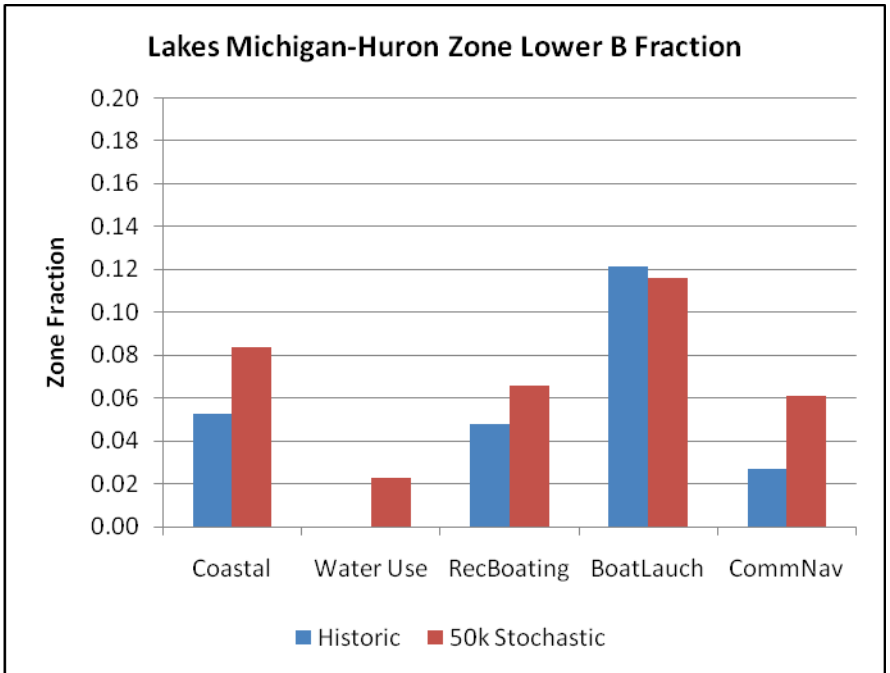
For any adaptive management process, monitoring is critical. The data gathered through carefully designed monitoring allows evaluation of the performance of the regulation plan and the need for changes, including regulation rule changes, changes to plan objectives or other possibilities that we cannot anticipate. The observations will provide direct feedback on plan performance. In addition, monitoring will be designed to evaluate the degree to which the coping zones are effective in estimating plan performance. Since there is uncertainty in the estimation of the coping zones by the working groups, it is possible that significant negative impacts may be accumulating for a stakeholder group despite lake levels remaining out of zone C. Adjustment to the zones themselves may be necessary.

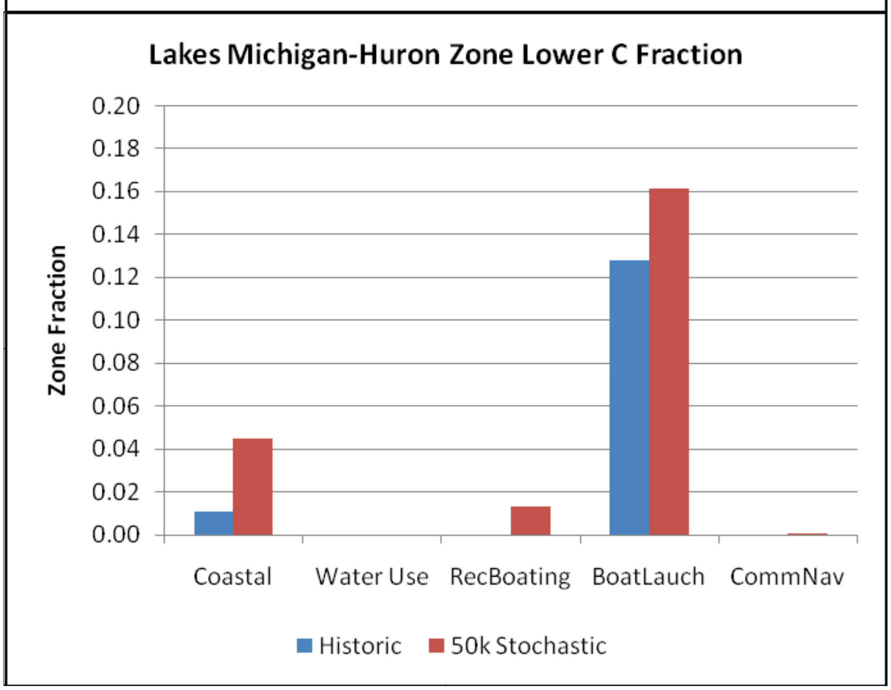
In order to sustain monitoring and provide mechanisms for use of the collected data in decision making, an institutional framework for the adaptive management process is required. Previous studies have shown that adaptive management often fails do to a lack of institutional buy in (Walters, 2007; Allan and Allan, 2005). The IUGLS study board is committed to implementing an adaptive management process. The AM leadership is conducting an institutional analysis that will investigate how the process will be funded, who would be responsible for each element of the plan, and how decisions will be made and implemented. The study board will recommend adaptive management to the IJC, and the common assumption is that a number of U.S. and Canadian agencies would agree to carry out different elements of the plan. This will not guarantee that adaptive management will occur even if these tasks are done well. But the adaptive management process has been designed to improve the odds of successful implementation.

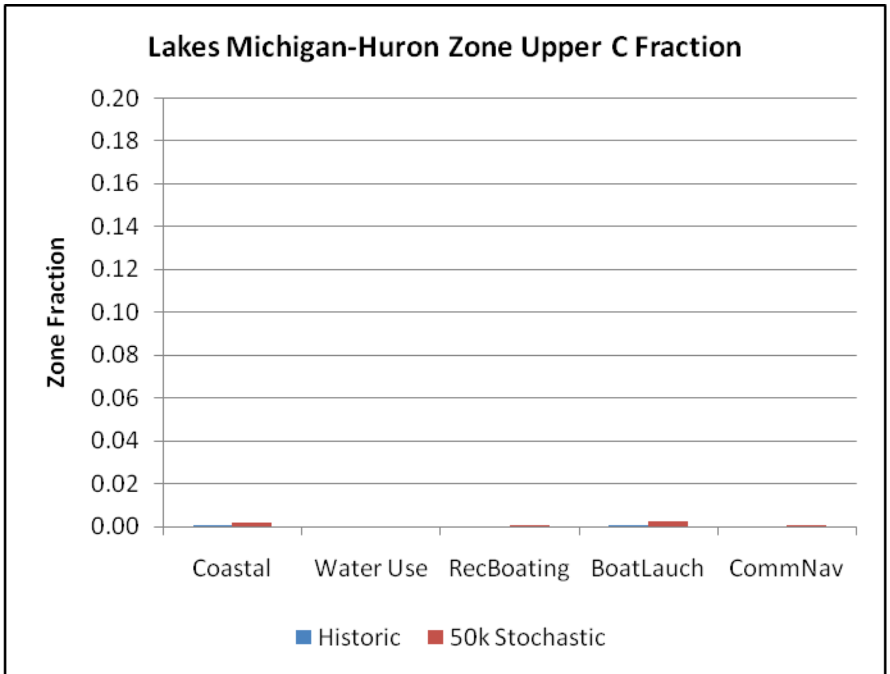
Next Steps: Support the AM leadership with results, especially estimates of plausible risk and risk beyond regulation, that can be used to build the institutional case for the need for adaptive management and to identify and enlist partners in the effort to improve the adaptive capacity of the Great Lakes stakeholders and affected communities.

APPENDIX

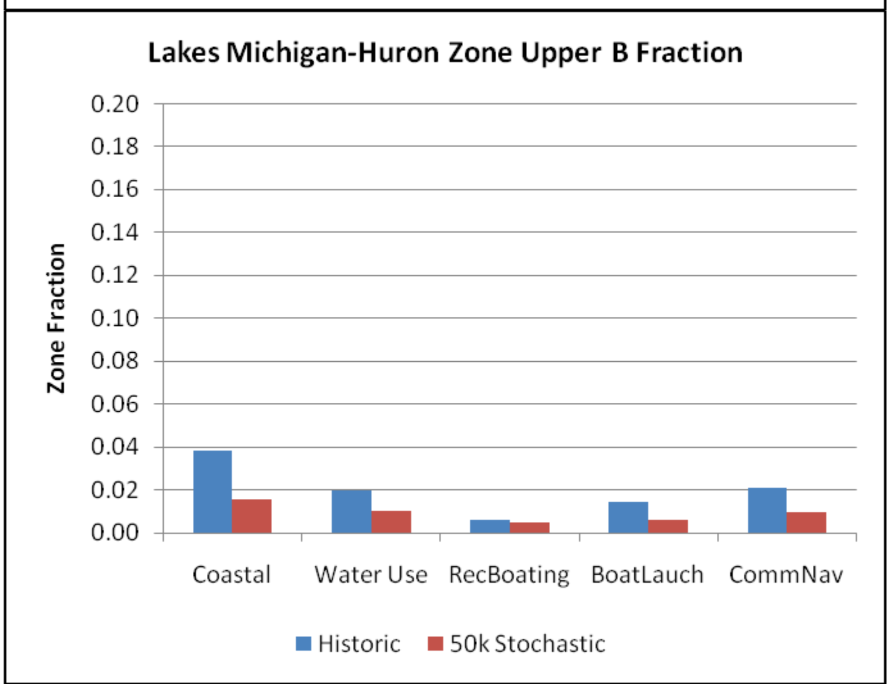
Figures A1 - A4. Coping Zone occurrences on Lakes Michigan-Huron.





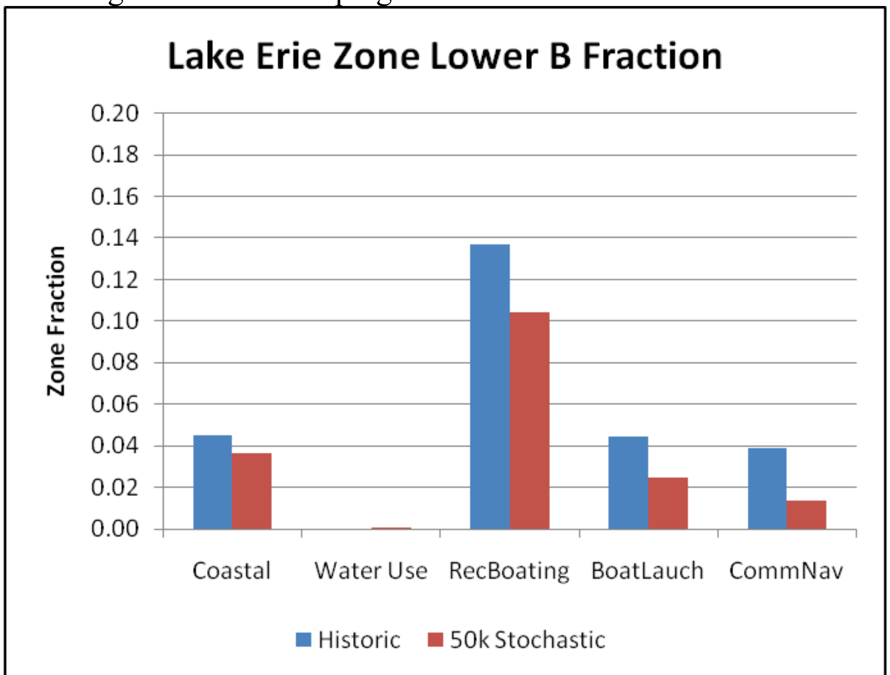


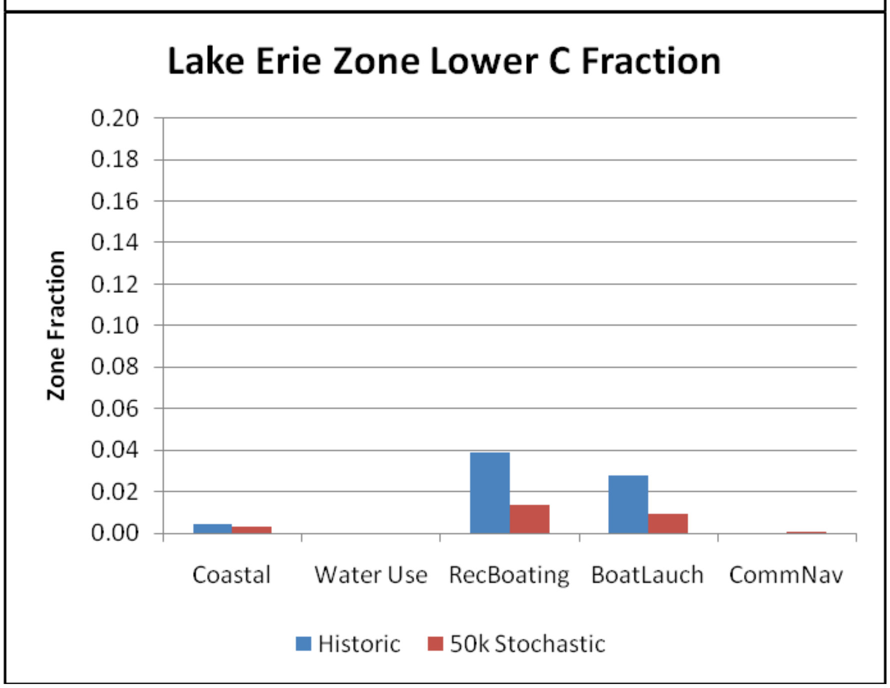
Lakes Michigan-Huron Zone Upper C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

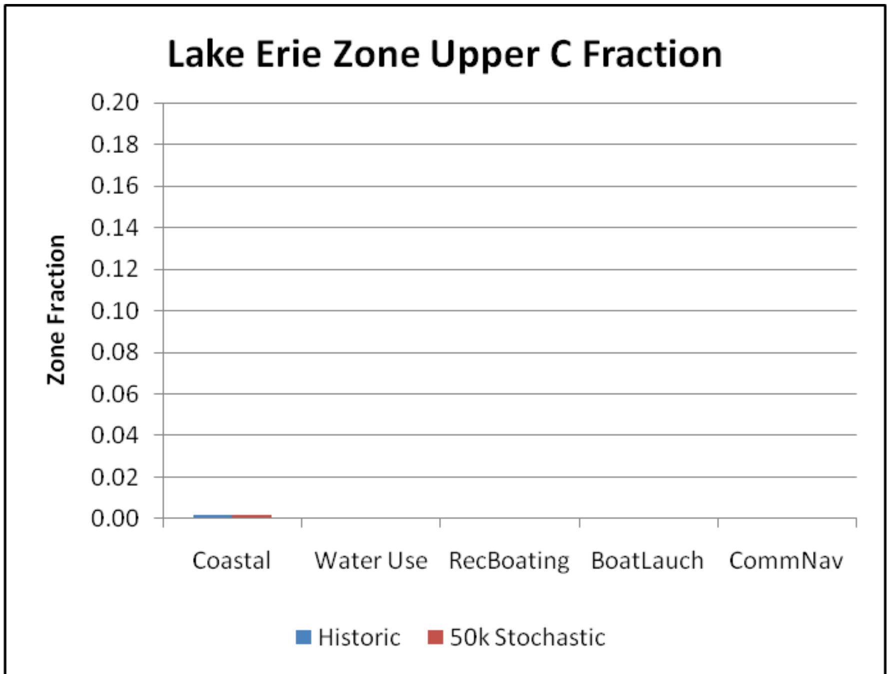


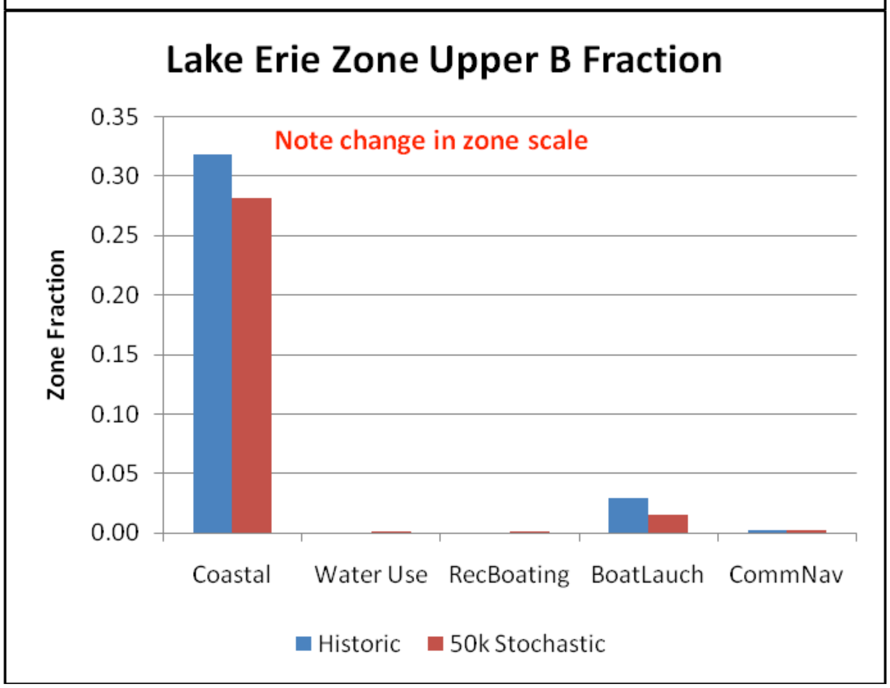
Lakes Michigan-Huron Zone Upper B Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

Figures A5-A8. Coping Zone occurrences on Lake Erie.

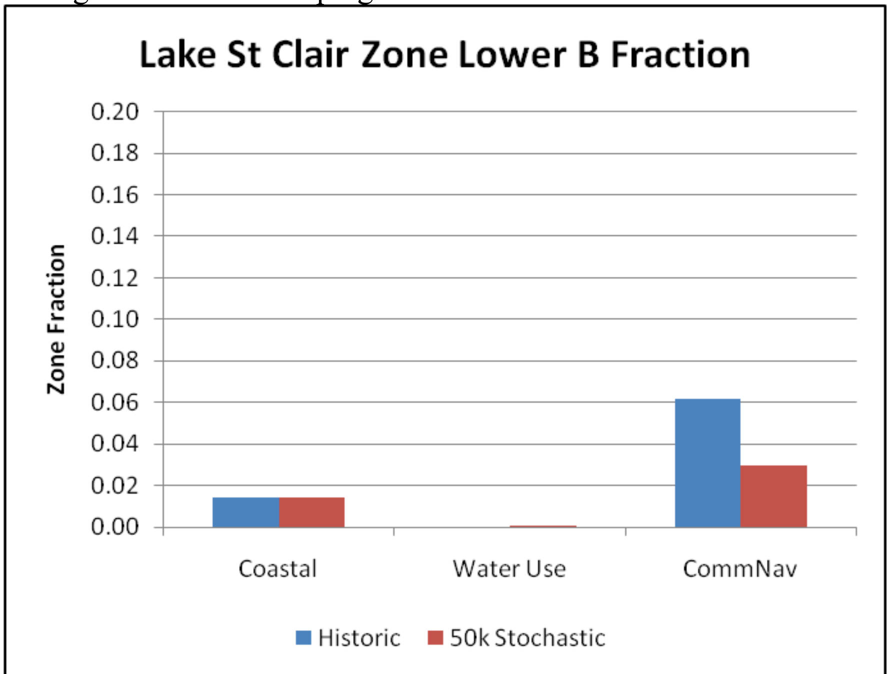


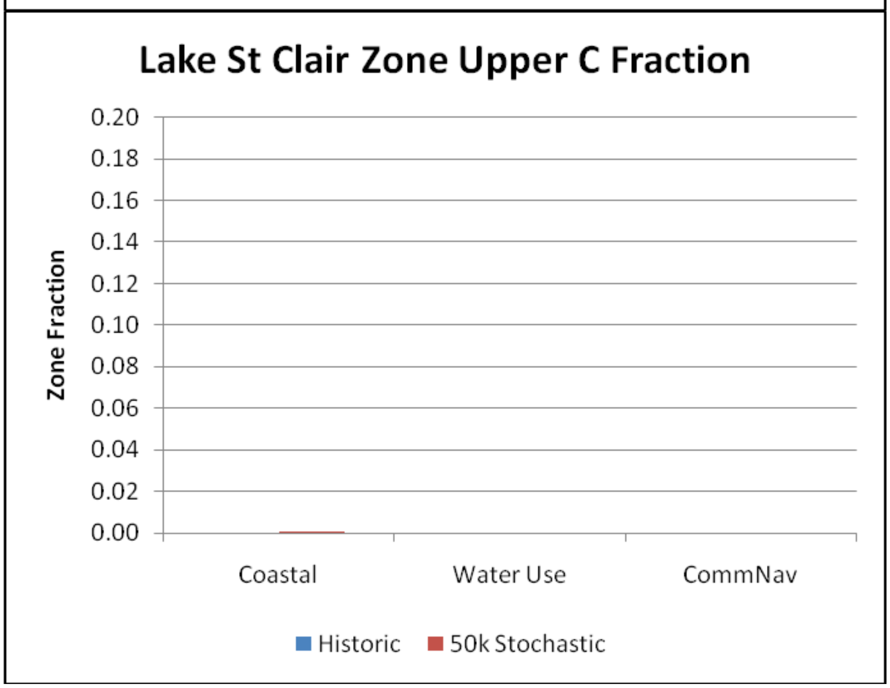


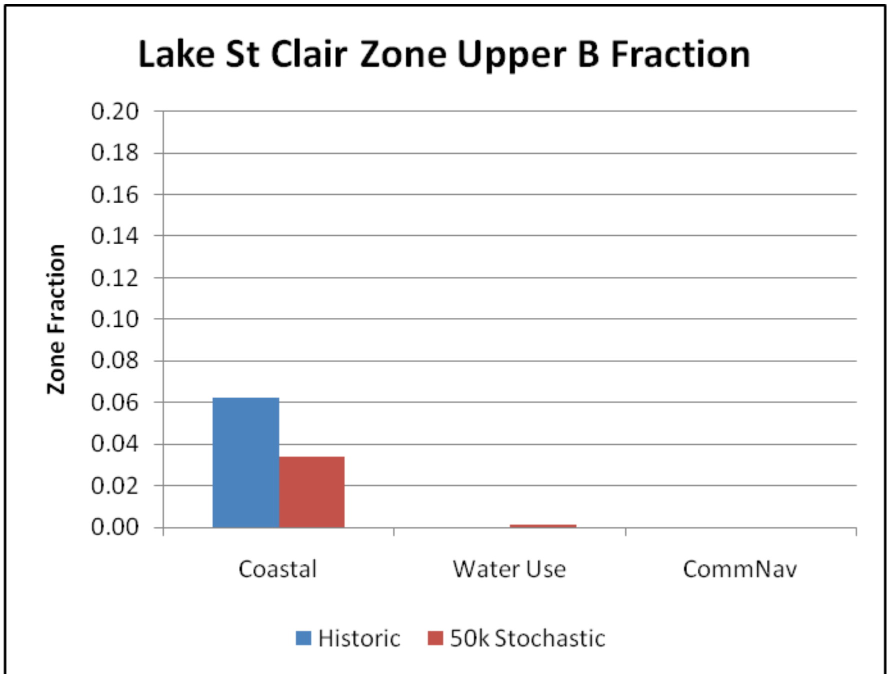

Lake Erie Zone Upper C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic


Lake Erie Zone Upper B Fraction
Note change in zone scale
Zone Fraction
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0.00
Coastal
Water Use
RecBoating
BoatLauch
CommNav
Historic
50k Stochastic

Figures A9-A12. Coping Zone occurrences on Lake St. Clair.






Lake St Clair Zone Upper B Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
CommNav
Historic
50k Stochastic

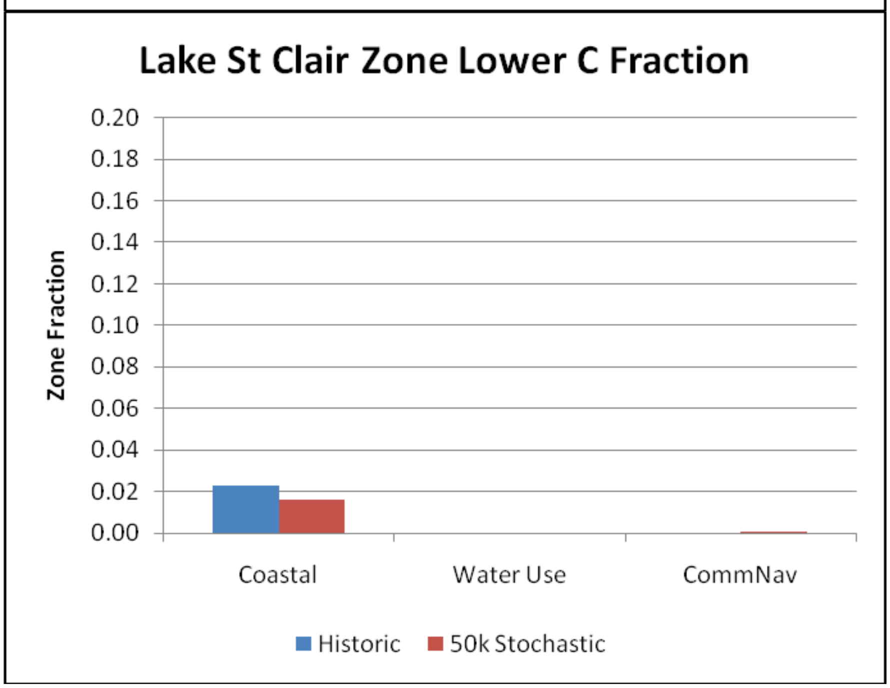

Lake St Clair Zone Lower C Fraction
Zone Fraction
0.20
0.18
0.16
0.14
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Coastal
Water Use
CommNav
Historic
50k Stochastic

Figure B1. Lake Superior levels based on a 5% increase in 77A Lake Superior outflow from Jan through Aug based on a high lake level with a high NBS forecast. Note that AM refers to 77A with forecast adjustment.

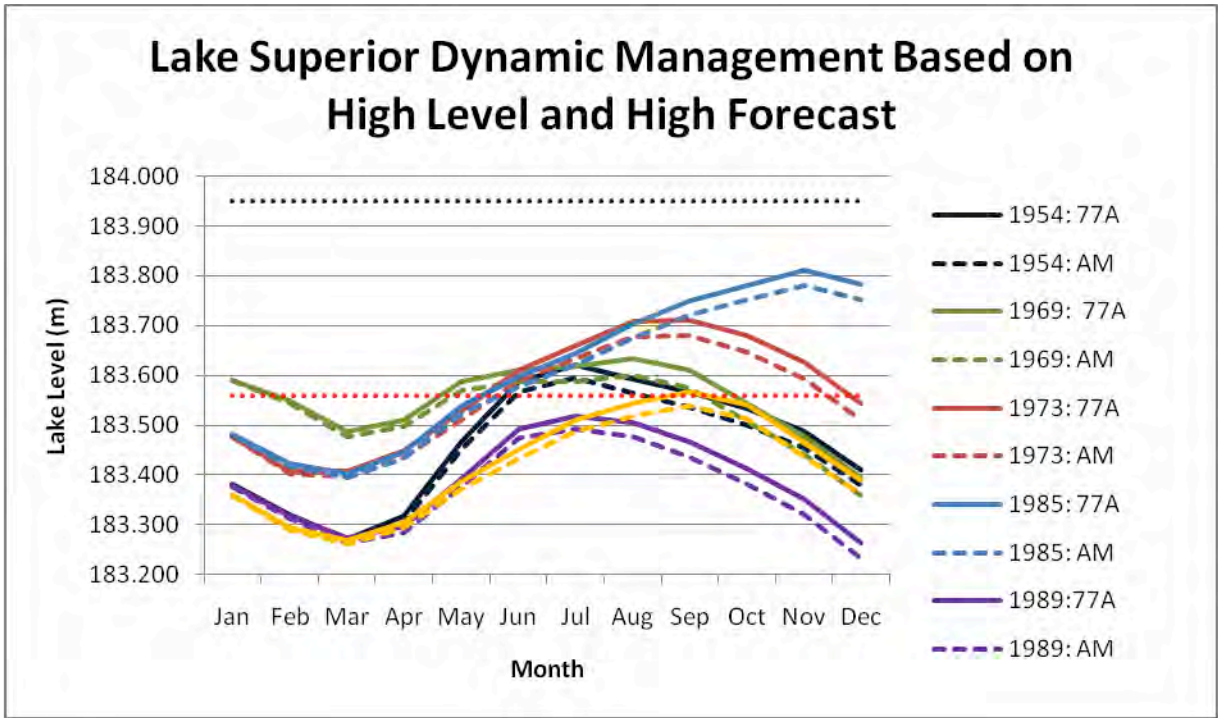


Figure B2. Results for a correct forecast based on 1969 showing the forecast-based release rule results in a lower high level and a shorter duration in coping zone B.

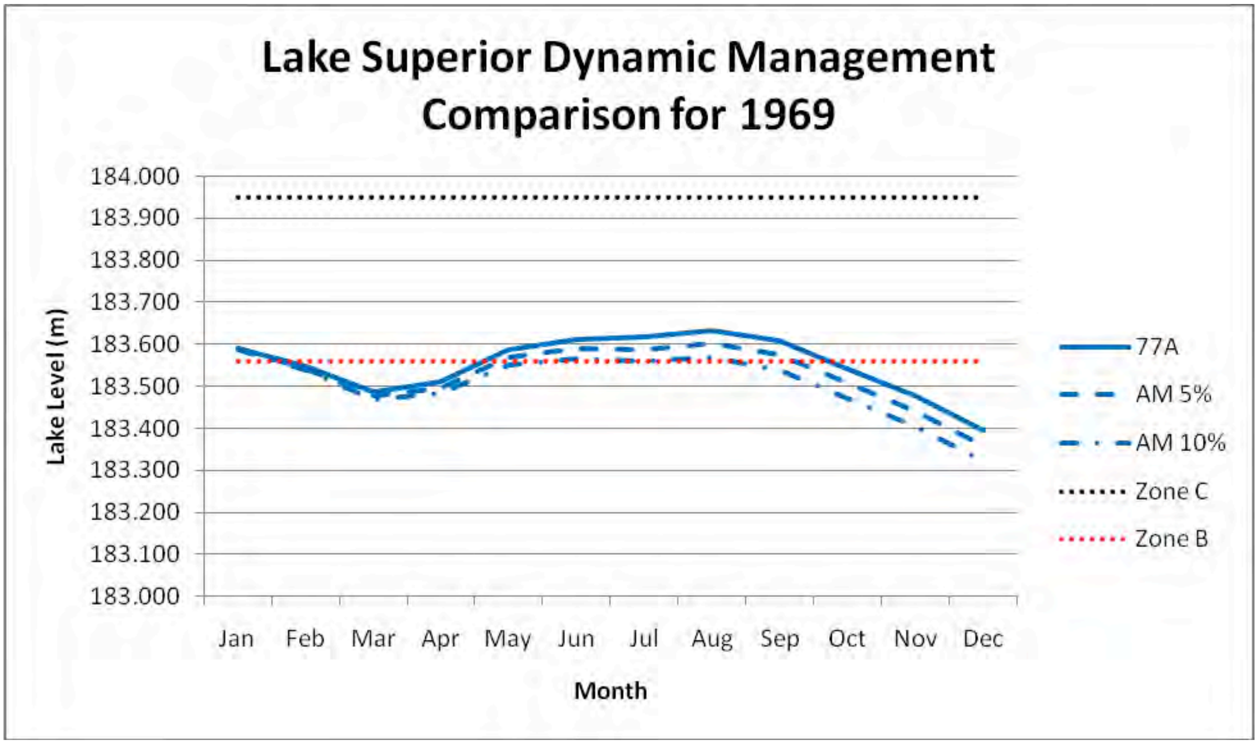


Figure B2.

Figure B3. Impact of a 5% reduction in 77A Lake Superior outflow from Jul through Jun based on a low year with a low forecast.

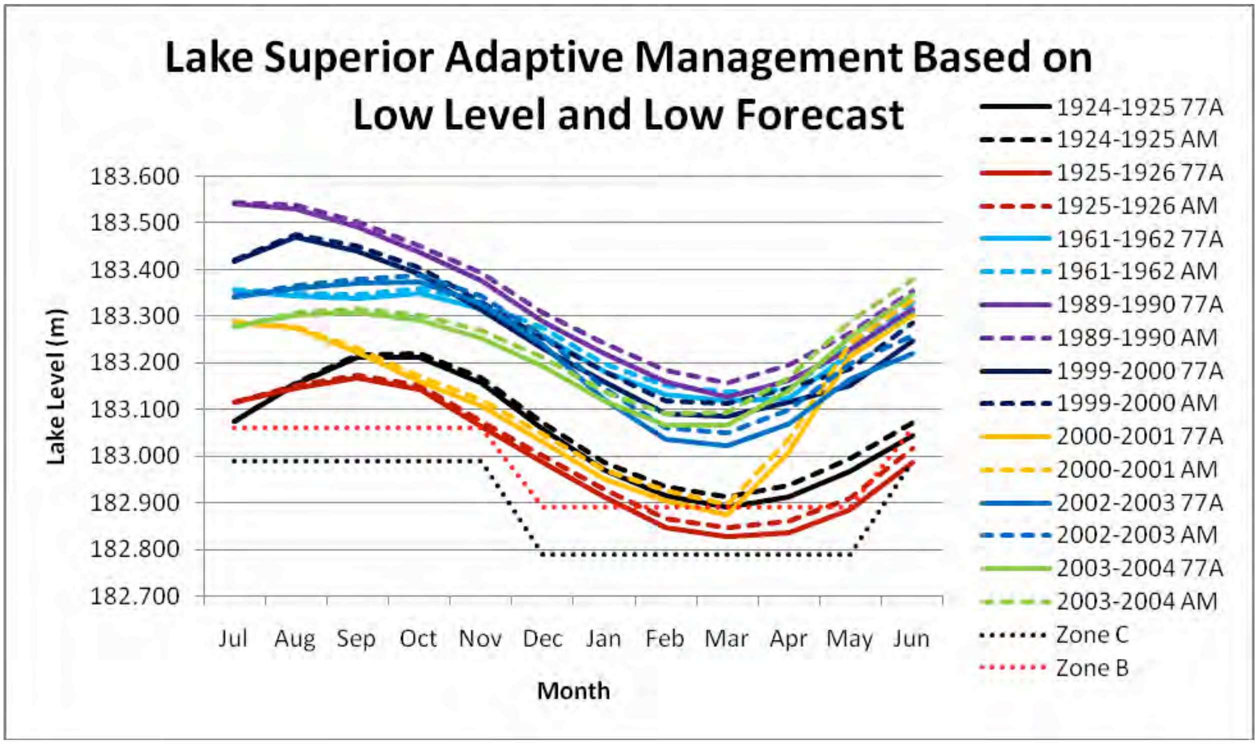


Figure B3.

Figure B4. Results for a correct forecast based on 1925-1926 showing the forecast-based release rule results in a higher low level and a shorter duration in coping zone B.

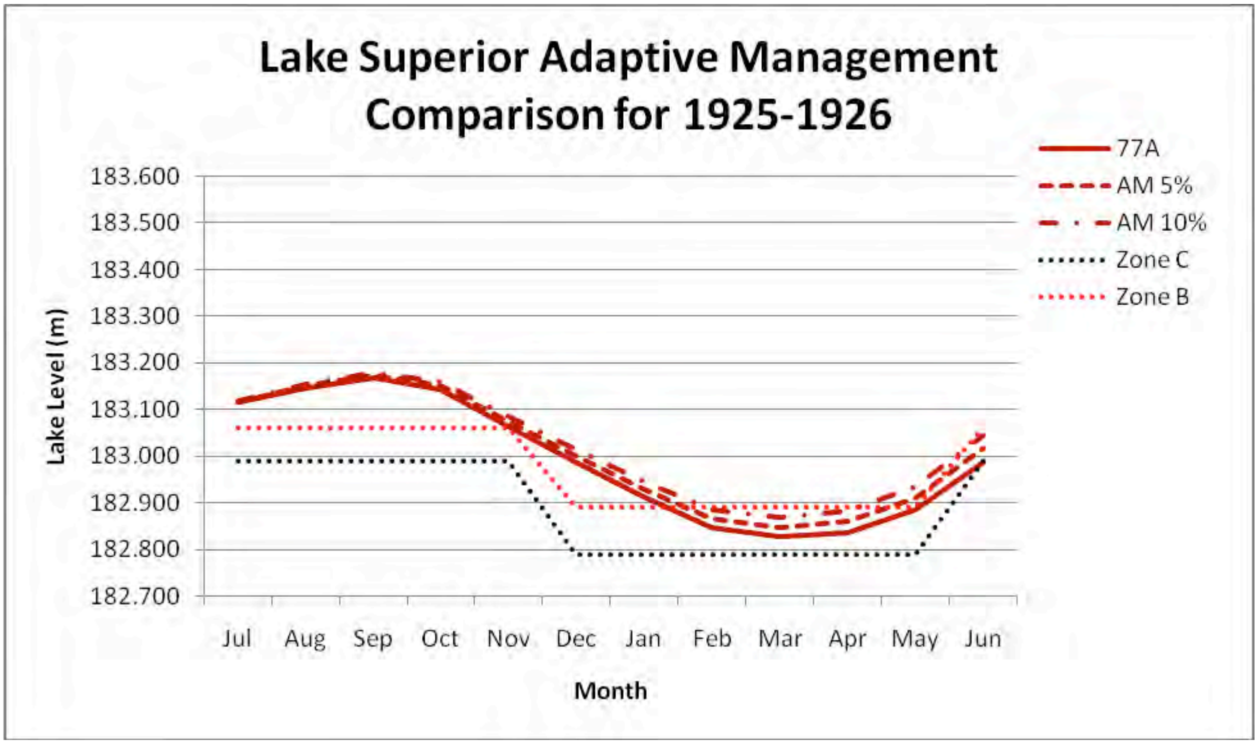


Figure B4.

Figure B5. The impact of false alarm when low NBS is predicted but high NBS is observed. The results show that there is no negative impact of the false alarm.

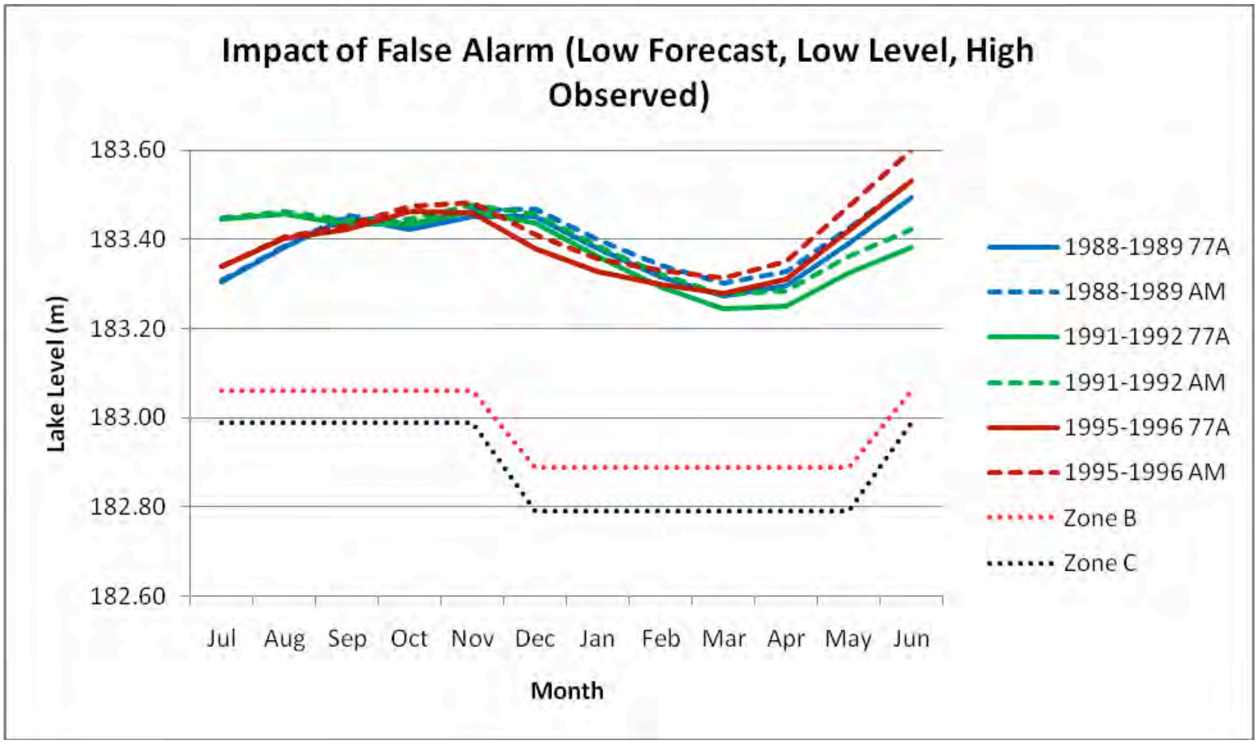


Figure B5.

Figure B6.

Figure B6. Comparison of Matlab 77A model compared with Fortran 77A.

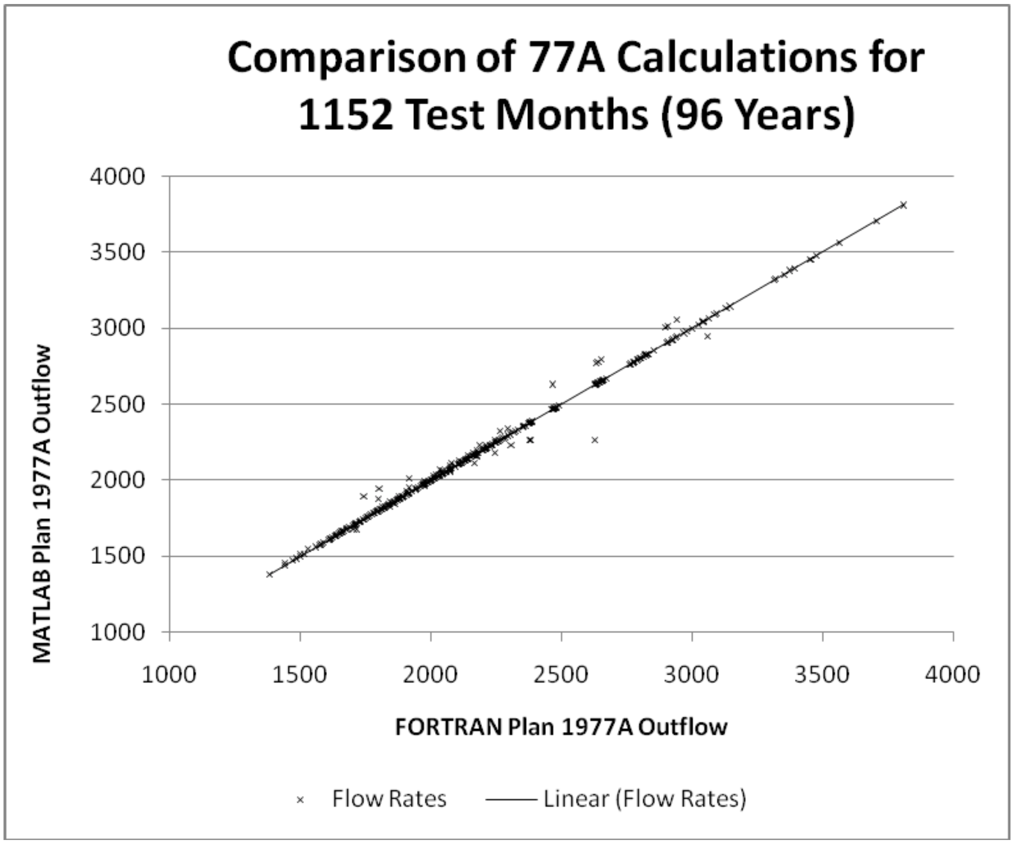


Figure B7. Results of dynamic 77a with forecast based releases and perfect forecasts based on the Vincent Fortin operation forecast period. Results show reduction in occurrences of high lake levels and high zone B and general compression of lake levels. Levels on Michigan-Huron were minimally impacted.

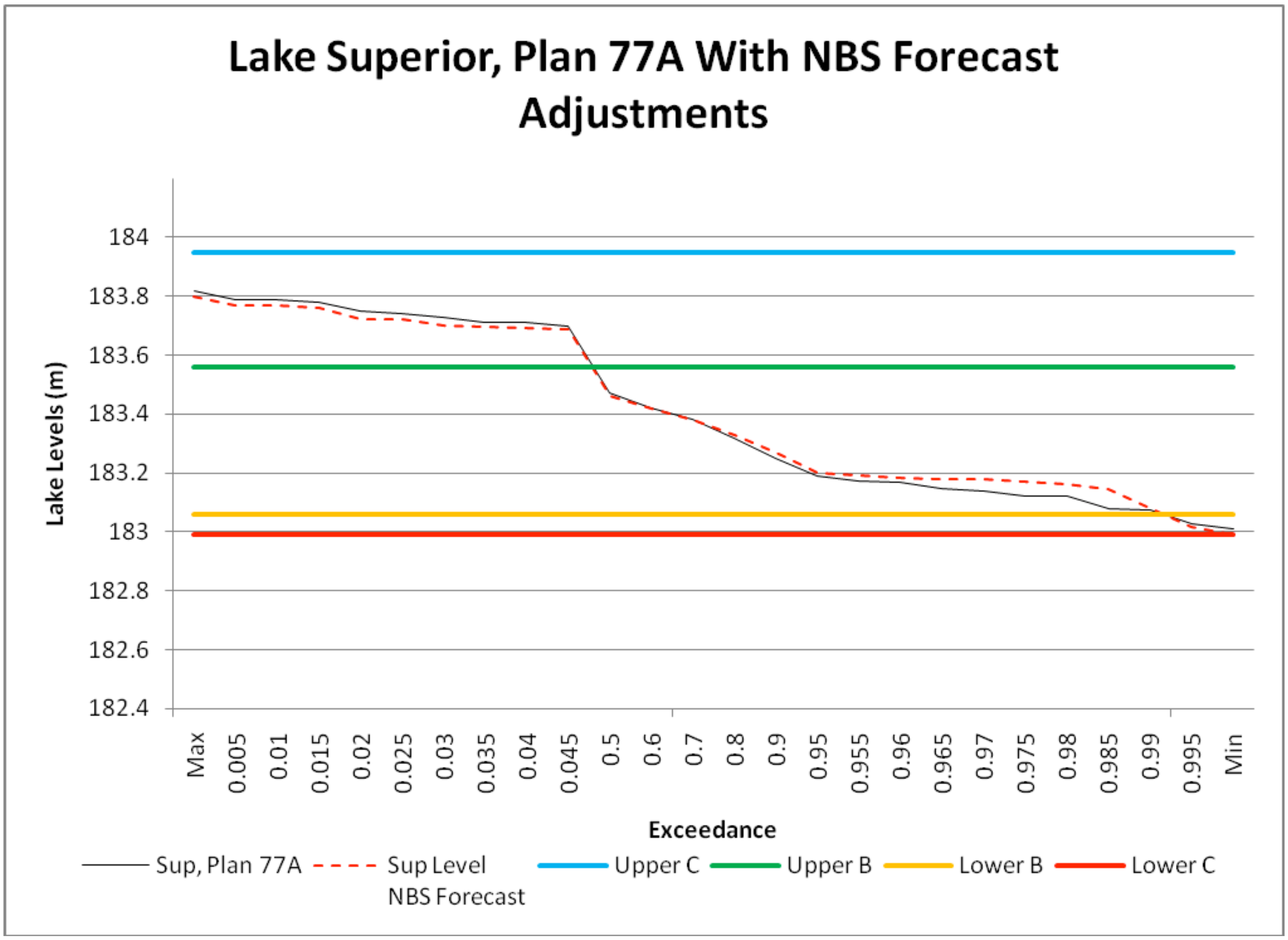


Figure C1. The current approach to management of Lake Superior: a regulation plan is enacted on the lake system which is subjected to exogenous factors resulting in the observed lake levels. The lake levels result in impacts that are primarily observed by stakeholders. Feedback is provided to the International Joint Commission often as complaints from the public.

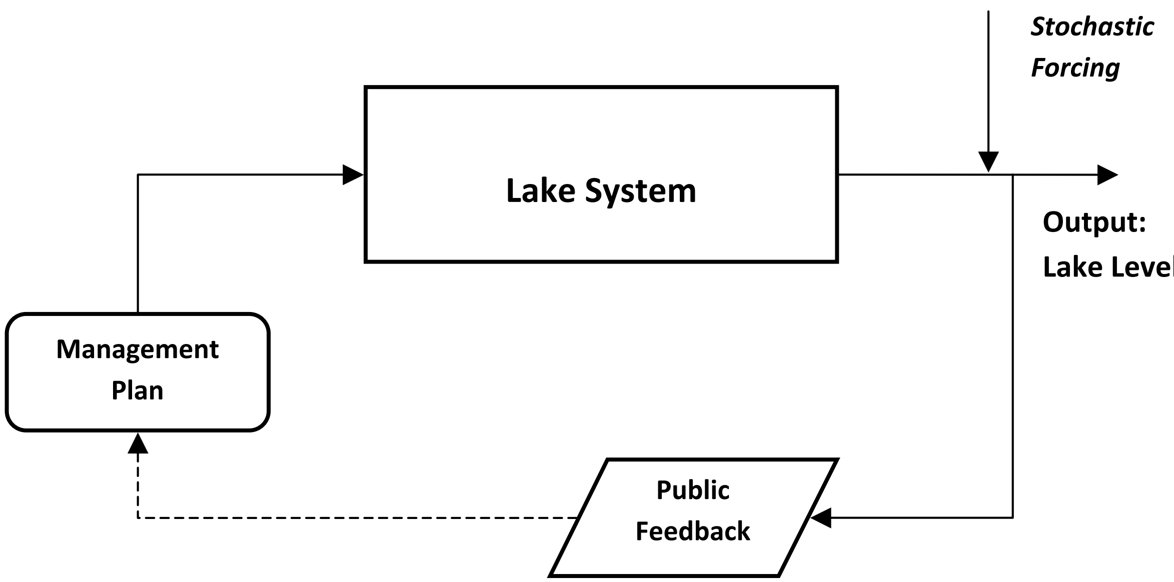
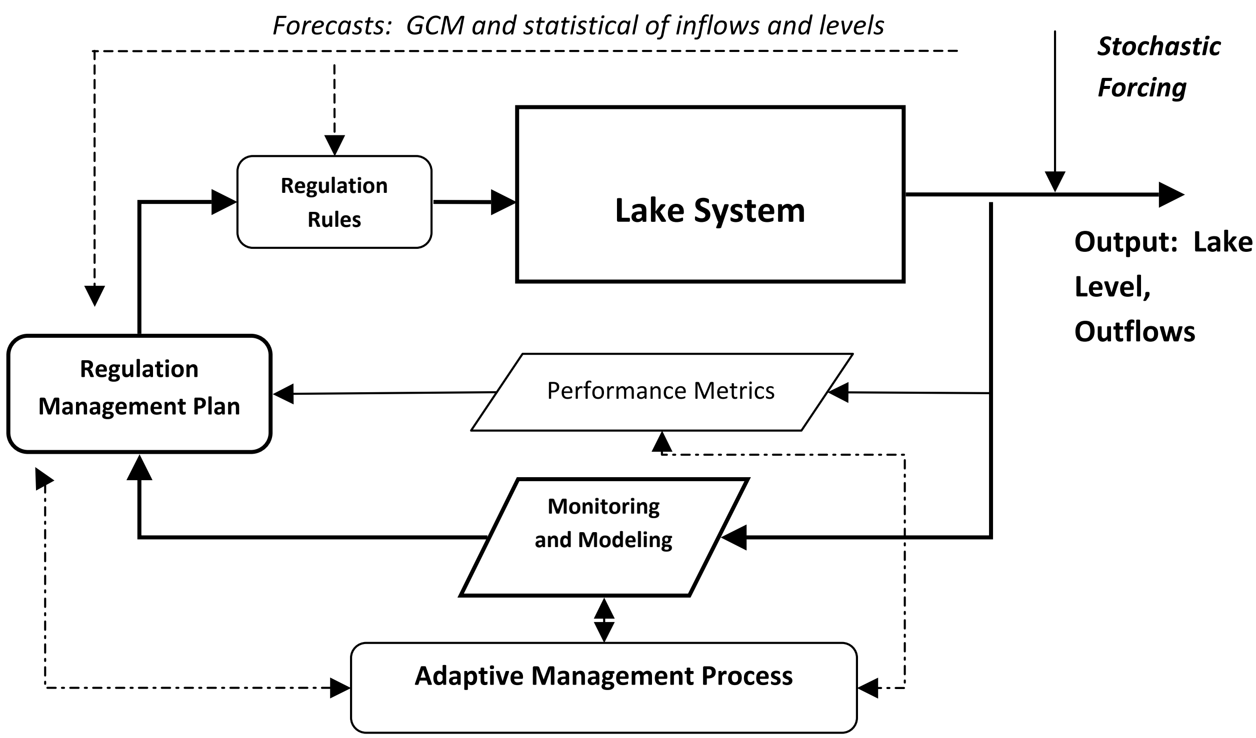


Figure C2. The proposed hierarchical adaptation strategy for the regulation of Lake Superior will utilize a dynamic regulation plan to select among several regulation approaches depending on the plan performance and the observed climate conditions. Feedback is provided via a monitoring program and ongoing evaluation of performance metrics related to coping zone status. At the highest level of the hierarchy, the performance of the dynamic regulation plan, including the performance metrics themselves and the monitoring program, is evaluated and when necessary, improved through an adaptive management process



Information Transfer Program Introduction

A significant portion of 104B funds retained at the Center supports the information transfer objective of 104B.

Our main information transfer tool is the Annual Water Resources Conference, initiated in 2003 by then Director David Reckhow. The conference provides an interdisciplinary forum for scientists, practitioners, and policy makers to discuss current critical water research, foster greater collaboration among scientists and practitioners, and strengthen the connection between research, education, and policy. Participants include researchers, stakeholders, and managers of water resources from academia, government, non-profits, and the private sector. The 8th Annual Water Resources Research Center Conference is described in the subsequent section. The Center publishes programs from all of our conferences on our website (http://www.umass.edu/tei/wrrc/WRRC2004/WRRCconferences.html).

The Center relies heavily upon the Internet for information transfer. Several of the Center's projects have significant Internet information transfer elements that are still in existence and utilized today. One of these funded through 104B is the Acid Rain Monitoring Project (ARM) (http://umatei.tei.umass.edu/ColdFusionProjects/AcidRainMonitoring/). Another information transfer relationship we are cultivating is with the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI), in order to make data more readily available in a "data clearinghouse" used by local, regional, and national organizations.

2010 Water Resources Conference

Basic Information

Title:	2010 Water Resources Conference
Project Number:	2010MA263B
Start Date:	3/1/2010
End Date:	2/28/2011
Funding Source:	104B
Congressional District:	1st
Research Category:	Not Applicable
Focus Category:	Water Quality, Water Quantity, Climatological Processes
Descriptors:	
Principal Investigators:	Paula Lynn Sturdevant-Rees, Marie-Francoise Hatte

Publications

There are no publications.

2010 Water Resources Conference

The Water Resources Research Center organized the eighth annual Water Resources Conference on the UMass Amherst campus on April 7, 2011. While the conference took place in April 2011, most of the work for this conference was accomplished in the reporting period. The Cooperative State Research, Education, and Extension Service New England Regional Program again cooperated in planning the conference. Four additional co-sponsors helped underwrite the cost of the conference.

Thirty-two posters were presented and there were 30 platform presentations in three concurrent sessions. The presentations were grouped into the following 9 sessions:

Climate Change and Stream Crossings in the Northeast
Monitoring and Detecting Harmful Algal Blooms
Nutrients Management in Water
Climate Change Adaptation Implementation Strategies
Fish Passage and Stream Continuity
Findings of the Connecticut River Targeted Watershed Initiative
Climate Change Adaptation and Decision Making
Tools for Water Management in the Connecticut River Basin
Stormwater and Low Impact Development

There were three Plenary Addresses at the beginning of the conference:

- "The University Perspective" by Rick Palmer, Professor and Department Head, Dept. of Civil and Environmental Engineering, UMass Amherst
- "The State of Massachusetts Perspective" by Vandana Rao, Assistant Director for Water Policy, Mass. Executive Office of Energy and Environmental Affairs
- "The New England Regional Perspective" by Jessica Cajigas, Environmental Analyst, New England Interstate Water Pollution Control Commission.

The Keynote Address was given by Dr. Richard Vogel, Professor of Civil and Environmental Engineering and Director of the Graduate Program in Water: Systems, Science and Society, Tufts University, on "Water Resources Planning in a Changing World."

181 people registered for the event, representing 14 colleges and universities, 23 companies, 15 governmental agencies, 4 non-profit organizations, and 13 municipalities.

Twenty-four students (from 6 different institutions) participated in the Best Student Poster Competition, evaluated by 14 judges. Liam Bevan of UMass Amherst Geosciences (and a WRIP research project awardee this fiscal year) and Barabara DeFlorio of UMass Amherst Veterinary & Animal Sciences tied for first place. Bevan's poster was entitled "Water Flux at Till/ Bedrock Interfaces in Central Massachusetts." DeFlorio's poster's title was: Optimizing Vegetative Filter Strips Treating Runoff from Turf."

Students supported by project

1 BS student in Mathematics at UMass Amherst
1 BS student in Chemical Engineering at UMass

USGS Summer Intern Program

None.

Student Support					
Category	Section 104 Base Grant	Section 104 NCGP Award	NIWR-USGS Internship	Supplemental Awards	Total
Undergraduate	9	0	0	0	9
Masters	5	1	0	0	6
Ph.D.	11	1	0	0	12
Post-Doc.	0	0	0	0	0
Total	25	2	0	0	27

Notable Awards and Achievements

As a result of research done in project 2010MA248B, The Corporate Toxics Information Project received the second year of a two-year grant from Food and Water Watch, a Washington, D.C.-based non-profit organization to further develop methodology for the assessment of population risk and environmental-justice disparities from toxic industrial water pollution reported in the U.S. EPA's Toxics Release Inventory and the Risk Screening Environmental Indicators model.

As a result of research done in project 2010MA248B, James K. Boyce (PI) and Michael Ash (co-PI) submitted a successful proposal to the National Science Foundation in collaboration with researchers at the University of Southern California and the University of Michigan to continue research on environmental justice with the Risk Screening Environmental Indicators geographic microdata.

As a result of research done in project 2010MA248B, Michael Ash (PI) submitted a proposal and received an allocation on TeraGrid, National Science Foundation's effort to build and deploy the world's largest distributed infrastructure for open scientific research, to use TeraGrid resources to manage the databases for the project.

Based on the findings of research project 2010MA231B, Yu submitted a NSF proposal and it was successfully funded. Qian Yu (PI) and Co-PI: Anna Liu collaborated with Yong Tian and Robert Chen at UMass-Boston. The collaborative research is entitled: Modeling DOC dynamics from landscapes to coasts: hydrological connectivity and estuary processes, NSF Collaboration in Mathematical Geosciences (CMG), #1025547. The awarded funding: $517,987 ($329,346 on Amherst Campus), Sept 2010 - Aug 2013.

Based on the findings of research project 2009MA199B, Suzanne LePage was awarded 2nd Place for Poster entitled "An Investigation into the Water Quality Impacts of a Green Roof", which was presented at the Seventh Annual Water Resources Research Conference Poster Contest on April 8, 2010.

Publications from Prior Years

1. 2003MA19G ("A Regional Approach to Conceptualizing Fractured-Rock Aquifer Systems for Groundwater Management") - Articles in Refereed Scientific Journals - Manda, A.K. and S.B. Mabee, 2010. Comparison of three fracture sampling methods for layered rocks, International Journal of Rock Mechanics and Mining Sciences, v.47, no.2, pp.218-226.
2. 2003MA19G ("A Regional Approach to Conceptualizing Fractured-Rock Aquifer Systems for Groundwater Management") - Articles in Refereed Scientific Journals - Manda, A.K., S.B. Mabee, D.F. Boutt and M.L. Cooke. Effects of fracture configurations and properties on the hydraulic properties of three-dimensional networks, submitted to Water Resources Research, "in review".
3. 2003MA19G ("A Regional Approach to Conceptualizing Fractured-Rock Aquifer Systems for Groundwater Management") - Conference Proceedings - Manda, A.K. and S.B. Mabee. 2010. Using GIS and discrete fracture network models to compare three fracture sampling techniques, Geological Society of America, Northeastern/Southeastern Section Meeting, Abstracts with Programs, 42, no.1, p.168.
4. 2008MA135B ("Toxicity of carbon nanotubes to the activated sludge process: protective ability of extracellular polymeric substances") - Articles in Refereed Scientific Journals - Luongo, L. and Zhang, X. (2010) Toxicity of carbon nanotubes to the activated sludge process. Journal of Hazardous Materials, Volume 178, Issues 1-3, 15 June 2010, Pages 356-362