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South Carolina Water Resources Conference (SCWRC)
The purpose of the SCWRC is to provide an integrated forum for discussion of water policies, research projects and water management in order to prepare for and meet the growing challenge of providing water resources to sustain and grow South Carolina’s economy, while preserving our natural resources. The conference is a biennial event, held in even-numbered years. (www.scwaterconference.org)

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Introduction
Timothy J. Callahan, Ph.D.
Journal of South Carolina Water Resources Editor

“Managers are not confronted with problems that are independent of each other, but with dynamic situations that consist of complex systems of changing problems that interact with each other. I call such situations messes. Problems are extracted from messes by analysis. Managers do not solve problems, they manage messes.”


In our day-to-day work, ‘managing messes’ seems like an apt description of how we react to what needs to be done. I started this note last July when South Carolina was in the midst of a mild yet potentially deepening drought. The early October storm turned things upside down, disrupting the state, leading to the deaths of a dozen people, and producing damage estimated to be in the billions of dollars. As of November 10, Charleston has recorded 71 inches of rain and should surpass the record yearly rainfall of 73 inches, set in 1964.

As we work on recovery amid continued wet conditions, fortunately the state of the state of water planning is improving, thanks to efforts by the Department of Natural Resources and multiple partners. The Division of Land, Water, and Conservation at DNR has been tasked with building on existing knowledge from its multiple partners and collaborators to develop a State Water Plan (http://www.dnr.sc.gov/water/waterplan/surfacewater.html) that will take into account surface water availability for the eight major river basins in the state. This large task requires collaboration across multiple groups and integrating data at a wide range of scales. With this guidance, local municipalities and state agencies across disciplines will be armed with information to plan for a rapidly-growing population and to make data-based management decisions that balance economic needs with the protection of our water resources. As Jeff Allen of the South Carolina Water Resources Center describes in the Foreword, large collaborations like this are challenging but sorely needed in order to inform stakeholders.

The articles chosen for our second volume address the environmental and economic value of our water resources, from understanding the changing river flows, the water quality threats in the different river basins, and how climate and weather patterns influence water availability across river basins from the individual ecosystem to regional scales of influence.

Because South Carolina and the Southeast U.S. is blessed with rich resources - natural, social, historical, and cultural - we hope this second volume of articles will be informative to water resource scientists, managers, academics, and other stakeholders. Fortunately, water is not something South Carolinians take for granted. Because we have such a strong connection to nature, our waters are something with which we all have a vested interest in.
Reflecting on 2015 from a water resources perspective, especially in the state of South Carolina, brings many thoughts to mind from the widest array of water events the state has witnessed in quite some time. From a physical standpoint, starting the year with normal rainfall, moving into a drought, and then faced with catastrophic flooding within just a few months was unprecedented. Indeed, from my recollection there has never been an instance where counties in the state were qualified to receive disaster relief for farmers for drought, then floods, and all before the growing season ended. The flooding was the result of yet another “perfect storm” situation where an Atlantic hurricane and an inland low-pressure system squeezed together and for several days pumped unimaginable amounts of rainfall statewide. The sad result was loss of life and a crippling impact to roads, bridges and property totaling in the billions of dollars.

When events like these occur, the scientific and water management community get asked questions, and those asking the questions often demand answers. Sometimes the answers are easy, but most often they are not, usually complex and intertwined with economic, social, political, cultural and environmental issues. And as usually is the case, the science takes too much time to get from the laboratory to water managers and citizenry.

In my role as director of the S.C. Water Resources Center at Clemson University, I enjoy the privilege of belonging to a national network of water resource research institutes called the National Institutes for Water Resources (NIWR). This network provides a conduit of shared water science across all fifty states as well as U.S. territories in the Pacific and Atlantic Oceans. Fifteen years ago, Dr. Doug Ward (then NIWR president) shared with the NIWR membership a communication he had with Dr. Doug James regarding the challenges of water scientists and resource managers. As much as things have changed since the year 2000, these words ring as true to me now as they did then.

As I look at the grand challenges that you list as facing water resource managers, I find an oft-repeated list that covers so many things as to scatter any effort so much as to make it unmanageable. The underlying issue would seem to be what can ‘science’ do about it all? What can you, the university community, do to give science the focus needed to make people ‘feel’ important contributions? People are tired of our saying fund more studies on topics that seem to mesh with a list of priority problems. What sort of focus is then needed? Some of the basic issues to address are

- The ‘hard’ sciences generally study ‘water’ at small scales, using laboratories or small field plots. Social sciences deal with much larger communities. Climate people look at a worldwide grid at say 50km spacings. How can we bring studies across scales for meaningful connects?

- Science generally becomes quite specific on the way things are. Water managers operate under a great deal of uncertainty and among people who disagree strongly on what they want. Differences of opinion become arguments to support different actions. How can we be constructive in coping with uncertainty in trying to change water management policies that are rooted in long tradition in ways that have created powerful vested interests? Issues tend to come down to relative power among the vested interests rather than on what scientists say.

- We do not have the data gathering network and the educational system needed to move water science forward on the frontiers that are important. Point measurements can go only so far in understanding the watershed scale. Education in hydrology can go only so far in engaging much needed expertise in ecology, information science, etc.

The point of this overview is not to provide thorough coverage, but a framework that will help define relationships and help the (hydrologic) community synthesize results. A working integrating framework can be found by picturing our understanding of the hydrologic cycle as existing at the center of an expanding science with multiple frontiers where the different studies are working at different frontiers. The probing at each boundary gathers information and expands networks that ‘heartland’ hydrologists can ‘digest’ and use.

And so, you may ask, have we made any progress in 15 years? I would argue that while many of the issues still remain - especially the gaps between water science, policy, and management - we have made strides in understanding the complexities of these issues and how we may need to incorporate the science in the implementation of water management. The articles presented in this issue of the Journal of S.C. Water Resources will help to add to the foundation of knowledge, as well as building the bridge between those gaps we all know too well.
South Carolina’s Climate Report Card:  
The Influence of the El Niño Southern Oscillation Cold and Warm Event Cycles on South Carolina’s Seasonal Precipitation

Hope Mizzell and Jennifer Simmons

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Abstract. This study was driven by the need to better understand variations in South Carolina’s seasonal precipitation. Numerous weather-sensitive sectors such as agriculture and water resource management are impacted by the seasonal variability and distribution of precipitation. Studies have shown that El Niño-Southern Oscillation (ENSO) has varying effects on seasonal temperature and precipitation across the United States.

The purpose of this study was to determine the relative influence of ENSO cold and warm event cycles on interannual variations of South Carolina’s seasonal precipitation (1950-2015). The relationship between seasonal precipitation departures from normal and the average Multivariate ENSO Index was analyzed. Seasonal precipitation totals for each of South Carolina’s seven climate divisions and for three key city locations (Greenville-Spartanburg Airport, Columbia Airport, and Charleston Downtown) were examined.

Results from the study indicate that the magnitude, seasonal variation, and consistency of the precipitation response to ENSO vary spatially and from episode to episode. Winter precipitation tends to be enhanced during the warm phase (El Niño) and reduced during the cold phase (La Niña). There is a less consistent signal during fall and no evident connection between ENSO and spring and summer precipitation.

INTRODUCTION

South Carolina has a mild climate and, in normal years, adequate precipitation. While there is no distinct wet or dry season in South Carolina, average precipitation does vary throughout the year (Figure 1). Summer precipitation is normally the greatest, but the most variable, occurring mostly in connection with localized showers, sea breeze convection, and diurnal thunderstorms. Fall is historically the driest season. Any heavy precipitation during this period is likely a result of tropical features, early winter storms, or stalled boundaries. Precipitation during winter and spring occurs mostly in connection with frontal passages. The seasons are climatologically defined as winter (December-February), spring (March-May), summer (June-August), and fall (September-November).

South Carolina’s precipitation varies geographically. Annual precipitation in South Carolina is heaviest in the Northwest and Mountain regions, averaging between 70 to 80 inches at the highest elevations. The driest portion of the state is the central region, where annual totals average between 39 to 42 inches.

South Carolina’s seasonal weather varies from localized events to larger-scale, multi-year events. The year-to-year variations in the weather patterns are often associated with changes in wind, pressure, storm tracks, and the jet stream. These weather pattern changes are often linked to large-scale shifts or oscillations of the ocean-atmosphere system such as the El Niño-Southern Oscillation (ENSO) in the Equatorial Pacific (Climate Prediction Center, 2015).

Most research has focused on the relationship between precipitation and ENSO at global and regional scales (Barlow et al., 2001; Dai et al., 1997; Groisman and Easterling, 1994; Ropelewski and Halpert, 1986). While some of this research
includes South Carolina precipitation, the work is broader in scope and not focused on documenting and detecting localized changes in seasonal precipitation patterns due to fluctuations of the ENSO cycle.

PROJECT OBJECTIVES

The objectives for this study are: (1) develop a time series of seasonal precipitation (1950-2014) for each of South Carolina's seven climate divisions and three key city locations (Greenville-Spartanburg Airport, Columbia Airport, and Charleston Downtown); (2) utilize the Multivariate ENSO Index to classify each season as Neutral, Strong La Niña, Moderate to Weak La Niña, Moderate to Weak El Niño, or Strong El Niño; and (3) examine how seasonal precipitation in South Carolina responds to the varying strengths of the warm and cold ENSO episodes.

PROJECT BACKGROUND AND DESCRIPTION

ENSO is an important coupled ocean-atmosphere phenomenon in the equatorial Pacific region. Through complex interactions between the oceans and the atmosphere, the ENSO can directly and indirectly have an impact around the world. El Niño and La Niña represent opposite phases in this naturally occurring climate cycle (Climate Prediction Center, 2015; University Corporation for Atmospheric Research, 2015). They are associated with opposite extremes in sea-surface temperature departures across the central and east-central equatorial Pacific, and with opposite influences on convective precipitation, surface air pressure, and atmospheric circulation. El Niño refers to the above-average sea-surface temperatures that periodically develop across the east-central equatorial Pacific. It represents the warm phase of the ENSO cycle. La Niña refers to the periodic cooling of sea-surface temperatures across the east-central equatorial Pacific. It represents the cold phase of the ENSO cycle. ENSO-neutral refers to those periods when neither El Niño nor La Niña is present. During ENSO-neutral periods, the ocean temperatures, tropical precipitation patterns, and atmospheric winds over the equatorial Pacific Ocean are near the long-term average.

El Niño and La Niña are typically strongest during winter and spring because the equatorial Pacific sea-surface temperatures are normally warmest at this time of the year. However, there is considerable variation in the intensity and duration of each ENSO cycle. Scientists from the National Oceanic and Atmospheric Administration and other agencies use a variety of tools and techniques to monitor and forecast changes in the Pacific Ocean. The Multivariate ENSO Index (MEI) is one method used to monitor the ENSO based on six main variables over the tropical Pacific: sea-level pressure, zonal and meridional components of the surface wind, sea surface temperature, surface air temperature, and total cloud cover fraction of the sky (Wolter and Timlin, 2011). MEI is calculated as the first unrotated Principal Component of all six observed fields combined. Positive MEI values are related to warm phase or El Niño events and negative values with cool phase or La Niña events.

ENSO events have varying effects on temperature and precipitation across the United States. There is research that documents the impact of ENSO in the Southeast U.S. with El Niño typically associated with wet and cool winters and the La Niña typically associated with dry and warm winters (Ropelewski and Halpert 1986; Schmidt et al., 2001). El Niño and La Nina produce extensive yet differing redistributions of precipitation across the tropical Pacific as well as extensive teleconnections that affect synoptic weather patterns extending across the continental United States. Since much of the research is broad in scope or specific to other Southeast states, this project will focus on documenting and detecting localized changes in South Carolina’s seasonal precipitation patterns due to fluctuations of the ENSO cycle.

Climate divisional data were utilized for this project. Each U.S. state is subdivided into climatic divisions with boundaries that are delineated partially on climatic conditions, but also reflect county lines, drainage basins, or major crops. The area of each of the U.S. contiguous states has been divided into between one and 10 climate divisions (National Centers for Environmental Information, 2015). South Carolina has seven climate divisions. (Figure 2).

Climate division data is provided on a monthly basis by the National Centers for Environmental Information. The climate divisional dataset consists of monthly average temperature, precipitation, heating/cooling degree days, and various drought indices since 1895. The data are derived from area-weighted averages of 5km by 5km grid-point estimates interpolated from station data (Vose et al., 2014). The Global Historical Climatology Network is the source of station data. The number of stations utilized in each month’s analysis varies due to station additions, closures, or station removals due to data errors. Despite some weaknesses, the

![Figure 2. South Carolina Climate Divisions (MTN=Mounds, NW=Northwest, NC=North Central, NE=Northeast, WC=West Central, C=Central, S=Southern).]
divisional dataset has proven to be useful for putting anomalous meso-scale and macroscale weather events into historical perspective (Guttman and Quayle, 1996).

**METHODOLOGY**

A time series of seasonal precipitation (1950-2014) for South Carolina’s seven climate divisions and three key city locations was developed. Greenville-Spartanburg Airport, Columbia Airport, and Charleston Downtown were selected as the city locations. Greenville-Spartanburg’s period of record was shorter, beginning in 1963. Percent of normal precipitation values for each season were computed using a base period of 1901-2000 for each climate division and for each station’s period of record. The bimonthly MEI was utilized to classify each season as Neutral, Strong La Niña, Moderate to Weak La Niña, Moderate to Weak El Niño, or Strong El Niño. For each season, the three bimonthly values were averaged (e.g., November/December, December/January, and January/February for Winter). Once the average seasonal MEI was obtained, values greater than 1.0 were designated as El Niño and less than -1.0 as La Niña events. All values between -1.0 and +1.0 were considered ENOS neutral and discarded. The values greater than 1.0 and less than -1.0 were then separated based on percentiles. Values less than or greater than the 25th percentile were classified as strong events. Values between the 25th and 75th percentiles were considered Moderate to Weak ENOS events.

Seasonal precipitation totals for each of the seven South Carolina climate divisions and for three key city locations (Greenville-Spartanburg Airport, Columbia Airport, and Charleston Downtown) were examined. Percent of normal precipitation for each season was analyzed and graphed with respect to the MEI ENOS classification. The percent of normal precipitation values were then averaged for all seasons in each ENOS phase and presented by season and by climate division.

**RESULTS**

The effect of ENOS on precipitation in South Carolina is not uniform. There appear to be seasonal precipitation differences between upstate, central, and coastal portions of the state. Table 1 displays the seasonal percent of normal precipitation values averaged for each type of ENOS phase. Figures 3 and 4 show the individual winter percent of normal values for each ENOS phase for six of the seven climate divisions. The graph for the Mountain Division was not displayed since the division covers a small geographic area.

The most notable precipitation signal across South Carolina occurs during winter. There is an overall negative winter precipitation anomaly in all seven climate divisions for Moderate/Weak and Strong La Niña events (Table 1). Likewise there is an overall positive winter precipitation anomaly in all seven climate divisions for Strong El Niño events and for six out of the seven climate divisions for Moderate/Weak El Niño events. Another notable result is that there is a 23% to 56% increase in precipitation during Strong El Niño winters compared to Strong La Niña winters depending on climate division.

The winter El Niño signal is the highest for the Southern climate division with 39% higher than average precipitation during Strong El Niño events and 17% higher than average during Moderate/Weak El Niños. Figures 3 and 4 show all ENOS winters and the percent of normal precipitation that occurred in each climate division. The graphs display the range of percent of normal precipitation for each ENOS phase (driest winter to the wettest). For example, for the Southern climate division (Figure 4), during Strong El Niño winters, the precipitation departures ranged from 1% below normal to 117% above normal. Three out of the six Strong El Niño winters received greater than 30% of normal precipitation.

In order to evaluate whether the precipitation was above, below, or normal during each season, a +/-30% of normal criteria was established. Precipitation was considered normal for each season if the average departure from normal was between +30% above normal and -30% below normal. There appears to be a clear ENOS influence on winter precipitation statewide even though the averages were less for some of the climate divisions. At least 16 out of 19 La Niña winters (Weak to Strong) experienced normal to below normal precipitation for all climate divisions. The only La Niña winters that recorded above normal precipitation were 1961-1962, 1973-1974, and 1974-1975. El Niño’s influence on winter precipitation was equally as substantial, but with opposite results as expected. At least 13 out of 15 El Niño (Weak to Strong) winters for all climate divisions experienced normal to above normal precipitation. The only El Niño winters that were dry (< -30% of normal precipitation) were 1979-1980 and 1987-1988.

Results from the key cities reinforced results from the climate divisions. The period of record analyzed for Greenville-Spartanburg is 1963-2014. The study period for Columbia and Charleston is 1950-2014. Charleston received an average 47% increase in winter precipitation during Strong El Niño winters and a 15% increase during Moderate to Weak El Niño winters. Columbia received an average 27% increase in winter precipitation during Strong El Niño winters and an 18% increase during Moderate to Weak El Niño winters. The influence of El Niño on precipitation in Greenville-Spartanburg was less obvious with only a 17% average increase during Strong El Niño winters and 9% average increase during Weak to Moderate El Niño winters. During La Niña winters, average precipitation was reduced by 18% to 20% in Charleston, by 6% to 12% in Greenville-Spartanburg, and less than 9% in Columbia.

The strength of the ENOS did not seem to be a factor in whether or not the signal was consistent. For instance, not all Strong El Niño winters had above normal precipitation nor did all Strong La Niña winters record below normal precipitation. Several Moderate to Weak El Niño winters recorded higher precipitation totals compared to Strong El
Table 1. Seasonal percent of normal precipitation values averaged for each ENSO phase, 1950-2014. Each season was classified as one of the following ENSO phases: Strong La Niña, Moderate/Weak La Niña, Moderate/Weak El Niño, or Strong El Niño. The seasonal percent of normal precipitation was then averaged for each ENSO phase. The years included in each ENSO phase will vary depending on season. For example, Strong El Niño Winters include 1957-58, 1972-1973, 1982-83, 1991-92, 1997-98 and Strong El Niño Summers include 1965, 1972, 1982, 1983, 1997.

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Figure 3. Winter percent of normal precipitation values for each ENSO phase for the Northwest, North Central and West Central Climate Division. Displays the range of percent of normal precipitation for each ENSO phase (driest to wettest).
Figure 4. Winter percent of normal precipitation values for each ENSO phase for the Central, Northeast and Southern Climate Division. Displays the range of percent of normal precipitation for each ENSO phase (driest to wettest).
Figure 5. Fall percent of normal precipitation values for each ENSO phase for Greenville-Spartanburg Airport, Columbia Airport and Charleston Downtown. Displays the range of percent of normal precipitation for each ENSO phase (driest to wettest).
Niño events. Likewise several Moderate to Weak La Niña winters were drier than some of the stronger La Niña winters.

While ENSO’s influence on winter precipitation had the most consistent signal, Moderate to Weak El Niño episodes appear to enhance fall precipitation in the Mountain, Northwest, North Central, and West Central climate divisions. The signal was less obvious during Strong El Niño events, except in the Northeast and Southern climate divisions where Strong El Niño episodes had a more apparent signal. Since several of the above normal fall seasonal totals included precipitation from tropical systems, additional investigation is needed to determine the influence of the tropical precipitation on the departures compared to an overall El Niño induced pattern change. La Niña does not appear to have a clear signal on fall season precipitation.

Figure 5 displays the fall ENSO events for Greenville-Spartanburg, Columbia, and Charleston. All three stations experienced normal to above normal precipitation during most of the fall season Moderate to Weak El Niño events. Strong El Niño events during fall appear to produce normal to above normal precipitation for Columbia and Charleston, but normal to below normal precipitation for Greenville-Spartanburg. La Niña’s influence on fall precipitation was not consistent.

CONCLUSIONS

The magnitude, seasonal deviations, and consistency of the precipitation response to ENSO vary spatially and from episode to episode in South Carolina. Results reveal that ENSO’s impact on South Carolina’s climate is most notable during winter. The effect of ENSO on precipitation is not uniform. There are seasonal precipitation differences between upstate, central, and coastal portions of the state. There is a negative winter precipitation anomaly during La Niña events and a positive winter precipitation anomaly during El Niño events.

The winter El Niño signal is the highest for the Southern climate division with 39% higher than average precipitation during Strong El Niño events. Charleston experienced, on average, a 47% increase in winter precipitation during Strong El Niño winters. La Niña episodes had the opposite impact, reducing winter precipitation with a consistent influence statewide. The strength of the ENSO did not always control the precipitation signal (i.e. not all Strong ENSO events were wetter or drier than Moderate or Weak ENSO Events).

ENSO’s influence on fall precipitation is less obvious. El Niño seems to enhance precipitation, but it varies by climate division and by strength of the ENSO event. La Niña does not appear to have a clear signal on fall precipitation. Additional investigation is needed to determine whether precipitation from tropical systems is influencing the departures rather than an overall El Niño-induced pattern change.

El Niño and La Niña are important drivers of the natural variability of regional, U.S. and global climate. ENSO provides some predictable effects to weather patterns. However, every ENSO event differs in magnitude and in duration. Additional research is needed since ENSO may be masked by other weather and climate signals. Single extreme events can alter the overall signal or trend. Future research should expand the investigation to include ENSO’s influence on South Carolina temperature. Future analysis should include additional climate patterns that exert important influences on regional climates such as the Pacific Decadal Oscillation and North Atlantic Oscillation.

LITERATURE CITED


Use of a Volunteer Monitoring Program to Assess Water Quality in a TMDL Watershed Utilized for Recreational Use, Pickens County, South Carolina

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Abstract. Municipalities, regulatory agencies, and resource advocacy organizations are often tasked with the enormous responsibility of monitoring water quality and implementing management strategies for vast areas within their jurisdictions. A potential means for addressing the resulting sampling shortfall is the use of volunteer monitoring programs. The project reported herein demonstrates the use of QA/QC protocols developed by Georgia Adopt-a-Stream (AAS) to monitor water quality issues for Twelve Mile Creek located in Pickens County, SC. The Twelve Mile watershed has a storied past as a U.S. EPA Superfund site due to industrial PCB contamination. Recent mitigation efforts involving the removal of two concrete dams have resulted in the creation of a nearly two-mile section of whitewater which is used by the local paddling community and is being marketed as a recreational destination. However, the Twelve Mile watershed also has a TMDL Implementation Plan in place due to chronic impairment from fecal coliform bacteria. Using sampling and monitoring methods developed by AAS, this project determined that E. coli levels increase significantly during high-flow discharges due to storm events and there were no significant differences in E. coli concentrations among sites located along a longitudinal gradient following the proposed Twelve Mile Creek Blueway. Ironically, the popularity of this area for paddling increases during periods of high discharge, thus recreational users are likely exposed to unhealthy levels of bacteria under these “desirable” conditions.

Volunteer monitoring programs like AAS exhibit tremendous potential for gathering water quality data that may not be possible if left solely up to other stakeholders. Appropriately managed volunteer monitoring programs have the capability to increase the resolution, reach, and efficiency of existing monitoring programs and serve to benefit a variety of stakeholders.

INTRODUCTION

The Clean Water Act of 1972, and its numerous revisions, attempts to address surface water pollution from a variety of directions including permitting and monitoring at federal, state, and local levels. However, efforts of regulatory agencies are limited in that it is impossible to monitor each and every waterway, tributary, and headwater stream in a given watershed. One way to address this monitoring shortfall is to make use of volunteer water quality monitoring programs (Bonney et al., 2009; Cohn, 2008; Conrad & Hilchey, 2011; Overdevest et al., 2004; Silvertown, 2009). Effective volunteer water quality monitoring programs are desirable in that they have the potential to inexpensively and efficiently gather large amounts of data with a higher frequency and over a larger geographic area than regulatory agencies are able to do.

As an example, the Adopt-a-Stream Foundation was established in 1985 with the goal of encouraging water quality awareness by promoting watershed education and engaging citizens in a volunteer monitoring program utilizing their local waterways. Specifically, Georgia Adopt-a-Stream (hereto after referred to as AAS), funded through a federal 319(h) grant and operated through the Georgia Environmental Protection Division, has developed a robust program consisting of manuals, training, and network support and has become a model for volunteer water quality monitoring programs in the southeast (AAS, 2014). Volunteers are trained using quality assurance/quality control (QA/QC) protocols for measuring biological, chemical, and physical parameters and must obtain certification via practical and written exams in order to become a “QA/QC volunteer.” This designation enables volunteers to enter data into an online AAS database which, in turn, can be accessed by a variety of entities including universities, environmental groups, and regulatory agencies for the purpose of monitoring the health of local waterways. The project described here demonstrates the ability of AAS protocols to gather useful, quantitative data which can be used for compiling baseline water quality information and addressing research questions.
PROJECT DESCRIPTION

Study Site

The focus of this project was Twelve Mile Creek located in Pickens County in the northwestern corner of South Carolina. Twelve Mile Creek (R.61-69 classification of FW-Freshwaters) originates near the community of Nine Times and flows into and forms an upper arm of Lake Hartwell near the city of Clemson. The Twelve Mile watershed covers almost 99,000 acres (155 mi²) of which approximately 72 percent is forested. The remaining land use types include pasture land (13%), cropland (6%), urban areas (7%), and a small mix of wetlands, barren, and transitional land uses. The Twelve Mile watershed also contains the Town Creek drainage which was placed on the EPA's National Priority List (NPL) in 1990 because of contaminated debris, groundwater, sludge, sediment and fish tissue resulting from the operation of the Sangamo-Weston capacitor manufacturing facility from 1955 to 1987, the primary contaminant being polychlorinated biphenyls (PCBs) (Brutzman, 2012; U.S. EPA, 2012). Various mitigation and restoration efforts have taken place over the last two decades, and while PCB contamination in the main channel of Twelve Mile Creek apparently poses no significant public health risk, the problem is still being addressed (U.S. EPA, 2009).

From a human dimensions perspective, a portion of Twelve Mile Creek has recently been targeted for restoration as part of a mitigation settlement which required the removal of two concrete dams constructed in the early 1900s. The gradient of this section is approximately 56 feet per mile and removal of the dams opened up an approximately two-mile stretch of whitewater. The area has become a destination for whitewater paddlers and is being marketed as a recreational resource identified as the Twelve Mile Creek Blueway (ACA, 2014; Simon, 2011).

In addition to its history of industrial PCB contamination, Twelve Mile Creek and several of its tributaries have been regularly identified on the State of South Carolina 303(d) List for Impaired Waters with the primary contaminant being fecal coliform bacteria. High bacteria levels have been documented during both storm events and low flow periods with sources likely including wildlife, failing septic systems, and livestock (S.C. DHEC, 2003, 2013a, and 2013b). Moreover, a Total Maximum Daily Load (TMDL) Development Plan for the Twelve Mile watershed to address bacterial waste loads has been in effect for approximately ten years (S.C. DHEC, 2003).

Research suggests that in some watersheds which are impaired due to high bacteria, indicator bacteria levels increase with increasing flow rate, usually immediately after significant rainfall events (Tiefenthaler et al., 2011; Marsalek and Rochfort, 2004). Ironically, it is under increased flow conditions after rainfall events that Twelve Mile Creek experiences its highest use by paddlers (AWA, 2014). Since high levels of indicator bacteria are correlated with increased incidence of gastrointestinal illness (Frenzel and Couvillion, 2002; O’Shea and Field, 1992), being able to document and monitor bacteria levels and potential health risks, under both baseflow and stormflow conditions, will be useful to a number of stakeholders including paddlers, regulatory agencies, community planners, and local tourism officials.

Project Objectives

By utilizing the formal sampling protocols created, administered, and regulated by Georgia Adopt-a-Stream, this project demonstrated the use of these methods to gather useful, quantitative data for monitoring water quality in a Total Maximum Daily Load (TMDL) watershed which is also being marketed for recreational use. As a point of reference, AAS, in line with current EPA practice, utilizes Escherichia coli as an indicator organism for the presence of pathogenic bacteria (AAS, 2009).

The questions addressed in this project were:

1. Is there a relationship between discharge and E. coli concentrations in Twelve Mile Creek?
2. Does Twelve Mile Creek exhibit changes in E. coli concentrations among sites along a longitudinal gradient commonly used for recreational paddling?
3. Can protocols utilized by volunteer monitoring programs like Adopt-a-Stream provide useful data to address questions such as these?

METHODS

Three sites along the proposed Twelve Mile Creek Blueway corridor were chosen based on strategic location (put-in and take-out spots) and ease of access. Sites were as follows: Site 1 - SC Highway 137 approximately 100 meters upstream from the Virgil Mitchell Memorial Bridge; Site 2 - Lay Bridge Road, approximately 100 meters upstream from the iron bridge; and Site 3 - Maw Bridge Road, approximately 100 meters upstream from the bridge crossing Lake Hartwell.

Between February and September 2014, each site was sampled approximately once a month during baseflow conditions (no rain in at least five days) and within eight hours after substantial rainfall (≥ 1.25 cm or 0.5 inches) had occurred. Rainfall and discharge data were monitored remotely using the USGS Twelve Mile Creek gage near Liberty, SC (Gage #02186000). This gage is located approximately 6.8, 9.6, and 12.8 kilometers (4.2, 6.0, and 8.0 river miles) upstream from Sites 1, 2, and 3, respectively. Samples for bacteria were obtained onsite following AAS QA/QC protocols (AAS, 2009). Plating, incubation, and counting were conducted in a lab setting on the campus of Southern Wesleyan University, Central, SC, using E. coli Coliform Petrofilm® (3M) media.

To explore the relationship between discharge and E. coli levels, discharge was recorded in cubic feet per second (cfs) and bacteria counts in colony-forming units (cfu) per 100 ml of sample. Utilizing data from Site 2, because it is a popular take-out spot for paddlers, seven samples were obtained during baseflow conditions and six
Use of a Volunteer Monitoring Program to Assess Water Quality during stormflow conditions. These data were evaluated using simple regression analysis. To address differences in bacteria concentrations among the three study sites along the paddling corridor, medians of observed bacteria counts were compared using Kruskal-Wallis and Mann-Whitney U Tests.

RESULTS

During the sampling period, discharge levels ranged from a minimum of 78 cfs to a maximum of 2110 cfs with medians of 149 cfs during baseflow conditions and 445 cfs during stormflow conditions. Across all sites, \( E. coli \) concentrations ranged from a minimum of 33 cfu/100 mL to a maximum of 5933 cfu/100 mL with medians of 233 cfu/100 mL during baseflow conditions and 1100 cfu/100 mL during stormflow conditions.

At Site 2, a strategic location within the paddling corridor, \( E. coli \) concentrations did increase with rising discharge levels during or following substantial rainfall. There was a strongly significant relationship between discharge (cfs) and \( E. coli \) levels (cfu) (\( R^2 = 0.644, n = 13, p = 0.00049 \)) (Figure 1).

Under baseflow conditions, Sites 1, 2, and 3 exhibited no significant differences in median \( E. coli \) levels (268, 268, and 168 cfu/100 mL respectively; \( H = 4.54, df = 2, p = 0.105 \)). Under stormflow conditions, Sites 1, 2, and 3 exhibited no significant differences in median \( E. coli \) levels (1100, 1350, 967 cfu/100 mL, respectively; \( H = 1.48, df = 2, p = 0.477 \)). \( E. coli \) levels were essentially the same across all three study sites regardless of flow condition (Figure 2). Anecdotally, Site 3 exhibited lower variability in \( E. coli \) levels which is likely due to the fact that this site is located at the confluence of Twelve Mile Creek with Lake Hartwell where conditions (flow rate, temperature, turbidity) tended to be much more constant, even during periods of stormflow.

Cumulative data for all three sites indicated that there was a significant difference in median \( E. coli \) levels during baseflow when compared to stormflow conditions (\( U = 11.5, df = 1, p < 0.05, n = 21, n = 18 \), respectively).

DISCUSSION

During stormflow conditions, there were no significant differences in \( E. coli \) concentrations among the three study sites along a two-mile corridor utilized by recreational paddlers. Since \( E. coli \) concentrations were virtually the same for all three sites within the paddling corridor, it can be assumed that the primary source of bacteria is located upstream.

As is true with many impaired watersheds, Twelve Mile Creek does experience elevated bacteria counts during stormflow discharges. While not surprising, these observations are noteworthy because recreational paddling use of this section of Twelve Mile Creek is more “desirable” at higher discharge levels, for example above 500 cfs (AWA, 2014). At a discharge of 500 cfs, the regression plot generated from the data in this study (Figure 1) suggests \( E. coli \) concentrations would be greater than 1500 cfu/100 mL. The EPA’s criterion limit for \( E. coli \) for recreational waters is 126 MPN (most probable number per 100 mL) (U.S. EPA, 2014). Therefore, Twelve Mile Creek may pose the greatest health risks to users when it is at its most attractive for paddling.

An obvious question raised by this study involves what the source(s) of bacterial contamination is (are). Since approximately 20% of the land cover in the Twelve Mile Creek watershed is pasture and cropland, agricultural runoff is a possible explanation. In addition, the Cateechee community, a 1920s era cotton mill village, sits on a bluff near the southern portion of the watershed, just outside the town of Norris. While the mill closed in the 1970s, many of the homes are still occupied by residents. The area does have a wastewater

![Figure 1.](image-url)
treatment plant, but the system has been targeted for remediation through the Clean Water State Revolving Fund for Wastewater and Nonpoint Source Project program (S.C. DHEC, 2015), and leaking infrastructure could be a significant potential source of contamination. On-the-ground reconnaissance and source tracking are surely appropriate pursuits.

Additional questions that deserve future consideration include:

1. How quickly do bacteria levels return to normal after a stormflow event?
2. Does recreational use of Twelve Mile Creek during periods of higher discharge actually lead to a higher incidence of illness among those users?
3. What management actions should be taken in light of the findings of this study?

Also, this investigation spanned just eight months and contains a relatively small number of samples. Additional data are needed to confirm and corroborate conclusions. Likewise, there is a need for multivariate studies looking at other water quality parameters that may correlate with bacteria levels including temperature, dissolved oxygen, and turbidity.

This project addresses human dimensions of strategic water planning including land use, water quality, and recreational resources, and it demonstrates the utility of a volunteer water quality monitoring program (VM) to collect useful data that can be used for educational, monitoring, and research purposes. However, it should be noted that while Georgia Adopt-a-Stream does have QA/QC guidelines in place, most volunteer water quality monitoring programs, including this one, are not rigorous enough for their data to be utilized for regulatory purposes such as 303(d) listing, MS4 reporting, and compliance monitoring. For example, the use of 3M Petrifilms® is not an EPA-approved method for estimating E. coli levels (but see Vail et al., 2003 for a discussion of this situation).

On the other hand, appropriately-designed VM efforts do hold promise for use as screening tools. Despite their shortcomings, VM programs can provide another means for quantifying anthropogenic impacts on watersheds and monitoring potential health risks at a resolution, reach, and efficiency that municipalities and regulatory agencies may not be able to replicate. Moreover, effective volunteer water quality programs have potential for creating a mutually beneficial situation for a variety of stakeholders: Citizens develop a vested interest and sense of ownership in protecting the watersheds in which they live, municipalities and regulatory agencies have access to quality data that can be used in monitoring decisions, and ultimately, natural resources enjoy more conservation and protection due to increased attention.

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LITERATURE CITED


Use of a Volunteer Monitoring Program to Assess Water Quality


Decision Support System for Optimally Managing Water Resources to Meet Multiple Objectives in the Savannah River Basin

Edwin A. Roehl, Jr. and Paul A. Conrads

Abstract. Managers of large river basins face conflicting demands for water resources such as wildlife habitat, water supply, wastewater assimilative capacity, flood control, hydroelectricity, and recreation. The Savannah River Basin, for example, has experienced three major droughts since 2000 that resulted in record low water levels in its reservoirs, impacting dependent economies for years. The Savannah River estuary contains two municipal water intakes and the ecologically sensitive freshwater tidal marshes of the Savannah National Wildlife Refuge. The Port of Savannah is the fourth busiest in the United States, and modifications to the harbor to enable more ship traffic have caused saltwater to migrate upstream, reducing the Refuge’s freshwater marsh’s acreage more than 50 percent. A planned deepening of the harbor includes flow-alteration features to minimize further migration of salinity, whose effectiveness will only be known after all construction is completed.

One of the challenges of large basin management is the optimization of water use through ongoing regional economic development, droughts, and climate change. This paper describes a model of the Savannah River Basin designed to continuously optimize regulated flow to meet prioritized objectives set by resource managers and stakeholders. The model was developed from historical data using machine learning, making it more accurate and adaptable to changing conditions than traditional models. The model is coupled to an optimization routine that computes the daily flow needed to most efficiently meet the water-resource management objectives. The model and optimization routine are packaged in a decision support system that makes it easy for managers and stakeholders to use. Simulation results show that flow can be regulated to substantially reduce salinity intrusions in the Savannah National Wildlife Refuge, while conserving more water in the reservoirs. A method for using the model to assess the effectiveness of the flow-alteration features after the deepening also is demonstrated.

INTRODUCTION

The Savannah River Basin (Basin; Figure 1a) is a prototypical large basin whose water-resource managers face conflicting demands, such as wildlife habitat, water supply, wastewater assimilative capacity, flood control, hydroelectricity, and recreation. In the upper Basin, the U.S. Army Corps of Engineers (USACE) controls three large reservoirs - Lake Hartwell, Richard B. Russell Lake (Lake Russell), and J. Strom Thurmond Lake (Lake Thurmond). Lake Russell has comparatively little storage, leaving Lakes Hartwell and Thurmond to provide most of the regulated flow to the coast. Since 2000 the upper Basin has experienced three major droughts, resulting in record and near-record low reservoir water-level elevations that impacted dependent economies by reducing tourism and real estate values (Allen et al., 2010; USACE, 2014).

The Savannah River estuary (estuary; Figure 1b) contains two municipal water intakes and the ecologically sensitive freshwater tidal marshes of the Savannah National Wildlife Refuge (Refuge). The interaction of regulated streamflow, tides, and weather produces salinity intrusions more than 25 miles upstream at U.S. Geological Survey (USGS) gage 02198840. The gage is located where the river intersects Interstate 95 (I95), and near the City of Savannah’s municipal freshwater intake on Abercorn Creek.

The Port of Savannah is the fourth busiest in the United States, and modifications to the harbor to enable more ship traffic have caused saltwater to migrate upstream, reducing the Refuge’s freshwater marsh’s acreage more than 50 percent since the 1970s (Conrads et al., 2006). A currently planned deepening of the harbor includes flow-alteration features to minimize further salinity migration; however, the estuary’s complex hydrology and the extensive scope of all the construction make the final outcome uncertain. For example, a tide gate installed during the 1970s in the estuary’s Back River to reduce shoaling unintentionally increased salinities and decreased dissolved-oxygen levels in the habitat of a large striped bass population. A consequent
97% decrease in striped bass abundance led to the tide gate being decommissioned in 1991 (Reinert, 2004).

Managing the water resources of the Basin in an optimal way will require a tool that can determine on an ongoing basis how water should be allocated for multiple purposes, such as regional economic development, drought protection, and reducing salt-water intrusion and sea-level rise impacts. The general solution is to save water for future use by reducing regulated flows to the minimum volume needed to meet objectives prioritized by resource managers and stakeholders. Meeting the increasing and often conflicting usage demands in a constantly changing hydrologic system like the Savannah River Basin is an ongoing, multi-objective optimization problem. This paper describes the development of a decision support system (DSS) using artificial neural network (ANN) models that continuously optimize water levels in Lakes Hartwell and Thurmond while reducing salinity in the Refuge and near coastal municipal intakes.

PROJECT DESCRIPTION

This project built upon two previous studies. The first developed an empirical hydrodynamic and water-quality model of the lower Savannah River to estimate the impacts of the planned harbor deepening on the Refuge (Conrads et al., 2006). The model was developed from historical data using ANNs (Jensen, 1995), a form of machine learning. The model was packaged in a spreadsheet-based DSS (Roehl et al., 2006), making it easy for managers and stakeholders to use. It was named the Model-to-Marsh DSS (M2M-DSS) because it connected two other models together: a 3-D finite-difference hydrodynamic model of the estuarine rivers and shipping channel (Tetra Tech, 2005), and a “plant succession model” of the sensitivity of the Refuge’s marsh plant communities to salinity (Welch and Kitchens, 2006).

A second study modified the M2M-DSS, renamed M2M2-DSS, to estimate how sea-level rise and climate change would affect the magnitudes, frequencies, and durations of salinity intrusion events in the lower Savannah River (Conrads et al., 2013). This project developed a third version of the M2M-DSS, named M2M3-DSS, to study how the water resources of the upstream reservoirs could be managed differently to better protect the Refuge from salinity migration and to conserve water.

Figure 2a shows the normalized measured (m) water-level elevations (ELV) of Lakes Hartwell and Thurmond in feet (ft), labeled ELV.Hart.m and ELV.Thur.m, for the February 10, 2007, to January 8, 2012, study period. All the time series presented herein use a daily time step. Normalization was performed by subtracting full pond elevations from the measured elevations. Lakes Hartwell and Thurmond reached their lowest and second lowest elevations, respectively, during the winter of 2008.

Figure 2b shows two flows (Q), the measured regulated flow from Lake Thurmond (Q.Thur.m), and the streamflow at USGS gage 02198500 near the town of Clyo (Q.Clyo.m). The study period includes the climatic extremes of two severe droughts and an El Niño episode. The Lake Thurmond flow contributes most of the flow at Clyo, with additional flow due to rainfall runoff and groundwater discharge between the gaging sites. During the droughts, Q.Thur.m was held nearly constant at the regulatory minimum flow deemed necessary to protect downstream water intakes and the Refuge. Figure 2c shows the measured maximum (max) and minimum (min) water levels (WL) in Savannah Harbor (WL.max.m and WL.min.m) recorded at USGS gage 02198980. The major factors causing the water-level variability are tides, weather, and streamflow.

Figure 2d shows the measured maximum specific conductance in the Refuge and at I95 (SC.Rfg.max.m and SC.I95.max.m, respectively) recorded at USGS gages 021989784 and 02198840. Specific conductance (SC) is the field measurement used to compute salinity. The spikes indicate intrusion events that occur during spring tides of the new moon when tidal ranges are greatest. Tides having a low range are called neap tides. Annual specific conductance cycles coincide with those of the water levels in Figure 2c. Specific conductance is also modulated by weather and streamflow, which vary the magnitudes and durations of intrusion events.
METHODS

ANNs synthesize nonlinear functions to fit multivariate calibration data rather than use predefined functions like mechanistic and statistical models. ANNs can adapt to changing conditions by updating the calibration data, and have been applied to a number of hydrology problems. ANNs have been used to predict unmeasured riverine flows at locations between USGS gauging sites (Karunanithi et al. 1994), model flow conditions that lead to interfacial mixing in estuaries with vertical salinity and temperature stratification (Grubert, 1995), forecast salinity at an estuary site (Maier and Dandy, 1997), and forecast river stages in real-time (Thirumalaiiah and Deo, 1998). Conrads and Roehl (1999) and Conrads and Greenfield (2010) found that ANN models of the Cooper and Savannah River estuaries had significantly lower prediction errors than mechanistic models of the same systems and executed much faster.

Fast execution allows a model to be coupled to an optimization routine that systematically tests and finds values for the model’s controllable inputs, so that the model generates the output needed to meet one or more objectives. Dowla and Rogers (1995) combined an ANN-based groundwater model with optimization to evaluate millions of possible well patterns for remediating a contaminated site, and credited the approach with a potential $100 million savings in remediation costs. A DSS that incorporated dissolved-oxygen concentration models with optimization was used to estimate the total maximum daily loads (TMDL) for biochemical oxygen demand and ammonia for three wastewater treatment plants on the Beaufort River (Conrads et al., 2003). To facilitate the relicensing of hydropower facilities on the Yadkin River by the Federal Energy Regulatory Commission, a DSS was developed that incorporated salinity models of the Waccamaw River and the Atlantic Intracoastal Waterway with optimization. The DSS was used to estimate the minimum flows required to prevent salinity inundation of municipal freshwater intakes in the Grand Strand (Conrads and Roehl, 2007).

The M2M3-DSS’s optimization routine computes the predicted (p) flows at Clyo (Q.Clyo.p) needed to meet setpoints using ANN model predictions of average (avg)
and maximum specific conductance in the Refuge and at I95 (SC.Rfg.avg.p, SC.Rfg.max.p, SC.I95.avg.p, and SC.I95.max.p, respectively). The predicted flows from Lake Thurmond (Q.Thur.p) are calculated by subtracting the difference between the measured Clyo and Lake Thurmond flows from Q.Clyo.p. Lake elevation setpoints are input to the M2M3-DSS as hydrographs and are used to calculate the flows needed from each lake to meet Q.Thur.p. The user-specified specific conductance setpoints have priority over the elevation setpoints. Flows from Lakes Hartwell and Thurmond are balanced so that they are kept equidistant from their elevation setpoints. This closely matches the current management practice in which “rule curves” are used to set outflows according to the month of the year and the current elevations.

To develop the ANNs, historical USGS data were randomly partitioned into training and testing datasets. The measured Clyo flow and harbor maximum and minimum water-level signals were decomposed into different frequency components that represent variability on daily, weekly, monthly, and seasonal time scales. During training, an ANN effectively selects the frequency components that provide the best fit. Figure 3 shows the measured and predicted maximum specific conductance in the Refuge and at I95. The Refuge model’s coefficients of determination (R²) for the training and testing datasets were 0.76 and 0.71, respectively. The I95 model’s R² for the training and testing datasets were 0.67 and 0.72, respectively. More details about developing ANN models of estuary specific conductance are given in Conrads et al. (2013).

RESULTS

Two simulations were performed to evaluate different resource management issues. Conrads and Greenfield (2010) had used the M2M-DSS to estimate the effect of a timed streamflow pulse on a large intrusion event recorded at I95. Scenario 1 (s1) extended this idea to determine if modulating water releases according to changing conditions could reduce salinity in the Refuge and upstream at I95, while also conserving water in the lakes. Scenario 2 (s2) simulated a substantial change to the harbor to demonstrate how the M2M3-DSS could be used to detect changes in salinity behavior caused by alterations to the harbor.

Scenario 1 used five setpoints representing different optimization objectives. To simulate protecting the City of Savannah’s municipal freshwater intake, the setpoint for the predicted maximum specific conductance at I95 (s1.SC.I95.max.p) was 1,000 μS/cm, which equates to a commonly used upper limit for freshwater (freshwater limit) of 0.5 practical salinity units. The setpoint for the predicted maximum specific conductance in the Refuge (s1.SC.Rfg.max.p) was 2,000 μS/cm, a limit that was regularly exceeded in the measured data (Figure 2d). Note that frequently the reservoir outflows would need to be higher than the historical outflows to meet these two setpoints. To compensate for the higher outflows, a third setpoint for the predicted average specific conductance in the Refuge (s1.SC.Rfg.avg.p) was 650 μS/cm, which was higher than the 561 μS/cm measured average for the study period, but was still well below the freshwater limit. Two setpoints for predicted elevations of Lakes Hartwell and Thurmond (s1.ELV.Hart.p and s1.ELV.Thur.p) were full pond + 2.0 ft; these elevations have commonly been exceeded.
The Scenario 1 results showed that predicted flow from Lake Thurmond (s1.Q.Thur.p) was much more variable than the measured flow (Q.Thur.m) (Figure 4a). Apart from the El Niño episode, the s1.Q.Thur.p trends with the largely periodic, and therefore predictable, water-level signals shown in Figure 2c. The predicted flows eliminated most of the spikes in the Refuge’s measured average specific conductance SC.Rfg.avg.m, but also allowed the predicted average specific conductance s1.SC.Rfg.avg.p to rise to the 650 μS/cm setpoint when SC.Rfg.avg.m was lower than the setpoint, as seen during December 2007 (Figure 4b). The few predicted values above the setpoint resulted from an optimization constraint that limited daily flow changes in order to dampen flow variability. The number of days when the freshwater limit was exceeded was predicted to decrease from 230 to 34 (-85%).

Figure 4e compares the measured and predicted maximum specific conductance in the Refuge, SC.Rfg. max.m and s1.SC.Rfg.max.p. The number of days when the Refuge’s maximum specific conductance exceeded the 2,000 μS/cm setpoint was predicted to decrease from 126 to 10 (-92%). Figure 4d compares the measured and predicted maximum specific conductance at I95 (SC.I95.max.m, s1.SC.I95.max.p), and e) Lakes Hartwell and Thurmond elevations (ELV.Hart.m, s1.ELV.Hart.p, ELV.Thur.m, s1.ELV.Thur.p).

The Scenario 1 data showed that predicted flow from Lake Thurmond (s1.Q.Thur.p) was much more variable than the measured flow (Q.Thur.m) (Figure 4a). Apart from the El Niño episode, the s1.Q.Thur.p trends with the largely periodic, and therefore predictable, water-level signals shown in Figure 2c. The predicted flows eliminated most of the spikes in the Refuge’s measured average specific conductance SC.Rfg.avg.m, but also allowed the predicted average specific conductance s1.SC.Rfg.avg.p to rise to the 650 μS/cm setpoint when SC.Rfg.avg.m was lower than the setpoint, as seen during December 2007 (Figure 4b). The few predicted values above the setpoint resulted from an optimization constraint that limited daily flow changes in order to dampen flow variability. The number of days when the freshwater limit was exceeded was predicted to decrease from 230 to 34 (-85%).

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Scenario 2 demonstrated that a system similar to the M2M3-DSS could be used to promptly identify changes after the deepening occurs. The idea was that an accurate model of the estuary’s pre-deepening behavior can be used as a reference for quantifying the effects of later changes. The planned deepening will increase the depth of the navigation channel by 5.0 ft. To create a surrogate, post-deepening dataset, the effect of a 2.0-ft sea-level rise on the Refuge’s average specific conductance was simulated. The surrogate dataset, s2.SC.post, had a study-period average of 902 μS/cm versus 562 μS/cm for the measured data, an increase of 61% (Figure 5a). The number of days exceeding the freshwater limit increased 220%. The model’s representation of the Refuge’s pre-deepening behavior consisted of predictions made using only measured input data, s2.SC.pre. The 95th percentile prediction error (ε) was 348 μS/cm.

Figure 5b shows s2.SC.pre + ε and s2.SC.post with Running%, the running percentage of days from the start of the study period when s2.SC.post exceeded s2.SC.pre + ε. Initially, Running% generally followed the annual specific conductance trend, and then stabilized to a range between 40% and 50%. The higher s2.SC.post values shown in Figure 5a became apparent in the Running% within the first 3 months of the study period. Discriminating the higher values was made possible by the accuracy of the model’s representation of the pre-deepening system behavior. Quickly detecting and correcting adverse consequences of actions is necessary to manage the resource most effectively.

DISCUSSION

Scenario 1 indicated that episodic high salinity at multiple locations can be controlled by parsimonious reservoir releases that conserve water. The values chosen for the three specific conductance setpoints were somewhat arbitrary, with the freshwater limit being used as an example standard for two of them. However, all three setpoint values aimed to significantly reduce the predicted salinity below the measured salinity (Figures 4b, 4c, 4d), requiring reservoir outflows that were frequently much higher than the measured outflows (Figure 4a). The two chosen elevation setpoint values were consistent with observed USACE operating practices. In an operational M2M3-DSS, setpoint values could be adjusted when warranted by changing Basin conditions or new information. Additional setpoints and constraints could be added to represent other concerns, however, too many would limit operating flexibility.

Determining the dollar value of the water saved requires further study; however, the 2.7 and 3.4 ft average elevation increases in Lakes Hartwell and Thurmond equate to 151,200 and 241,400 acre-ft for deferred power generation, respectively. Given that droughts vary in severity and duration, and can appear in rapid succession (Figure 2b), slower elevation decreases would increase the probability that dependent economies would emerge from droughts less affected.

Scenario 2 showed how a model that is accurately calibrated for one set of conditions can be a tool for quickly detecting and quantifying adverse differences caused by a new set of conditions. The impetus for employing such a tool for the deepening emanates from: uncertainty about the net

Figure 5. Scenario 2 (s2) results for the Refuge: a) measured and surrogate post-deepening average specific conductance (SC. Rfg.avg.m, s2.SC.post), and b) s2.SC.post and simulated pre-deepening average specific conductance + 95th percentile model prediction error ε=348 μS/cm (s2.SC.pre + ε), and running percent of days when s2.SC.post exceeded s2.SC.pre + ε (Running%).
deepening impacts on the estuary’s intakes and ecosystems; the experience of the Back River tide gate, whose adverse impacts were not simulated by the pre-construction models; and the demonstrated performance of ANN-based estuary models in several projects.

A DSS similar to the M2M3-DSS could be deployed for daily (or more frequent) use. In Figure 6, current data from the USGS, USACE, and weather stations [a] are input [b] to the DSS’s database and then processed for quality assurance and input to the DSS’s near-term weather and tidal forecasts. Constraints [c] on the regulated streamflows, such as the minimums required for scheduled hydropower generation, are entered and stored in the database. Specific conductance and elevation setpoints [d] are similarly entered. The DSS computes “suggested” regulated flows [e] that are optimized for the current and near-term forecasted conditions for use by management personnel.

CONCLUSIONS

A decision support system like the M2M3-DSS can transform streams of data into the information needed to make informed water management decisions. The simulations described here indicate that a management approach that continuously optimizes water releases might substantially reduce salinity in the Refuge and near municipal intakes while conserving more water in the reservoirs. They also indicate that changes in salinity due to modifications such as the harbor deepening could be quickly quantified, allowing the performance of mitigation factors such as the planned flow-alteration features to be proactively evaluated. The overall approach could be expanded within the Savannah River Basin and possibly applied to other large basins.

LITERATURE CITED


Interstate Water Compacts: Partnerships for Transboundary Water Resource Management

Cindy G. Roper

Abstract. While there are both successes and challenges related to the use of interstate water compacts, in their most effective forms they allow states to take a comprehensive, holistic approach to water management. Successful compacts tend to encompass the natural hydrologic boundaries of the water basin. They are more likely to utilize a commission type governance structure with sufficient authority to carry out the mission and goals of the compacting agreement. Successful compacts are flexible and allow for future developments (including climate change) while being cognizant of the need to protect and enhance the environment. They are also sensitive to the needs and desires of various stakeholders, including federal, state, and local governments as well as non-governmental organizations.

Water compacts also face a variety of challenges. They must answer to a wide and diverse constituent base, often with conflicting interests. Stronger states can and do attempt to “bully” other states, severely limiting or eliminating altogether the usefulness of the compact. Governance structures that fail to integrate the interests of all states into a single body simply make the compact into an arena where small scale water wars can be fought.

To illustrate an area where interstate water compacts could make a significant contribution, this paper concludes with a case study highlighting South Carolina’s transboundary water issues with North Carolina and Georgia. Recommendations for South Carolina include beginning negotiations toward the development of federal-interstate compacts as well as considering action in the Supreme Court in the event that these negotiations fail.

INTRODUCTION

Of the fifty states that comprise the United States of America, only two - Alaska and Hawaii - do not share a ground or surface water resource with another state. Accordingly...the forty-eight contiguous states fall into one of two categories: those states that are (or have been) involved in an interstate water conflict or those states that are going to be involved in an interstate water conflict (Sherk, 2005, p.765).

That statement was made nearly ten years ago. Since then, increasing population, climate change and new technologies are putting even more pressure on water resources. States are having to re-evaluate how they manage these assets, both within their borders and those that are shared with neighboring states. As part of this process, state officials need to develop a clear picture of what future needs and conflicts may emerge and how these might be mitigated. They also need to prepare flexible mechanisms for dealing with the uncertainty that accompanies almost any planning effort. Without the means to successfully address transboundary water issues, options are limited and too often result in undesirable outcomes.

The purpose of this paper is to examine federal-interstate compacts as a possible solution to both existing and emerging issues related to shared water resources. It provides an overview of the advantages and challenges of utilizing interstate compacts as well as giving examples of compacts that have experienced various rates of success. Furthermore, it examines transboundary water issues and the prospect of compact development in South Carolina.

The information in this paper is especially relevant for those who are charged with providing solutions to problems emerging from shared water resources. It provides an alternative to piecemeal administration that is not equipped to deal with problems that require the broad participation of other parties to solve. Overall, this paper illustrates a mechanism that allows for extensive stakeholder participation within a comprehensive, flexible framework that has been shown to work in complex transboundary water resource situations.

METHODOLOGY

This paper is a comparative study of factors that likely influence the success or failure of interstate water compacts. It utilizes scholarly writings, legal and historical sources, governmental and non-governmental reports, and media sources. It also relies substantively on the work of the Utton Transboundary Resource Center at the University of New Mexico School of Law as a guide to compact development and function. It concludes with a case study that uses the
findings from the research to provide an example of the possible utilization of interstate compacts for water resource management in South Carolina.

BACKGROUND AND RELATED WORK

Under federalism, states have primary responsibility for water within their borders while the federal government regulates and manages water resources under the Commerce Clause, the Federal Clean Water Act, the Endangered Species Act, the Rivers and Harbors Act and in conjunction with the Army Corps of Engineers. The federal government also constructs and controls large-scale reclamation and flood control projects and licenses non-federal hydropower projects through the Federal Energy Regulatory Commission (Muys, 1995). This jumble of responsibilities often leads to ambiguity as to what federal or state entity has jurisdiction over a specific body of water or section of river, ultimately resulting in some degree of conflict (Mandarano, Featherstone, and Paulsen, 2008). Lepawsky stated the problem rather succinctly when he said, “Few functions of the American Federal system seem less suited physically to state boundaries than the management of our water resources” (1950, p. 631). Mandarano, Featherstone, and Paulsen comment in more detail.

Water and federalism are a complicated mix as water flows through the hydrologic cycle without regard to political boundaries. The physical boundaries of river basins do not coincide with the geographic boundaries of political jurisdictions. The management of interstate water resources is complicated by the multiple, conflicting, and overlapping functions and interests of federal and state governments, and is further complicated by conflicting regulatory authority and policy priorities between different federal agencies (2008, p. 136).

Compacts, as problem-solving mechanisms, date back to the pre-revolutionary period. Their origins emerged, for the most part, from early boundary disputes that were settled by negotiated agreements between the colonies and were contingent upon the approval of the Crown (Kearney and Stucker, 1985). Later, in the Articles of the Confederation, compacts reflected the need to settle disputes among states as well as to protect the new nation “from the destructive political combination of two or more States” (Frankfurter and Landis, 1925, p. 693). In the same vein, Article 1, §10, Clause 3 of the U.S. Constitution forbids states to enter into agreements among themselves without the approval of Congress (U.S. Constitution, Article I., n.d.), reinforcing the importance of compacts as tools for protecting the union as well as solving problems between states.

The Supreme Court has made itself clear on the issue of intervening between states. In Colorado v. Kansas, the Court explained why adjudication was not the most efficient way to solve interstate water disputes.

The reason for judicial caution in adjudicating the relative rights of States in such cases is that, while we have jurisdiction of such disputes… they involve the interests of quasi-sovereigns, present complicated and delicate questions, and, due to the possibility of future change of conditions, necessitate expert administration rather than judicial imposition of a hard and fast rule. Such controversies may appropriately be composed by negotiation and agreement, pursuant to the compact clause [emphasis added] of the federal Constitution. We say of this case, as the court has said of interstate differences of like nature, that such mutual accommodation and agreement should, if possible, be the medium of settlement, instead of invocation of our adjudicatory power (1943, 320 U.S. 383).

Furthermore, in Nebraska v. Wyoming, the Court reiterated that in undertaking the apportionment of an interstate river, they would “embark upon an enterprise involving administrative functions beyond our province” (1945, 325 U.S. 616). Clearly, the Court believes that compacts are a viable tool for managing water resources that cross state lines and should be utilized whenever possible.

As such, interstate compacts can serve as a platform for intergovernmental cooperation. They allow states to exercise authority over issues within their purview while relieving the federal government of responsibility for problems better left to the states. At the same time, they provide a method for states and the federal government to work together to “solve mutual problems in a collective fashion” (Kearney and Stucker, 1985, p. 210).

Basically, the compact is a legal agreement between two or more states entered into in order to deal with a problem or concern that crosses state boundaries. Because of its contractual nature, a compact takes precedence over prior law and over legislation that may later be enacted by member states. Because a compact is also a contract between the participating states, it differs from other statutes. As a contract, an interstate compact is binding on member states in the same manner as any other contract entered into by an individual or corporation. Once entered into, compacts cannot be unilaterally amended or repealed; they are binding on all citizens of the signatory states. If a state violates or fails to honor the terms of a compact, an offended state or states may sue in state or federal court (Florestano, 1994, p. 14).
Successful interstate water compacts tend to share certain characteristics. Viable compacts must be able to meet and negotiate changing conditions, therefore, they must be designed with flexibility in mind. Successful compacts are often those specifically created for individual circumstances. Also, successful compacts are those that can be implemented with few external constraints. Another characteristic of successful compacts is that they routinely involve water resource management experts who have a better understanding of technical data, long-term outcomes, and different available options (Tarlock, cited in Stephenson, 2000). George Sherk (2005) provides a list of institutional attributes that he argues can contribute to creating effective compacts. Many of these have been incorporated into the model compact developed by the Utton Transboundary Resources Center in the University of New Mexico’s School of Law (Muys, Sherk, and O’Leary, 2007) of which he is a co-author.

The Importance of Compact Commissions

Because compact commissions are such an essential part of many successful compacts they warrant special attention. Compact commissions are, as Stephenson says, “…how interstate water compacts make their greatest contribution to water resource management” (2000, p. 99). These permanent commissions provide authority and structure for the agreements (Stephenson, 2000), gather information, meet and discuss water problems, develop regulations to administer compacts, monitor water use, and mediate conflict (Schlager and Heikkila, 2009). Compact commissions also allow for the participation of stakeholders in decision making and for transparency in processes and outcomes (See the Delaware River Basin Compact, 1961).

According to the Utton model compact, commission members include the governors of all of the signatory states or their representatives, a single tribal representative elected from all tribes who are parties to water allocation agreements within the jurisdiction of the compact, and a federal representative. This federal representative is appointed by the President after consultation with federal agencies with interests in the basin and he or she will actively participate in

SUCCESSFUL INTERSTATE COMPACTS

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the commission’s discussions. The federal representative will coordinate the viewpoints of all federal agencies in the basin with responsibilities related to water resources and present a single, coordinated federal position during commission deliberations (Muys et al., 2007).

Some of the powers that are critical to a strong commission include the ability to carry out comprehensive planning, making and enforcing rules, monitoring compliance, financing and constructing projects, and approving intra- and inter-basin transfers. Commissions are also empowered to acquire, hold, convey and dispose of property, enter into contracts, sue and be sued, issue permits, collect fees, levy taxes, and establish standards. They can also negotiate for loans, grants, and services and perform all functions required by the compact. Other aspects of successful commission functioning include majority voting rules with a tie-breaker provision, allowing the federal member a vote, having power to act in an emergency, and other necessary and proper ancillary powers (Muys et al., 2007).

The Delaware River Basin Compact

The Delaware River Basin Compact (DRBC), the first federal-interstate compact developed, has emerged as a model compact (Dellapenna, 2006; Muys, 1995; Zimmerman, 2012). However, its beginnings were anything but ideal. New Jersey, New York and Pennsylvania began negotiations regarding a possible interstate compact as early as 1923. In 1926, New York City, in a predatory move toward Delaware, announced that it planned to utilize the river as the major municipal water source – even though the city has no remotely riparian claims on the river. This initiated an extended confrontation between New York, an upper basin state, and the lower-basin states of New Jersey and Pennsylvania. New Jersey sued in the Supreme Court which applied the doctrine of equitable apportionment with New York receiving about two thirds of what New York City had originally requested. This was later increased but not surprisingly, none of the states were satisfied with the Supreme Court ruling (Dellapenna, 2005).

The equitable apportionment ruling did not create a comprehensive integrated basin management system nor could the Court return to the allocation plan every time a new issue emerged. As a result, in 1936, the three states created the Interstate Commission on the Delaware River Basin (INCODEL). This commission was developed without Congressional approval, indicating that neither the states nor the federal government considered this agency a major player in the basin. After the INCODEL failed, Delaware joined the other three states in the basin in proposing the Delaware River Basin Commission Compact. Adopted by the states in 1949, it went into effect with congressional ratification in 1952. This compact also failed. The Commission lacked the powers needed to carry out the goals and objectives of the compact. Specifically, the Commission had no authority to regulate water usage even though different uses might interfere significantly with the plans of the agency. Shortly after that compact went into effect, the states began negotiations for the second compact, the current Delaware River Basin Compact (DRBC) (Dellapenna, 2005).

The current compact was approved by Congress in 1961. In addition to the inclusion of the federal government as a full partner, a major strength of this compact is that it is administered by a commission with broad powers to carry out its responsibilities. These powers include the critical ability to borrow money and issue bonds, giving the commission the wherewithal to maintain a necessary amount of independence. Other successful aspects of the DRBC include the ability to aid in the coordination and integration of federal, state, municipal, and private agencies and the development of a comprehensive plan addressing both immediate and long range water resource needs (Delaware River Basin Compact, 1961; Muys, 1995). In addition, the DRBC also recognizes the overarching importance of allocating water equitably, without regard for artificially imposed borders; “…to apply the principle of equal and uniform treatment to all water users who are similarly situated and to all users of related facilities, without regard to established political boundaries” [emphasis added] (Delaware River Basin Compact, 1961, Article 1, § 1.3, ¶ (e)).

This new compact resulted in a commission that was a regulatory agency in addition to its planning and operational functions. The new commission also has extensive authority for hydropower development, pollution regulation, watershed management, and the development of flood protection and recreational facilities in addition to its former mandate to provide public water supplies (Delaware River Basin Compact, 1961; Dellapenna, 2005).

Most importantly, for states concerned with issues of autonomy and sovereignty, it should be noted that in two important ways, the DRBC’s regulatory system is more limited than those of the states in the basin. First, water withdrawal permits are only needed in “protected areas” where water demand results in a shortage or interferes with the Commission’s comprehensive plan. These permits can be reviewed in any court with competent jurisdiction. Second, the authority to issue withdrawal or diversion permits rests with those states with an effective water use permitting system. In a water emergency, however, state permits may be superseded (Delaware River Basin Compact, 1961; Dellapenna, 2005). Although a number of suits have been brought against it, the success of the DRBC was such that, in 1970, it became the template for the Susquehanna River Basin Compact (SRBC) (Dellapenna, 2006).

COMPACTING CHALLENGES

Compacts can and do fail. There are a variety of barriers to developing and implementing successful interstate water compacts. There are often diverse cultural, political, historic and economic priorities that each group brings to the negotiations. Parties to these types of agreements must often cooperate and collaborate with others of widely divergent interests (Mulroy, 2008).

Developing and implementing an interstate compact can be a complicated, expensive, and time consuming project
(Burke, 2004; Meyers cited in Stephenson, 2000). Because of often substantial federal interests in the areas covered by compacts, these agreements must also account for the participation of these and other stakeholders (Mandarano et al., 2008; Sherk, 2005). In some cases, state elections, especially those for governor, may temporarily interrupt the administration of a compact given the relationship between that office and a compact commission. Ambiguous language and unresolved issues can also plague a compact (Burke, 2004) while a lack of accurate data and faulty or no planning for future development can threaten to derail elements of it years down the road (McClurg, 1997).

A major issue in compact development is that states are often reluctant to delegate significant authority to a regional commission or other authority that they realize may not always act in their best interest. Muys (1995) points out that states should take into consideration that as they are more able to restrain compact agencies to protect their sovereignty, they are also increasing the likelihood that regional water issues will escalate to the point that they will come under federal jurisdiction, overriding state or local authority.

The ACT and ACF Compacts

The Alabama-Coosa-Tallapoosa (ACT) and the Apalachicola-Chattahoochee-Flint (ACF) compacts are examples of failed efforts to find a solution to a growing water crisis. Conflicts over water between Georgia, Alabama, and Florida (the ACF) and between Georgia and Alabama (the ACT) had resulted in a prolonged attempt to develop and implement an interstate water compact. Although deadlines for compact development were extended several times, the states were unable to reach a compromise and no effective compact has emerged (Dellapenna, 2006). This failure can be attributed to several problems associated with water compacts.

Primarily, the states relentlessly protected their own interests and failed to negotiate in good faith (Mandarano et al., 2008; Stephenson, 2000). In addition, these compacts (ACT and ACF) lacked many of the attributes that made the DRBC and the SRBC so effective (Dellapenna, 2006). For example, while the DRBC Commission has the power to allocate waters to and among the compact states and to impose conditions, the ACT/ACF Commission was limited to planning, coordinating, monitoring, and making recommendations concerning the water resources of the basin (Delaware River Basin Compact, 1961; Alabama-Coosa-Tallapoosa River Basin compact.1997; Apalachicola-Chattahoochee-Flint River Basin compact.1997). Another problem with the ACF compact centers on the treatment of federal agencies (Dellapenna, 2006). Given the huge federal expenditures in the basin, in excess of $1.5 billion just for the Army Corps of Engineers, the proposed compact called only for minimal federal participation, an unacceptable situation for the U.S. Department of Justice (Reno cited in Sherk, 2005).

Some of the problems with the ACT/ACF were not related to compacts per se but are important in the negotiating process. First, there were problems with negotiating in a public forum. Stakeholders representing various organizations were present and were unable to compromise in many cases. Politically, there was fallout for current office holders as no matter what the outcomes, a number of stakeholders were not going to agree. There were also technical issues such as regulating flow versus regulating consumptive uses. Georgia was willing to talk about one but not both. Finally, Georgia negotiated from the position that they needed far more water than the other states (Kerr in Burke, 2004), a position that may have been hard to sell to Alabama and Florida.

The Colorado River Compact

Not the stunning success of the Delaware River Basin Compact nor the abysmal failure of the ACT/ACF, the 1922 Colorado River Compact continues to be a source of controversy. The primary purposes of the compact included dividing the river flow between the states of the Upper Basin (Colorado, Wyoming, Utah, and New Mexico) and the Lower Basin (Arizona, California, and Nevada), eliminating future disputes, and promoting the orderly development and management of the river (Colorado River Governance Initiative, 2010). However, the number and scope of “agreements, contracts, treaties, laws, and court decisions” (McClurg, 1997, p. 7) that make up “the law of the river” governing the Colorado today, indicate that there was a great deal of ground not covered in the original compact. These topics include environmental issues, increasing development, growing water shortages, water transfers, the rights of Native Americans to water, and a possible dispute with Mexico over water promised by treaty in 1944.

The Colorado River Compact was finally ratified as part of the Boulder Canyon Project Act of 1928, authorizing the construction of the Hoover Dam and apportioning the water of the lower basin between the states. A 1944 treaty with Mexico further apportioned the river and in 1948, the Upper Colorado River Basin Compact allocated the Upper Basin apportionment by percentages between participating states. Court cases and negotiated settlements delineate tribal rights whose allocations are taken from the state in which the reservation is located. The Law of the River also includes Congressional authorization for a number of water projects such as the Colorado River Storage Project Act in 1958 which provided an Upper Basin Development Plan and construction of the Glen Canyon Dam (Lake Powell). Even with the compact, the Supreme Court has had to step in and specify how much water each state was entitled to (See Arizona v. California, 1963). The Court has revisited the issue numerous times, the last time in 2006 (See Arizona v. California, 2006). In addition, there are also a number of national and regional environmental laws that are part and parcel of the Law of the River (Colorado River Governance Initiative, 2010).

Given that, on average, water demands have exceeded water supplies in the Colorado River Basin over the past decade, there is little doubt that changes will need to be made. In the future, decreased water flow due to even a modest change in climate will be problematic. At high levels of climate change, the lack of water will become disastrous (Colorado River Governance Initiative, 2010; Robison and
MacDonnell, 2014). In the event of a compact call, “Not only might the Law of the River prove unmanageable, but it may actually collapse under the weight of the situation” (Colorado River Governance Initiative, 2010, p. 18).

Other Challenges

The American Central Plains and Southwest regions are currently suffering from extremely warm and dry conditions which are expected to continue for decades (Cook, Ault, and Smerdon, 2015). The Rio Grande is now being reduced to “a trickle” due to lack of rain and continued consumption by both metro and agricultural users. Arizona is preparing for future cuts in its allocation from the Colorado River should the water level in Lake Mead continue to fall (Wines, 2015). On April 1st of this year, Governor Jerry Brown of California announced mandatory water restrictions to help address the current drought (Nagourney, 2015).

Some other problem spots for water resource management include the Catawba River between North and South Carolina as well as the Savannah River between South Carolina and Georgia. On the Catawba, it appears that neither state is willing to compact and disputes have already erupted, lessening the chance of a viable compact in the near future (Dyckman, 2008). In each case, critical decisions will have to be made about water resources and one of the best ways to do this will often be through interstate compacts.

SOUTHERN CAROLINA: A CASE STUDY OF PROSPECTIVE COMPACT DEVELOPMENT

In addition to providing drinking water, water for industry, irrigation, hydropower, waste assimilation, transportation, and flood control, South Carolina’s water ways also provide habitats for fish, wildlife and plant species as well as migration routes critical for species reproduction (Wachob, Park, and Newcome, 2009). Pressure is increasing on these resources due to population growth as well as changes in how water is used. In previous centuries, water use was, for the most part, limited to instream and non-consumptive uses - e.g., transportation, hydro-mechanical power, and fishing. Burgeoning technology, however, has brought new and more consumptive uses. In 2006, thermolectric power, for example, was second only to hydroelectric power in water use in this state, utilizing some 3.5 trillion gallons of surface water (Holman, 2008).

South Carolina, along with North Carolina and Georgia, is facing a number of critical water resource issues. These include but are not limited to water allocation, water quality, drought management, salt water intrusion, assimilative capacity, stream flow maintenance, ground water usage, and flood control (Catawba-Wateree Basin Advisory Commission, n.d.; Savannah River Basin Advisory Council, n.d.; Wachob et al., 2009). These issues are important in that efforts by one state to address a problem often impacts another state in negative ways. For example, ground water pumping to support development or combat water shortages in one state can lead to salt water intrusion in another (Wachob et al., 2009).

South Carolina has four major river basins, three of which it shares with neighboring states. The two largest, the Yadkin–Pee Dee and the Catawba–Santee (aka the Catawba-Wateree) are shared with North Carolina, The Savannah Basin is shared with Georgia with a small, northernmost portion located in North Carolina. The final basin, the Ashepoo-Combahee-Edisto is located entirely within South Carolina (Badr, Wachob, and Gellici, 2004) and is not subject to transboundary issues with another state.

Of the three states - North Carolina, South Carolina, and Georgia - South Carolina is currently the least populated and is growing at the slowest rate. Even so, the state is predicted to gain over a million people between 2000 and 2030. North Carolina and Georgia are both more populous and growing at considerably higher rates (See Table 1). From 2000 to 2030, according to predictions, North Carolina will gain just over 4 million people and Georgia just under 4 million (U.S. Census Bureau, Population Division, 2005). Given the population differences across the states and the needs of these populations for water as well as the desire for South Carolina to grow, a solution will be needed that balances these factors and the water resources equitably.

Even though it has a smaller and slower growing population, South Carolina’s water resources are heavily impacted by its faster growing neighbors. During drought conditions, for example, both North Carolina and Georgia increasingly rely on rivers shared with South Carolina. Coastal cities such as Myrtle Beach depend on water supplies from North Carolina and have experienced shortages. These conditions make maintaining stream flow a major challenge and increase the probability of conflict between states sharing these resources (Burke, 2004, p. 296; Holman, 2008; Wachob et al., 2009).

In 2009, South Carolina confronted North Carolina in the U.S. Supreme Court regarding proposed water withdrawals from the Catawba River. Interbasin transfers in North Carolina endanger water quality and flow in the coastal areas of South Carolina (League of Women Voters of South Carolina Water Resources Study Committee, 2011). This can be seen in South Carolina’s Yadkin-Pee Dee Basin where the river is impacted by the upstream needs of six reservoirs, all of which are located in North Carolina. At the same time, salt water incursion into the lower Pee-Dee River has resulted in a need for increases in the minimum

### Table 1. Population Projections for North Carolina (NC), South Carolina (SC) and Georgia (GA): 2015-2030.

<table>
<thead>
<tr>
<th></th>
<th>2015</th>
<th>2020</th>
<th>2025</th>
<th>2030</th>
<th>Change 2000 to 2030</th>
</tr>
</thead>
<tbody>
<tr>
<td>NC</td>
<td>10,010,770</td>
<td>10,709,289</td>
<td>11,449,153</td>
<td>12,227,739</td>
<td>51.9%</td>
</tr>
<tr>
<td>SC</td>
<td>4,642,137</td>
<td>4,822,577</td>
<td>4,989,550</td>
<td>5,148,569</td>
<td>28.3%</td>
</tr>
<tr>
<td>GA</td>
<td>10,230,578</td>
<td>10,843,753</td>
<td>11,438,622</td>
<td>12,017,838</td>
<td>46.8%</td>
</tr>
</tbody>
</table>

Note: Adapted from the U.S. Census Bureau, Population Division, Interim State Population Projections 2005.
flow required from North Carolina (Wachob et al., 2009). Meanwhile, in Georgia, Atlanta is seeking potential water sources that include Lake Hartwell (League of Women Voters of South Carolina Water Resources Study Committee, 2011), on the border between South Carolina and Georgia, and part of the Savannah River Basin.

The Savannah River Basin

The Savannah River begins in North Carolina, forms the boundary between Georgia and South Carolina and empties into the Atlantic at the port of Savannah. The river basin has a number of important issues that will either require cooperative efforts between the states or may escalate into litigation. Among these are water quality issues, drought, economic development and population growth, fish and wildlife concerns, regulatory issues and the Savannah Harbor Expansion Project (Georgia Environmental Protection Division and SC Department of Health and Environmental Control, n.d.; SC Savannah River Basin Advisory Council, n.d.).

In a 2004 report, the South Carolina Governor’s Water Law Review Committee (GWLRC) supported a compact with Georgia as a viable method to apportion the resources of the Savannah River Basin. However, while recognizing that both states have an interest in the entire river and that there is a need for consistency between the states in areas such as water quality standards and FERC relicensing (Governor’s Water Law Review Committee, 2004), it does not appear that there are any recommendations for a strong, resilient, basin-wide governing body similar to those found in more successful water compacts. In fact, the GWLRC specifically suggests that the compact utilize various protocols that would “obligate each state to manage its basin resources in a consistent manner” (Governor’s Water Law Review Committee, 2004, p. 24) but carefully avoids any commitment to common governance. That being said, the GWLRC has highlighted a number of elements that may contribute to the development and ratification of a successful Savannah River Compact.

When discussing the allocation of the usable water, the Committee acknowledges the many stakeholders involved, including the significant role of the Army Corps of Engineers (CoE) and other federal agencies. In case of drought, cooperation and coordination with the CoE will be essential since they control significant resources on the river. Another positive element from the GWLRC report is the recognition of the importance of accurate data. Unlike the Colorado River Compact, where the river was over-allocated from the beginning, having a realistic estimate of the available water supply can only enhance the working of any compact that may emerge.

The GWLRC proposal also advocates addressing the looming issue of interbasin transfers. It specifically notes that while Greenville and Beaufort-Jasper together are permitted to access 210 million gallons per day from the basin, Georgia also has the potential for a very large transfer from the Savannah. Again, inclusion of the CoE, who also oversees a major supplier of water to Atlanta, Lake Lanier, can, at the very least increase the scope, accuracy and reliability of the knowledge available and significantly improve any compacting efforts.

Although the GWLRC acknowledges that various state and federal agencies with interests in the environment conduct activities within the Savannah River Basin, there is no evidence of real concern about the environment itself. The only mention of the Clean Water Act is related to FERC relicensing and there is one mention of endangered species. In reality, the Clean Water Act and the Endangered Species Act will significantly impact the way states will manage the Savannah River resources. The possible consequences of these laws must be incorporated into any viable compact. In addition to support from the GWLRC, the 2004 South Carolina Water Plan (Badr et al.) also calls for the development of a compact between the state and others that share water resources. “Compacts”, the authors point out, “will promote interstate coordination, reduce potential disputes between the states, enhance the flow regime of many of South Carolina’s rivers and extend the availability of water during severe droughts” (p. vi).

Like its South Carolina counterpart, the Georgia Comprehensive State-wide Water Management Plan (Georgia Environmental Protection Division, 2008) recognizes the need for flexibility, the importance of including various stakeholders, and the need for relevant and accurate data. However, unlike the South Carolina plan, there is no mention of the possibility of a compact or any coordinated effort with South Carolina regarding the Savannah River.

The Savannah River Basin Water Caucus, a joint effort between South Carolina and Georgia, is composed of legislators from counties on both sides of the river. A major purpose of the Caucus is to avoid lengthy and costly litigation between the states as South Carolina threatens action against Georgia over water allocation. While there has been mention of an interstate water compact for the Savannah Basin (Cary, 2013), it is too early in the process to determine if this option will actually reach the Caucus’ agenda.

An earlier effort, the Savannah River Basin Partnership between the Georgia Environmental Protection Division and the South Carolina Department of Health and Environmental Control was established by Governor Sanford of South Carolina and Governor Perdue of Georgia in 2005. Major topics for this group included salt water intrusion into the Upper Floridan aquifer, dissolved oxygen standards along with associated Total Daily Maximum Loads (TDML), and sustainable water use in the basin. Currently, the status of shared planning for this group includes the previously mentioned Georgia Comprehensive State-Wide Water Management Plan (Georgia Environmental Protection Division, 2008) and the South Carolina Water Plan (Badr et al., 2004). There’s no indication of comprehensive basin-wide planning by the two states (Georgia Environmental Protection Division and SC Department of Health and Environmental Control, n.d.). In April of 2012, Governor Haley of South Carolina signed Executive Order 2012-05,
re-establishing the Governor’s Savannah River Committee of South Carolina, initiating another round in the South Carolina/Georgia talks.

**The Catawba-Wateree and Yadkin-Pee Dee Basins**

In the face of what will likely become critical water shortages, North and South Carolina have both developed legislation supporting River Basin Advisory Commissions for the Catawba-Wateree and Yadkin-Pee Dee basins (North Carolina General Assembly, n.d.; South Carolina General Assembly, 2004). Very similar to the process for developing a compact, each state adopted legislation that specifies the scope of each commission’s work as well as the composition, responsibilities, and powers of the commissions. In fact, there are many similarities between the legislation for these commissions and the Utton Center’s model compact (See Muys et al., 2007). A major difference however, is that by law, the commissions are advisory only in nature: There are no provisions for regulatory or other administrative authority.

It is important to note that the Catawba-Wateree River Basin Commission (CWRBAC or the Basin Commission) provides an example of a unified approach to basin management much like those found in the most successful compacts. Briefly, the Basin Commission provided a platform for South Carolina and North Carolina, along with Duke Energy, the Catawba River Water Supply Project, and other stakeholders to negotiate an agreement to resolve South Carolina v. North Carolina without further litigation and expense (See South Carolina v. North Carolina settlement agreement, 2010). The Settlement Agreement reflects the joint nature of the negotiations, especially given the alternative of further litigation -

…by reaching this Agreement the Parties will achieve a better result than could be achieved through the Litigation with a substantial cost savings to the taxpayers and ratepayers in both States. The Parties also believe that it is important that the States regard each other as close neighbors, which share the Catawba-Wateree River (“River”), rather than as a plaintiff and a defendant in a lawsuit and that this Agreement will be a model for regional cooperation (2010, p. 1).

In adopting a common approach to managing the resources of the river basin, North Carolina and South Carolina have taken an important first step toward an eventual compact should one be desired.

While the Catawba-Wateree Basin Advisory Commission has been active since its initial development, there is insufficient evidence to indicate that a viable Yadkin-Pee Dee Basin Advisory Commission has emerged. Perhaps the issues that led to South Carolina v. North Carolina acted to spur the creation and maintenance of the Catawba-Wateree Basin Commission whereas the Yadkin-Pee Dee has not yet reached that critical state.

**CONCLUSIONS AND RECOMMENDATIONS**

Georgia, South Carolina, and North Carolina are facing water resource management issues that are becoming increasingly common, even east of the 100th meridian. It is critical now that states develop a method or methods for solving their differences that reach beyond their tendency, and that of their agencies, to protect their own interests ahead of those of the basin. By utilizing interstate water compacts with adequate power and resources to carry out comprehensive planning, coordination, and management (Hayton and Utton, 1989), the basin itself, in the form of the commission, becomes a principal actor. Depending on the organizational structure, either state agencies or agencies developed and implemented by the commission itself are responsible for carrying out the mandates of the compact within the basin. Monitoring (See Ostrom, 1990), transparency, and accountability reduce agency costs and promote trust (See Gortner, Nichols, and Ball, 2007) in the commission even as it acts as an agent for the stakeholders in the basin.

Developing a compact can be a long and complex process. The time to start is now. In riparian states, the state with the fastest growth may have an advantage in court cases, especially when the faster growing state has already appropriated water. In these instances, courts may be unwilling to limit existing diversions (Kansas v. Colorado, 1907; Burke, 2004). If it’s true, as Burke suggests, that Metro Atlanta’s position as the fastest growing area in the region gives Georgia an advantage over Alabama with regard to the Chattahoochee (2004), then Georgia will also have the advantage over South Carolina regarding the Savannah. In addition, if the issues between North Carolina and South Carolina move to litigation in the Yadkin-Pee Dee Basin, South Carolina, because of its slower growth may be at a disadvantage there, too.

South Carolina and North Carolina have shown that they can work together in the Catawba-Wateree Basin. The next move forward is to begin conversations about the possibilities of developing federal-interstate compacts utilizing the components that have been so effective on the Delaware and Susquehanna river systems. As contracts, these compacts can provide protection from both prior and subsequent legislation (Florestano, 1994) and with the federal government as a partner, they can garner cooperation from a major player in water resource management.

Given Georgia’s history with Alabama and Florida, South Carolina should immediately exert concentrated and sustained efforts to start negotiations toward the development of a compact with that state. This effort should be made in good faith and with the understanding that each state has the right to an equitable utilization of the water resource as well as a duty to avoid appreciable harm to a co-basin state (See Hayton and Utton, 1989, p. 672). At the same time, South Carolina should also consider taking the initial steps prior to filing an action in the Supreme Court for equitable apportionment. This could provide leverage for South Carolina while encouraging Georgia to negotiate (Holman, 2008).
Compacts are not the solution for every interstate water problem. However, they do supply a platform that can be tailored to each unique situation and they can also have built-in flexibility to deal with issues that have not yet emerged when legislation is passed enabling the compact. By treating a river basin as a political entity in its own right, a compact can provide local stakeholders control of a resource that impacts so many water users in so many ways.

ACKNOWLEDGEMENTS

I would like to thank the reviewers for their helpful comments and I would especially like to thank Dr. Jeff Allen, Director of the South Carolina Water Resources Center at the Strom Thurmond Institute for his insights, encouragement, and humor. Thanks also to the journal staff, Dawn Anticole White of Clemson University, for her kindness in keeping things on track.

LITERATURE CITED


Arizona v. California, 547 U.S. 1, U.S. Supreme Court 2006.


Colorado v. Kansas et al., 320 U.S. 383 (U.S. Supreme Court 1943).


Abstract. A watershed-based plan was recently developed for Murrells Inlet, a moderately tidal, euhaline estuary located on the northern coast of South Carolina. One of the goals of this planning effort was to collate and analyze existing data to refine assessments of the sources of fecal coliform detected by SC DHEC’s shellfish monitoring program. Coastal Carolina University’s Waccamaw Watershed Academy (WWA) was engaged to lead this data analysis effort. The most important sources identified were urbanized wildlife and canines. Results from the data analyses were used to prioritize subwatersheds for remediation. This has led to proposed strategies that focus on interception and treatment of stormwater runoff as well as volume reduction, dredging of tidal creek sediments, and outreach education for pet waste control.

INTRODUCTION

Murrells Inlet is a moderately tidal, euhaline estuary located on the northern coast of South Carolina (Figure 1). The watershed encompasses 3748 hectares with 2560 hectares comprised of land draining into the estuary. The remaining area consists of open water, intertidal mudflats and marsh habitat. SC DHEC estimates that 1258 hectares are suitable habitat for production of shellfish (SC DHEC 2014). Shellfish harvesting is approved in 71% of this area and administratively classified as “Prohibited” in 5% due to the presence of marinas. In the remaining 24%, harvesting is restricted due to elevated fecal coliform levels reported from monitoring conducted by SC DHEC under their shellfish sanitation program.

TMDL’s are a tool under the federal Clean Water Act to help bring polluted waters into compliance with water quality standards, thereby enabling designated uses, such as shellfishing, to be supported. A Total Maximum Daily Load (TMDL) was approved by SC DHEC in 2005 to address these long-standing fecal coliform impairments (SC DHEC 2005). To implement the TMDL, a watershed plan was developed in 2014 that specifies prioritized actions to reduce loading of fecal bacteria into Murrells Inlet. These were developed from a detailed review of land use, watershed dynamics, regulatory controls, previous efforts at source assessment, and a new set of statistical analyses performed on SC DHEC’s shellfish monitoring data and the data collected by the Murrells Inlet Volunteer Water Quality Monitoring (VM) program. Details on the review of land use, including a change analyses, can be found in Fuss et al. (2014), watershed dynamics in Williams et al. (2014), and the regulatory context in Newquist et al. (2014).

In this paper, we review previous microbial source tracking (MST) work and discuss the new statistical analyses that were performed to prioritize locations and strategies for remediation. These analyses used SC DHEC and VM monitoring data to identify locations of the most significant fecal bacteria sources and transport pathways. This information comprised Element D.I and Appendix D in the watershed plan (Newquist 2014). The plan was approved by SC DHEC in 2014 and used by the stormwater managers in Horry and Georgetown Counties to obtain funding from the USEPA 319 program to support implementation of stormwater treatment practices in the subwatersheds prioritized by the statistical analyses.

BACKGROUND AND RELATED WORK

Regulatory Context. TMDL’s generally specify a quantified load reduction that once implemented will bring the impaired waters into regulatory compliance with water quality standards. Implementation of TMDL’s had been voluntary until the advent of the Municipal Separate Storm Sewer System (MS4) permit program which now requires permittees to take actions to bring waterbodies in their jurisdictions into compliance with water quality standards.

TMDLs must be developed by the states and approved by USEPA within 13 years of initial listing of a site on
the 303(d) list of impaired waterbodies. This listing is a consequence of frequent contraventions of water quality standards. In the case of shellfish waters, the standard is based on the National Shellfish Sanitation Program (NSSP) requirements that are used to determine if a shellfish bed can be approved for shellfish harvesting, i.e., the geometric mean (geomean) of fecal coliform concentrations is less than 14 MPN/100 mL and the estimated (Est.) 90th percentile is less than 43 MPN/100 mL (US FDA 2009)\(^1\). SC DHEC uses three years of data to generate these statistics in which samples are collected approximately once per month to generate a minimum of 30 and a maximum of 36 samples for each monitoring site. Sampling dates are randomly selected with respect to tidal stage and weather. Sampling sites in Murrells Inlet were originally selected in a stratified random manner. But due to increasingly limited resources, SC DHEC has adjusted their sampling over the decades to more closely define boundaries of closed shellfish beds, thereby minimizing the total area closed to shellfishing. This strategy has led to a decline in the number of relatively “clean” sites now being monitored.

In 2008, the USEPA recommended a watershed-based framework for TMDL development as opposed to using a single-segment approach. The goal is to provide a framework for more efficiently addressing the maximum number of impairments in a scientifically defensible manner (USEPA 2008). The original TMDL approved for Murrells Inlet encompassed eight sites that were determined to be influenced by drainage from three subwatersheds. Over successive 303(d) reporting periods, SC DHEC increased the number of monitoring sites covered by the fecal coliform TMDL to 20, with some of the additional sites having become impaired after 2005 (Table 1). Most of these additional sites are influenced by drainage from subwatersheds other than the three for which load reductions were specified in the TMDL approved in 2005. Those load reductions were approximately 71 to 81%. They were determined by modelling monitoring data collected from 2001 to 2004. The host animal source assessment was qualitative and concluded that wildlife was the most significant source. The TMDL approved in 2005 did not recommend any means by which the load reductions could be attained.

**Microbial Source Tracking Investigations.** Efforts have been undertaken to identify sources and transport pathways of the fecal bacteria in Murrells Inlet. These have included: (1) assessment efforts conducted from 2005 to 2006 associated with the development of a Special Area Management Plan (SAMP) by SC DHEC Ocean and Coastal Resource Management (OCRM) (Bennett 2007); (2) volunteer water quality monitoring initiated in 2008 (Libes et al. 2012); (3) microbial source tracking using multiple antibiotic resistance and GIS modeling (Kelsey et al. 2003; Kelsey et al. 2004); and (4) spatial surveys conducted by SC DHEC, Georgetown County, Coastal Carolina University, and the volunteer water quality monitors (all since 2010). The latter included measurement of fecal bacteria in sediments (Anderson and Greoski, 2010). A multiple tracer study performed in the northern end of Murrells Inlet using genotypic assays was completed in 2013 and hence was not available for inclusion in the watershed plan (Sturgeon et al. 2014).

The work of Kelsey et al. (2003 and 2004) supported stormwater transport as the major pathway by which fecal coliforms are being conveyed into Murrells Inlet. Microbial antibiotic resistance measurements suggested that the major host animal sources were wildlife. The work of Sturgeon et al. (2014), conducted on samples collected from the north end of Murrells Inlet in 2012 and 2013, confirmed that humans are not a significant source, but that canines, inclusive of coyotes, and aquatic birds were significant sources. The work of Bennett (2007), which was designed to assess the efficacy of two stormwater treatment practices, documented elevated levels of fecal bacteria at nearby control sites. These were located in two tributary streams discharging into Murrells Inlet. A volunteer water quality monitoring program was then instituted to further investigate the role of small tributary streams as a significant source of fecal bacteria to the creeks of Murrells Inlet.

The volunteer monitors sample sites located at the terminus of six small tributary streams and at two shore-based sites in the Inlet proper, with one in the north (Horry County) and one in south (Georgetown County). See squares in Figure 1 for sampling site locations. Sampling has been conducted bimonthly since July 2009 for fecal bacteria (*E. coli* and total coliform using Micrology's Easygel™ dual confirmation media). This monitoring documented that several of the tributary streams frequently had high levels of fecal bacteria (Table 2). Fecal bacteria were

<table>
<thead>
<tr>
<th>Data years</th>
<th>Number of TMDL sites</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000 to 2004</td>
<td>8</td>
</tr>
<tr>
<td>2002 to 2006</td>
<td>9</td>
</tr>
<tr>
<td>2004 to 2008</td>
<td>17</td>
</tr>
<tr>
<td>2006 to 2010</td>
<td>20</td>
</tr>
<tr>
<td>2008 to 2012</td>
<td>20</td>
</tr>
</tbody>
</table>

1 These are the Class SFH (shellfish harvesting) water quality criteria specified in SC R.61-68. Concentrations are reported as Most Probable Number (MPN) per 100 mL of sample. The Est. 90th percentile is used to minimize the impact of rare random pollution events that could skew the 90th percentile because of a few high MPN values.

2 This information is available at: [http://www.scdhec.gov/HomeAndEnvironment/Water/ImpairedWaters/Overview/#4](http://www.scdhec.gov/HomeAndEnvironment/Water/ImpairedWaters/Overview/#4). Although the TMDL sites are not on the 303(d) list, SC DHEC provides an additional table.
Figure 1. Subwatersheds of Murrells Inlet as delineated by Williams et al. (2014). Boundaries are shown by blue lines. Names are shown in the boundaries. Subwatersheds in pink were prioritized for remediation as a result of the data analyses. Currently active SC DHEC monitoring sites (green dots) are labelled with the names of the subwatersheds contributing drainage. Labels in red indicate sites with highest fecal levels. Volunteer monitoring sites (square dots) are shown in red for those with high fecal bacteria levels and in green for those with typically low levels.
detected in the sediments of these small tributary streams, with concentrations being highly variable over space and time (Anderson and Greoski, 2010). Concurrent water measurements performed at upstream sites under wet and dry conditions generated similar results, suggesting episodic inputs from wildlife living in the stream corridors. This finding was also observed by other spatial surveys conducted over the years by SC DHEC, Georgetown County, and the volunteer monitors.

**Watershed Plan Development.** Concern voiced by the volunteers over their findings led the local community group, Murrells Inlet 2020, to lobby for development of a watershed plan (Young et al. 2014). In 2012, SC DHEC awarded the Waccamaw Council of Governments (COG) US EPA Section 319 funding to lead development of a watershed-based plan. The primary goal of the plan was to outline strategies for achieving fecal coliform load reductions. The COG developed this plan collaboratively with a steering committee comprised of stormwater managers from Horry and Georgetown counties, Murrells Inlet 2020, volunteer water quality monitors, Earthworks, Inc., scientists from Coastal Carolina and Clemson Universities, and concerned members of the community (Newquist 2014).

The plan was approved by SC DHEC in 2014. It includes a detailed list of prioritized fundable remediation projects designed to reduce fecal coliform loading to Murrells Inlet. These projects were developed from an understanding of fecal sources and transport pathways obtained from a comprehensive review of all existing data and prior microbial source tracking efforts.

### PROJECT OBJECTIVE/GOALS

The primary objective of the data review was to assess all existing information to obtain a state of the knowledge understanding of fecal bacteria sources and transport pathways in the Murrells Inlet watershed. The watershed plan had to be developed in one year, so no new data could be collected. Thus an additional goal was to identify crucial data gaps and develop action items to include in the plan for obtaining these data.

Statistical analyses of existing fecal bacteria data from SC DHEC and the volunteer water quality monitoring program were designed to answer the following questions as posed by the steering committee that formulated the watershed plan:

1. Are some impaired locations more problematic than others, i.e. which sites have persistently elevated concentrations of fecal bacteria?
2. What are the ultimate source(s) and transport process(es) contributing to the bacterial water quality impairments?
3. Why are some sites attaining water quality criteria and others not?
4. Where and why has the acreage of shellfish closures been changing, i.e., have the fecal bacteria levels at any sites increased or decreased over time? If so, what has been causing these trends?

Ultimately these questions were used to prioritize subwatersheds for remediation efforts. Drivers of fecal bacteria trends that were evaluated to help answer these questions included: rain, i.e., transport via overland runoff due to stormwater flows, as well as tides, salinity, and changes in land use/land cover.

It was hypothesized that

1. Rainfall is a major transport agent of fecal bacteria, i.e., fecal bacteria concentrations would be higher in samples collected following rainfall as compared to antecedent dry periods and fecal bacteria concentrations would be higher in samples of lower salinity.
2. Sites with higher fecal bacteria levels are located immediately downstream of the most urbanized subwatersheds.
3. Shellfish beds not subject to closure are located furthest from land.
4. Fecal bacteria concentrations have increased in subwatersheds where urbanization has increased.
5. Tidal flushing reduces fecal bacteria numbers, i.e., fecal bacteria concentrations are higher in samples collected during low tide as compared to high tide, and fecal bacteria concentrations are higher in samples of lower salinity.

### METHODS

Bacteria concentration data from SC DHEC’s shellfish monitoring program (1967-2011) and a local volunteer water quality monitoring program (2008-2012) were used to elucidate spatial and temporal trends in bacteria levels and their causative drivers. Other ancillary data evaluated...
included rain, salinity, subwatershed boundaries, and land use-land cover. Details are provided below including inherent limitations in the fecal bacteria indicator data as they relate to sample site location and sampling frequency.

The steering committee that developed the watershed plan participated in selection of statistical tests, reasonable assumptions, and modes of data presentation including GIS mapping. This committee also reviewed the results and collaboratively crafted summary conclusions that were incorporated into the watershed-based plan. A technical advisory committee provided peer review of the data analyses.

Data Sources. Watershed mapping was performed by Earthworks, Inc. This included delineation of 51 subwatersheds and mapping of flow paths, soils, and impervious surface. These were used to generate maps of curve numbers (Williams et al. 2014). Using these maps, Georgetown County performed peak flow determinations for 2-year design rain events using the TR-55 model that is designed for small urbanizing watersheds. Comparison of land-use land cover maps for 1994 and 2012 was used to identify subwatersheds that had undergone recent significant urbanization (Williams et al. 2014).

Statistical tests were performed on fecal coliform data from SC DHEC’s National Shellfish Sanitation Program collected from 1967 to 2011. This was the entire period of record that SC DHEC could provide within the project time frame. Data collected after 2011 were used in some analyses as they became available.

Mike Pearson (SC DHEC) provided shellfish program data from 1990 to present, including salinity and tidal stage. To interpret the tidal stage information, SC DHEC provided their field sheet coding information. Legacy data were obtained by download from STORET. The earliest data obtained are from 1967. Significant data gaps are present for periods that Mike Pearson suggests should have data. For some years, no data are present for any of the sites. Another limitation to the data collected prior to the early 1990’s includes a change in analytical methodology. This involved a delay in adoption of a modernized version of the look-up tables provided in Standard Methods used to transform tube counts into MPN/100 mL. Some of the legacy data suggest that special studies were done such that multiple samples were collected per month and in some cases, per day.

Statistical tests were also performed on E. coli and total coliform data collected at eight sites by the Murrells Inlet volunteer water quality monitoring program from 2008 to 2012. Ancillary data used from this monitoring program included conductivity. Most of these sampling sites are located in freshwater tributary streams that discharge into saline tidal creeks, so salinities are generally below 5. This is also why the freshwater fecal indicator bacteria, E. coli, is measured by this program. The National Shellfish Sanitation Program requires the use of fecal coliforms as the fecal indicator bacteria.

Daily rain accumulations from the nearest National Climate Data Center (NCDC) monitoring station was used to obtain a rain record back into the 1960’s. This sampling site (COOP:381093) is located 1.5 miles inland in Brookgreen Gardens, Murrells Inlet, SC. We acknowledge that the highly localized nature of rainfall along the southeastern coast, especially during the summer, limits the usability of this source. The binning of data on a daily basis also creates limitations in interpreting the fecal coliform data.

Data Analyses. All hypothesis testing relied on nonparametric approaches since the fecal bacteria data are not normally distributed. Nonparametric tests are less powerful than the analogous parametric tests, making it more difficult to detect significant trends or differences. This leads to a conservative reporting of significant differences or trends. In other words, absence of significance does not mean the differences or trends were not present; they just couldn’t be detected by the nonparametric test.

All statistical analyses and graphing were performed with Microsoft’s Excel 2007 and Systat’s SigmaPlot V12.3. Mann-Kendall tests for time series trends were performed with code downloaded from Helsel et al. (2006).

Results were presented as time trend graphs, box plots, bar graphs, scatter plots, and matrices. All the statistical test results were collated by site into a summary matrix to provide a weight-of-evidence approach to support overarching spatial and temporal trends. In this summary matrix, sites were grouped by subwatersheds and information on peak runoff and land use/land cover were included to provide insight into terrestrial drivers of spatial trends. This visualization also helped identify subwatersheds to prioritize for remediation.

Spatial and temporal trends in the SC DHEC data were visualized in several ways:

1. Graphically by plotting geometric means (geomeans) and Est. 90th percentiles for each monitoring site as reported in the SC DHEC shellfish reports.
2. As a color-coded matrix to show water quality criteria contraventions by site and year.
3. Annual box plots for each site with a LOWESS curve fit (locally weighted scatterplot smoothing).
4. GIS mapping of concentrations binned by quartiles for two decades: 2000 to 2009 and 2009 to present. The volunteer monitoring data are included in these maps.

Wet vs Dry Tests for Difference. To test the hypothesis that fecal coliform levels are higher under wet as compared to dry conditions, the Mann-Whitney U test was used to compare wet and dry data. Definitions of wet and dry conditions were optimized from a sensitivity analysis.

This resulted in the following bins: wet data were ones collected within 3 days of a daily rain accumulation of at least 0.5” (12.7 mm), and dry data were ones collected after at least 3 days of 0.0” (0.0 mm) daily rain accumulation. The most recent complete decade was selected for study, i.e., 2000 to 2009, to produce a dataset of large enough size to

3 This is also a standard rain event used in NPDES permits and regulations.
enable detection of significant differences and to best reflect
current conditions. The results were presented visually on
the boxplotted data and rated as either: (1) highly significant
(p<0.05), (2) significant (0.05<p<0.10), or (3) no significant
difference (p>0.10). The latter means that the test failed to
find sufficient evidence of difference.

Several approaches were used to test the hypothesis
that fecal coliform levels have increased over time. This
hypothesis was formulated in recognition of: (1) a historical
increase in the number of TMDL sites from 8 in 2005 to
20 in 2012 and (2) increasing trends in the time trend plots
of the geomeans and Est. 90th percentiles (Figures 2 and 3)
especially at the ends of the record, generally starting with
the 2007 shellfish report. An independent verification that
the trend analyses were done appropriately is provided by
the finding of decreasing trend at sites that are no longer
being sampled, i.e., SC DHEC Sites 04-01a, 04-03, 04-04,
04-05, 04-17, and 04-22, and increasing trend at sites that
had been added to the TMDL due to their non-supporting
status, i.e., Sites 04-04a, 04-17a, 04-28 and 04-31.

Tests for Time Trends. The Mann-Kendall test (Hirsch and
Slack 1984) was used to test for the presence of a monotonic
increasing or decreasing trend over time. At least five years of
data are required. The test is robust against data gaps (Meals
et al. 2011), is non parametric, and has been widely used for
evaluating trends in fecal bacteria data. It is also robust against
changes in units, which accommodates SC DHEC’s upper
reporting limit of 1600 MPN/100 mL for fecal coliforms.

The Mann-Kendall test for trend was run for each site
using all the data available. In some cases, data prior to the
method change in 1990 were available and used. The earliest
data dated from 1967. Site 04-01 is notable for having data
back to 1967 and makes a good test case as it is also the
most contaminated of the sites in Murrells Inlet. Values
for the slope of the linear trend and p value were used to
classify the strength of the relationship between fecal
coliform concentrations and time. Results were considered
highly significant if p<0.05 and significant if 0.05<p<0.10.

Although this test can be used to control for effects of
seasonality, there was no process-based reason to hypothesize
seasonality in Murrells Inlet and data exploration did not reveal
evidence of such trends. The Mann-Kendall trend test also has
an option to control for the effect of rainfall. Since the Mann-
Whitney U test results suggested a significant influence of rain
at most sites, the Mann-Kendall test was also performed to
test the hypothesis when the influence of rain was removed.

The Mann-Kendall test was rerun using only the last five
years of data (2007-2012) to verify the visual observation
of recent increasing trends in the geomeans and Est. 90th
percentile time trend plots (Figures 2 and 3).

For the volunteer monitoring data, less than five years of
data were available, so the presence of a trend was evaluated by
performing a linear regression on the log transformed E. coli data.

Effect of Tidal Stage. To test the hypothesis that fecal
coliorm levels are higher during low tide as compared to
high tide, the Mann-Whitney U test was used to compare
data binned into two categories of tidal stage. Low tide was
defined as stages from ¾ ebb to ½ flood and high tide from ½
flood to ¾ ebb. Differences were considered highly significant
for p<0.05 and somewhat significant for 0.05<p<0.10. This
was tested at each site with SC DHEC fecal coliform data
collected from 1990 to 2011.

Effect of Salinity. To test the hypothesis that fecal coliform
levels are higher when salinity is lower, a linear regression
was used on data from all sites. To ensure sufficient low
salinity data to detect a trend, the regression was performed
on data from all sites combined from 1990 to 2011.

Subwatershed Prioritization. All the statistical test results
were collated by sampling site into a summary matrix
(Table 10) to provide a weight-of-evidence approach to
understanding the causes of overarching spatial and temporal
trends. Sites were grouped by subwatershed. Information on
peak runoff and drainage acreage was included to provide
insight into terrestrial drivers of spatial trends. The matrix
was aligned against a map of sampling sites to help visualize
spatial trends. Color coding was used to highlight large
subwatersheds with high storm flows as inferred from
TR-55 2-year event calculations. A similar color-coding
scale was used to identify statistical results that indicated
persistent, high and increasing levels of fecal bacteria from
the SC DHEC and volunteer monitoring data. This enabled
identification of subwatersheds with persistent and high
levels of fecal bacteria contamination as compared to ones
with low levels and subwatersheds with increasing levels as
compared to others with declining or stable levels.

RESULTS

Graphical visualization. The geomeans and Est. 90th
percentiles published in the SC DHEC shellfish reports for
each site from 1992 to 2011 are plotted as time series graphs
in Figures 2 and 3. Each data point represents three years of
data, using the middle year for the x axis label, so there
is overlap in the data analogous to a moving average. The
water quality criteria are represented by the red line. These
plots are grouped into three tiers based on concentration
range. The highest concentrations were put into Tier 1. The
Est. 90th percentile is a tighter criterion than the geomean
threshold, so the former is more frequently contravened
than the latter. Nonetheless, the tier groupings are consistent
between the two water quality criteria, which provides for an
identification of sites that have been most consistently and
highly contaminated (04-01, 04-16, and 04-8). Sampling at
Site 04-01A ended in 2001.

Tables 3 and 4 are color-coded matrices that show
water quality criteria contraventions by site and year for the
geomeans and Est. 90th percentiles, respectively. The site
results were split into quartiles to identify sites with the highest
frequency of contraventions (Table 5). These rankings were
used to generate the index labeled “Hot” in the summary matrix in Table 10. These sites are identical to the ones identified in Figures 2 and 3 and demonstrate that the sites with the highest levels of fecal coliforms have also been consistently contaminated. The results were also used to identify years with the highest frequency of contraventions (Table 6). The last two shellfish reports had unusually high levels of contraventions suggesting an increase in contamination over this period.

Annual boxplots with LOWESS curve fit (locally weighted scatterplot smoothing) were used to illustrate the results of the trend analyses performed with the Mann Kendall test. An example is provided in Figure 4 for Site 04-01, which had a pronounced trend of increasing fecal coliforms as determined by the Mann-Kendall test and visualized by the LOWESS curve fit. The annual boxplots demonstrate that data variance has been uniform over time, enabling use of the Mann-Kendall test for monotonic trend.

Figure 5 shows GIS mapping of the Est. 90th percentiles binned by quartiles for two decades: 1999 to 2009 and 2009 to present. The volunteer monitoring data are included in these maps as their 90th percentiles. These maps illustrate that the most contaminated sites are located nearest the mainland where urbanization is highest (middle to north end of the Inlet) and that the volunteer monitoring sites with the most persistently high E. coli levels are located upstream of the SC DHEC sites that have the most persistently high fecal coliform levels.

**Wet vs Dry Tests for Difference.** Table 7 lists the results of the Mann-Whitney U tests used to determine significant differences between wet and dry weather data by site. The results include the sensitivity analysis that evaluated the appropriate time window to use for antecedent dry and wet conditions. The 3-day window did best at detecting significant differences, providing some insight into the time to concentration within the subwatersheds. Of the sites monitored during this period, 17 of 28 (61%) had some evidence of higher fecal coliform concentrations under wet as compared to dry conditions. A similar test for difference was performed on the volunteer monitoring E. coli data. Half of the sites had evidence of wet samples having higher fecal bacteria concentrations than during dry weather using the 3-day window.

**Tests for time trends.** Table 8 lists the results from the Mann Kendall and linear regression tests used to test for a significant increase in fecal bacteria concentrations over time. If slopes were 0.00, a trend was not considered to be present even if the p value was highly significant.

Tests performed on the entire dataset are designated as being “with rain” or “wet”. The Mann Kendall results obtained by controlling for the influence of rain (data collected within 3 days of a 0.5” daily rain accumulation) are labelled “dry”; this version of the test looks for trend in the absence of the controlled driver, i.e. rain. For the entire dataset, two sites (04-01 and 04-04b) had evidence for increasing fecal coliform levels under wet and dry conditions. Four sites had evidence for increasing trend under dry conditions only (04-02, 04-06, 04-26 and 04-31). Three sites had evidence of decreasing trend under dry and wet conditions (04-17 and 04-01a (both no longer sampled) and 04-29). Two sites had evidence for increasing trend under dry conditions only (04-23 and 04-24). These tests were rerun for data from 2007-2012 since Figures 2 and 3 suggested a recent increase in fecal coliform levels. Significant increasing trends under wet and dry conditions were detected at six sites (04-01, 04-04a, 04-04b, 04-04c, 04-06, and 04-30) and under wet conditions only at two sites (04-02 and 04-27).

Site 04-01 had the largest slopes by far of all the sites, with the most recent increasing trend being on the order of 10 MPN/100 mL per year. This begged the question as to whether Site 04-01, the most contaminated of the sites, was ever attaining the Class SFH water quality criteria, i.e. could natural sources always have been present and flushing so limited that this site was always “contaminated”? To answer this question, legacy data were analyzed by binning geomeans and Est. 90th percentiles in 3-year running groups similar to Figures 2 and 3. The resulting time trend plots suggest that by the mid 1970’s, the geomean and Est. 90th percentile criteria were no longer being met. But due to missing data, these conclusions are not robust. To address this, the data were binned into larger and non-overlapping time steps to generate sample sets of similar size. Time trends were explored for percent exceedances in two other water quality criteria used in the NSSP, i.e., 88 and 260 MPN/100 mL. These time trends indicate that the fecal coliform levels were contravening the 88 and 260 MPN/100 mL criteria in less than 10% of the samples until the mid-1970’s. This provides additional evidence that water quality was meeting NSSP criteria even at the presently most contaminated site (04-01) prior to the 1980’s.

For the volunteer monitoring data, less than five years of data were available, so the presence of a trend was evaluated by performing a linear regression on log transformed E. coli data. The only trends detected were declining ones at BHR and HS.

**Effect of tidal stage.** Before binning data into high and low tide cohorts, the data were evaluated to verify that sampling had been conducted equally at all 8 stages of the tide distinguished by SC DHEC as part of their shellfish program monitoring protocol. The Mann-Whitney U test was used to detect differences in fecal coliform levels between low and high tide. The results are shown in Table 9. Twenty-six of the 30 sites had evidence for significantly higher fecal coliform on low tide as compared to high tide. Of the other four sites, one is relatively clean (04-04C) having only one sample >50 MPN/100 mL, one is near the SC Department of Natural Resources’ boat ramp where resuspension from heavy boat use is likely occurring at all tidal stages, and the other two sites are located in Oyster Cove (04-29 and 04-30) where flushing is likely highly restricted.

Sites that had much higher frequencies of relatively high fecal coliform concentrations (aka >50 MPN/100 mL) during low tide as compared to high tide are interpreted as having the largest difference in water flows between low and high tide. These are sites 04-01A (no longer sampled), 04-07, 04-08A, 04-18, and 04-28. These sites are likely to benefit most from dredging as a strategy for reducing fecal coliform levels.
Figure 2. Geomean of fecal coliform concentrations (MPN/100 mL) from SC DHEC Shellfish monitoring reports. Water quality criteria is shown by the red line. Stations are grouped into three tiers reflecting high, medium and low levels of contamination. Site locations are shown in Figure 1.
Figure 3. Est. 90th Percentile fecal coliform concentrations (MPN/100 mL) from SC DHEC Shellfish monitoring reports. Water quality criteria is shown by the red line. Stations are grouped into three tiers reflecting high, medium and low levels of contamination. Site locations are shown in Figure 1.
Table 3. Contraventions of geomean water quality criteria as reported in the annual Shellfish Monitoring Reports. U = under the water quality criteria. O = over the water quality criteria.

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Table 4. Contraventions of Est. 90th percentile water quality criteria as reported in the annual Shellfish Monitoring Reports. U = under the water quality criteria. O = over the water quality criteria.

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Figure 4. Annual boxplots of fecal coliform concentrations (MPN/100 mL) at SC DHEC Site 04-01. Blue line is the LOWESS fit.
Figure 5. Decadal trends in 90th percentile of fecal bacteria concentrations for the volunteer monitoring and SC DHEC data. The latter are the estimated 90th percentile as used by the National Shellfish Sanitation Program. The panel on the left (A) shows data from 1999 to 2009 and the panel (B) on the right from 2009 to 2012. Color coding is grouped by quartiles. Fecal coliform concentrations are in MPN/100 mL and E. coli concentrations are in CFU/100 mL.
Table 5. (A) Percent exceedance of geomean and Est. 90th percentile in SC DHEC fecal coliform data from 1992 to 2011. Results are color coded from lowest to highest (green, yellow, organic, red). Sites that are no longer sampled are shaded yellow. Original TMDL sites are designated with an “o”. Sites now within the TMDL are marked with an “x” as per Table 1. Of these, sites 04-03A, 04-03B, 04-04A, 04-04C and 04-17A are located near marinas. Sites in red font are in Tiers 1 and 2 as per Figures 2 and 3. Site 04-32 is a new site so no data were reported through 2011. (B) Quartiles of geomean and Est. 90th percentile in the fecal coliform data from all sites based on the most recent shellfish report (2009-2011). This color coding is used in Table 5A in the site column to identify which sites are currently most contaminated.

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<th>Est. 90th Percentile</th>
</tr>
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<td>100%</td>
</tr>
<tr>
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<td>100%</td>
</tr>
<tr>
<td>04-02</td>
<td>12%</td>
<td>82%</td>
</tr>
<tr>
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<td>0%</td>
</tr>
<tr>
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<td>22%</td>
</tr>
<tr>
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<td>0%</td>
<td>22%</td>
</tr>
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<td>10%</td>
</tr>
<tr>
<td>04-04a</td>
<td>0%</td>
<td>13%</td>
</tr>
<tr>
<td>04-04b</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td>04-04c</td>
<td>0%</td>
<td>0%</td>
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<td>0%</td>
<td>29%</td>
</tr>
<tr>
<td>04-08a</td>
<td>0%</td>
<td>0%</td>
</tr>
<tr>
<td>04-08a</td>
<td>100%</td>
<td>100%</td>
</tr>
<tr>
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<td>63%</td>
</tr>
<tr>
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<td>56%</td>
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<td>82%</td>
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<td>35%</td>
</tr>
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<td>04-29</td>
<td>0%</td>
<td>12%</td>
</tr>
<tr>
<td>04-30</td>
<td>0%</td>
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<td>33%</td>
</tr>
<tr>
<td>04-32</td>
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<td>0%</td>
</tr>
</tbody>
</table>

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<th>Percentile</th>
<th>Geomean</th>
<th>Est. 90th Percentile</th>
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<tr>
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<td>324</td>
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<tr>
<td>75%</td>
<td>14</td>
<td>97</td>
</tr>
<tr>
<td>50%</td>
<td>8</td>
<td>43</td>
</tr>
<tr>
<td>25%</td>
<td>5</td>
<td>23</td>
</tr>
<tr>
<td>10%</td>
<td>4</td>
<td>19</td>
</tr>
</tbody>
</table>

Table 6. Percent of sites exceeding the geomean and Est. 90th percentile water quality criteria for each shellfish report issued between 1992 and 2011. Results are color coded from lowest to highest (green, yellow, organic, red). Shellfish reports with the highest percent exceedances are labeled in red font.
Table 7. *p* values for Mann-Whitney U test for significant difference between dry and wet weather samples. Significant *p* values are in red and less significant ones are in pink. Three windows of antecedent dry and wet weather were evaluated (1, 2 and 3 day). W = wet weather concentration > dry weather concentrations. D = dry weather concentrations > wet weather concentrations. Yellow cells had similar wet and dry concentrations. Black cells indicate no wet and/or dry data met the window selection criteria. See text for details on selection criteria. Site color coding is same as for Table 5.

<table>
<thead>
<tr>
<th>Sites</th>
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<th>2 day</th>
<th>3 day</th>
</tr>
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<tr>
<td></td>
<td><em>p</em></td>
<td>higher</td>
<td><em>p</em></td>
</tr>
<tr>
<td>04-01</td>
<td>&lt;0.02</td>
<td>W</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>04-01a</td>
<td>0.03</td>
<td>W</td>
<td>0.08</td>
</tr>
<tr>
<td>04-02</td>
<td>0.04</td>
<td>W</td>
<td>0.00</td>
</tr>
<tr>
<td>04-03</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-03a</td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-03b</td>
<td>0.09</td>
<td>W</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>04-04</td>
<td>&lt;0.001</td>
<td>W</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>04-04a</td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-04b</td>
<td>0.08</td>
<td></td>
<td>0.03</td>
</tr>
<tr>
<td>04-04c</td>
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<td>0.01</td>
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<td>04-06</td>
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</tr>
<tr>
<td>04-07</td>
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<td>0.01</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
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<td></td>
<td></td>
</tr>
<tr>
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<td>0.01</td>
<td>W</td>
<td>0.00</td>
</tr>
<tr>
<td>04-18</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>04-25</td>
<td>&lt;0.001</td>
<td>W</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>04-26</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-27</td>
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<td></td>
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<tr>
<td>04-28</td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
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<td>x</td>
<td>0.01</td>
<td>W</td>
</tr>
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<td>W</td>
</tr>
<tr>
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<td>x</td>
<td>0.07</td>
<td>W</td>
</tr>
<tr>
<td>04-32</td>
<td></td>
<td></td>
<td></td>
</tr>
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</table>
Table 8. p values from the Mann Kendall test for monotonic trend in SC DHEC fecal coliform data. Results on left are from 1967 to 2011 and on the right from 2007-2012. Highly significant increasing trends are shaded in red and somewhat significant trends are in pink. Highly significant declining trends are in dark green and somewhat significant declining trends are in light green. Site color coding is same as for Table 5. Slopes are given for sites where trends were significant. Overall trends called “total” refer to the results for tests that did not include the rain control (w/ rain). Dry trends are the result from the Mann-Kendall test controlled for rain. See text for details.

<table>
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<tr>
<th>Site</th>
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<th>Overall Trends</th>
<th></th>
<th>Overall Trends</th>
</tr>
</thead>
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<tr>
<td></td>
<td>w/rain</td>
<td>slope</td>
<td>dry</td>
<td>slope</td>
<td>w/rain</td>
</tr>
<tr>
<td>04-01</td>
<td>0.00</td>
<td>0.92</td>
<td>0.00</td>
<td>1.21</td>
<td>Increasing total + dry</td>
</tr>
<tr>
<td>04-01a</td>
<td>0.03</td>
<td>-0.56</td>
<td>0.10</td>
<td>-0.64</td>
<td>Decreasing total + dry</td>
</tr>
<tr>
<td>04-02</td>
<td>0.43</td>
<td>0.11</td>
<td>0.03</td>
<td>Increasing dry</td>
<td>0.04</td>
</tr>
<tr>
<td>04-03</td>
<td>0.26</td>
<td>0.95</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>04-03a</td>
<td>0.79</td>
<td>0.35</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-03b</td>
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<td>0.39</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>04-04</td>
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<td>0.19</td>
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<td></td>
<td></td>
</tr>
<tr>
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<td>0.28</td>
<td>0.28</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>04-04b</td>
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<td>0.32</td>
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<td>Increasing total + dry</td>
<td>0.03</td>
</tr>
<tr>
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<tr>
<td>04-05</td>
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<td>0.03</td>
<td>0.00</td>
<td>No evidence for trend</td>
</tr>
<tr>
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<td>0.04</td>
<td>0.06</td>
<td>Increasing dry</td>
<td>0.04</td>
</tr>
<tr>
<td>04-07</td>
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</tr>
<tr>
<td>04-08a</td>
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<td>0.00</td>
<td>0.19</td>
<td>No evidence for trend</td>
<td>0.01</td>
</tr>
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<td>0.51</td>
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</tr>
<tr>
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<td>0.05</td>
<td>-0.10</td>
<td>Decreasing total + dry</td>
</tr>
<tr>
<td>04-17a</td>
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<td>0.19</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-18</td>
<td>0.03</td>
<td>0.00</td>
<td>0.62</td>
<td>No evidence for trend</td>
<td>0.03</td>
</tr>
<tr>
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<td>0.31</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>04-23</td>
<td>0.00</td>
<td>0.00</td>
<td>0.02</td>
<td>-0.01</td>
<td>Decreasing dry</td>
</tr>
<tr>
<td>04-24</td>
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<td>0.00</td>
<td>0.11</td>
<td>-0.01</td>
<td>Decreasing dry</td>
</tr>
<tr>
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<td>0.14</td>
<td>No evidence for trend</td>
<td>0.10</td>
</tr>
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<td>0.00</td>
<td>0.22</td>
<td>Increasing dry</td>
<td>0.07</td>
</tr>
<tr>
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<td>0.90</td>
<td></td>
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</tr>
<tr>
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<td>0.56</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>04-29</td>
<td>0.00</td>
<td>-0.02</td>
<td>0.00</td>
<td>-0.07</td>
<td>Decreasing total + dry</td>
</tr>
<tr>
<td>04-30</td>
<td>0.06</td>
<td>0.00</td>
<td>0.49</td>
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<td>0.00</td>
</tr>
<tr>
<td>04-31</td>
<td>0.41</td>
<td>0.11</td>
<td>0.11</td>
<td>Increasing dry</td>
<td>0.00</td>
</tr>
</tbody>
</table>
**Effect of salinity.** Figure 6 shows the linear regressions used to test for an inverse correlation of fecal coliform concentrations with salinity in the SC DHEC data from 1990 to 2011. The salinity data were binned into 9 categories as illustrated in the histogram shown in Figure 6A. The fecal coliform data were log transformed. The p value for the linear regression was highly significant (p = 0.0000) with a correlation coefficient of -0.62, which suggests that as hypothesized, fecal coliform concentrations decrease with increasing salinity. This driver accounted for 62% of the variability in the fecal coliform concentrations.

**Subwatershed prioritization.** The results of all the statistical analyses were collated into the matrix presented in Table 10 to provide a weight-of-evidence approach to identification of subwatersheds with highest degrees of contamination or recent increasing trends. Three sets of subwatersheds were identified as problematic, i.e., the ones draining into Sites 04-01, 04-26 and 04-02 on the northernmost end of the Inlet, the ones draining into Sites 04-16, 04-08 and 04-06 on the mainland coastline at the middle of the Inlet, and the ones draining to Site 04-28 on the south end. The latter represents a site of recent shellfish bed closure, suggesting an increasing trend that could be most readily reversed by management intervention.

**DISCUSSION**

The results were collated in a map-based matrix that included subwatershed characteristics such as acreage and TR-55 estimated peak flows. This format was used to facilitate prioritization of subwatersheds for remediation via use of stormwater treatment practices. The spatial analyses illustrated that the sites located near commercial shellfish beds have high water quality, as they infrequently contravened water quality criteria. In contrast, most of the shellfish beds that are closed due to water quality impairments are on state grounds. The general driver behind these spatial trends is proximity to land with most of the approved beds being located in deeper portions of the estuary and the state grounds being located on the water frontage of the mainland.

In general, the highest fecal coliform levels are consistently observed at sites in tidal creeks with frontage on the mainland, suggesting a land-based source of the fecal bacteria. This was supported by statistical tests that found significantly higher fecal coliform levels under wet as compared to dry conditions at many sites and an inverse relationship with salinity. This also suggests that stormwater runoff from the land is an important transport agent. The inverse relationship with salinity likely arises from several related processes: (1) periods of lower salinity are associated with less dilution and flushing by seawater; (2) less die-off occurs in low salinity waters due to less contact with saline seawater; and (3) a greater likelihood of resuspension of sediment to which bacteria are adsorbed, following periods of stormwater runoff.

**CONCLUSIONS**

Much evidence was identified supporting the importance of land-based sources of fecal bacteria to Murrells Inlet, especially during wet weather, although legacy sediment contamination in the tidal creek bottoms could also be a contributor. Evidence for increasing fecal coliform concentrations was found only at a few sites and could be associated with changes in rainfall, land use, and reduced flushing. The latter could be due to infill sedimentation in the tidal creeks. This process has a natural component to it as well as an anthropogenic influence as development mobilizes sediment from such processes as removal of vegetative buffers from stream and creek banks. Many of the near-shore sampling sites that exhibit the highest fecal coliform levels have been consistently exceeding the shellfish water quality criteria since at least the early 1980’s and possibly earlier.
Figure 6. Fecal coliform concentrations (MPN/100 mL) versus salinity (psu). (A) Sample counts in each salinity bin. (B) Linear regression of log-transformed fecal coliform against salinity using binning shown in panel (A). Data are shown as boxplots with 10th and 90th percentiles defining the hinges. Water quality criteria (geomean and Est. 90th percentile) are represented by the lower and upper orange lines, respectively.
Table 9. Influence of tidal stage on SC DHEC fecal coliform concentrations. See text for explanation on how samples were binned into low and high tide. Results of the Mann-Whitney U test for difference between the low and high tide fecal coliform levels are shown as p values. Percent occurrence of high fecal coliform levels (>50 MPN/100 mL) is shown for high and low tide samples. Last column shows the difference in percentage between the two. Large differences are in red font. Sites shaded yellow are no longer sampled.

<table>
<thead>
<tr>
<th>Site</th>
<th>Counts</th>
<th>% of samples binned into High and Low categories</th>
<th>Mann Whitney U test for difference between high and low tide (p value)</th>
<th>Sample counts in High and Low categories for FC &gt; 50 MPN/100 mL</th>
<th>% occurrence of FC &gt; 50 MPN/100 mL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Samples</td>
<td>High Tide</td>
<td>Low Tide</td>
<td>Mann Whitney U test</td>
<td>High Tide</td>
</tr>
<tr>
<td>04-01</td>
<td>261</td>
<td>101</td>
<td>107</td>
<td>80%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-01A</td>
<td>160</td>
<td>45</td>
<td>76</td>
<td>76%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-02</td>
<td>303</td>
<td>114</td>
<td>109</td>
<td>74%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-03</td>
<td>167</td>
<td>57</td>
<td>73</td>
<td>78%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-03A</td>
<td>119</td>
<td>52</td>
<td>34</td>
<td>72%</td>
<td>0.001 **</td>
</tr>
<tr>
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<td>123</td>
<td>54</td>
<td>33</td>
<td>71%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
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<td>204</td>
<td>67</td>
<td>66</td>
<td>75%</td>
<td>0.007 **</td>
</tr>
<tr>
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<td>112</td>
<td>53</td>
<td>36</td>
<td>79%</td>
<td>0.002 *</td>
</tr>
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<td>98</td>
<td>48</td>
<td>29</td>
<td>79%</td>
<td>0.003 **</td>
</tr>
<tr>
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<td>259</td>
<td>95</td>
<td>109</td>
<td>79%</td>
<td>0.006 **</td>
</tr>
<tr>
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<td>308</td>
<td>115</td>
<td>114</td>
<td>74%</td>
<td>0.012 **</td>
</tr>
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<td>280</td>
<td>103</td>
<td>113</td>
<td>77%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-08A</td>
<td>263</td>
<td>97</td>
<td>102</td>
<td>76%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-08B</td>
<td>276</td>
<td>101</td>
<td>112</td>
<td>77%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
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<td>264</td>
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<td>100</td>
<td>75%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
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<td>53</td>
<td>69</td>
<td>78%</td>
<td>0.060 *</td>
</tr>
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<td>53</td>
<td>36</td>
<td>72%</td>
<td>0.462</td>
</tr>
<tr>
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<td>303</td>
<td>111</td>
<td>120</td>
<td>76%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-12</td>
<td>54</td>
<td>9</td>
<td>36</td>
<td>83%</td>
<td>0.012 **</td>
</tr>
<tr>
<td>04-13</td>
<td>279</td>
<td>102</td>
<td>114</td>
<td>77%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-14</td>
<td>272</td>
<td>99</td>
<td>116</td>
<td>79%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-15</td>
<td>304</td>
<td>118</td>
<td>108</td>
<td>74%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-16</td>
<td>267</td>
<td>112</td>
<td>90</td>
<td>79%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-17</td>
<td>253</td>
<td>103</td>
<td>50</td>
<td>74%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-18</td>
<td>274</td>
<td>113</td>
<td>94</td>
<td>76%</td>
<td>&lt;0.001 **</td>
</tr>
<tr>
<td>04-19</td>
<td>279</td>
<td>114</td>
<td>90</td>
<td>76%</td>
<td>0.233</td>
</tr>
<tr>
<td>04-20</td>
<td>278</td>
<td>115</td>
<td>97</td>
<td>76%</td>
<td>0.863</td>
</tr>
<tr>
<td>04-21</td>
<td>175</td>
<td>75</td>
<td>52</td>
<td>73%</td>
<td>0.008 *</td>
</tr>
<tr>
<td>04-22</td>
<td>8</td>
<td>5</td>
<td>3</td>
<td>100%</td>
<td>0.036 **</td>
</tr>
</tbody>
</table>
Table 10. Summary matrix showing results of statistics for each site arranged by subwatersheds (basins). Top set of basins are on the west side of the Inlet listed from northernmost to southernmost coinciding with map shown in left panel. (See Figure 1 for a map with subwatershed names). Bottom set of basins are on the east side of the Inlet. Acreage and peak flow (2-yr storm) are color coded to shown increasing values from green to yellow to orange to red. Volunteer monitoring statistics are color coded as follows: “Persistently elevated” sites are graded in pink (high) and red (highest). Time trends are green for decreasing Escherichia coli. Wet > Dry cases are red. Salinity trends: FW = as salinity decreases, E. coli increases. SW = as salinity increases, E. coli increases. SC DHEC monitoring sites statistics are color coded as follows: SC DHEC site IDs shaded yellow are no longer sampled. Those shaded in sepia are near marinas. TMDL sites are colored red. Those marked with “o” are original sites. “Persistently Elevated” and “Currently Elevated” sites are shaded red. “Long time trends” and “Recent time trends” are shaded red for increasing and green for decreasing fecal coliform concentrations. NE = no evidence for trend. TD = trend with and without rain adjustment. D = trend only with rain adjustment (dry). T = trend without rain adjustment (wet). “Recent Wet > Dry” are shaded red and pink for highly significant and significant results, respectively. Green shading is used for significant results where fecal coliform concentrations were higher during dry as compared to wet conditions.

<table>
<thead>
<tr>
<th>Basin</th>
<th>Acreage (ac)</th>
<th>Peak Flow</th>
<th>E. coli (2-yr)</th>
<th>Volunteer Monitoring</th>
<th>SC DHEC Monitoring</th>
</tr>
</thead>
<tbody>
<tr>
<td>MELODY</td>
<td>59, 03</td>
<td>VDP</td>
<td>SW</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POINT DRIVE</td>
<td>97, 43A</td>
<td>PDC</td>
<td>04-01</td>
<td>o</td>
<td>TD, TD</td>
</tr>
<tr>
<td>BUM GULLY</td>
<td>48, 243</td>
<td>1GC</td>
<td>04-27</td>
<td>o</td>
<td>D, T</td>
</tr>
<tr>
<td>SUNNYSIDE N</td>
<td>72, 213</td>
<td>1MC</td>
<td>04-01</td>
<td>o</td>
<td>D, T</td>
</tr>
<tr>
<td>MARINER</td>
<td>85, 145</td>
<td></td>
<td></td>
<td>04-08</td>
<td>D</td>
</tr>
<tr>
<td>VAUX HALL</td>
<td>86, 172</td>
<td>HS</td>
<td>04-17A</td>
<td>o</td>
<td>D</td>
</tr>
<tr>
<td>NACHTAW</td>
<td>60, 67</td>
<td>BHR</td>
<td>04-10</td>
<td>o</td>
<td>W</td>
</tr>
<tr>
<td>MARINER/ MURPHY</td>
<td>196, 408</td>
<td>1BH</td>
<td>04-08</td>
<td>o</td>
<td>D</td>
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<tr>
<td>HUNT BEACH ST PK N BCH</td>
<td>171</td>
<td></td>
<td>04-06</td>
<td>o</td>
<td>D, TD</td>
</tr>
<tr>
<td>BIKE BR</td>
<td>180, 565</td>
<td>BB</td>
<td>04-24</td>
<td>D</td>
<td></td>
</tr>
<tr>
<td>OYSTERLAND</td>
<td>8, 57</td>
<td>OLB</td>
<td>04-32, 04-33</td>
<td>D, W</td>
<td></td>
</tr>
<tr>
<td>HUNT BCH STRK MN BCH</td>
<td>12, 324</td>
<td></td>
<td>04-18, 04-19</td>
<td>D, TD</td>
<td></td>
</tr>
<tr>
<td>BROOK GREEN N</td>
<td>20</td>
<td></td>
<td>04-25, 04-26</td>
<td>D</td>
<td></td>
</tr>
<tr>
<td>DOGWOOD SOUTH</td>
<td>20</td>
<td></td>
<td>04-26</td>
<td>o</td>
<td>D</td>
</tr>
<tr>
<td>SOUTH WACAMAW</td>
<td>35</td>
<td>04-06, 04-07, 04-13</td>
<td>o, TD</td>
<td></td>
<td></td>
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<tr>
<td>MARINER QUAY</td>
<td>15</td>
<td></td>
<td>04-04, 04-04C, 04-06</td>
<td>o, TD, TD</td>
<td></td>
</tr>
<tr>
<td>INLET PT</td>
<td>78</td>
<td></td>
<td>04-22</td>
<td>D</td>
<td></td>
</tr>
</tbody>
</table>
The most important bacterial sources identified were wildlife and canines. These conclusions were based, in part, on additional data collection conducted concurrently with the US EPA 319 project. These were funded by Georgetown and Horry Counties. They included efforts by the volunteers to track upstream sources (Young et al. 2014) and by CCU’s Environmental Quality Lab to identify host animal sources using genotypic and chemical markers in the northern reaches of Murrells Inlet (Sturgeon et al. 2014). Genotypic source tracking efforts are pending for the middle and southern reaches.

The urbanized wildlife of greatest concern are raccoons and opossum. In urban settings, raccoons are known to reach extraordinarily high population densities due to lack of predators, abundant food supply, and their problem-solving abilities (Prange et al. 2003). The relative contribution of fecal indicator bacteria by urbanized wildlife has likely been enhanced by land-use changes associated with increased development as this leads to increased overland flows, and hence less infiltration of runoff and associated removal of microbes from the waters discharging into Murrells Inlet. This increase in overland flow arises from increased imperviousness and from the associated ditching and piping that have traditionally been used to manage increased stormwater flows and prevent flooding. These wildlife also are likely to frequent wetland areas to avoid human contact and as a water and food resource.

One of the most important outcomes of this collaborative data analysis was a better understanding of the SC DHEC shellfish monitoring data. This provided cautionary insight into how best to perform statistics and track general trends. For example, sample sites have been relocated over time to better define the boundaries of closed shellfish beds and thereby reduce the area subject to closure. In other words, the sampling sites are not representatively spaced through the Inlet. Over time, their locations have become concentrated in contaminated regions. A very large portion are located near land, close to the most likely source(s) of the fecal bacteria. Without recognizing this sampling shifts, a cursory assessment of trend would indicate that water quality conditions have worsened.

Since only one fecal coliform sample is collected at each SC DHEC site per month, the potential exists for bias if rain days are over sampled. Efforts were made to check for this bias by comparing wet weather sampling frequency to rain frequency. No evidence was found to support bias over time scales of decades. The data did not support a higher resolution investigation. It is possible that rain could be causing a bias over short timescales. This could contribute to short timescale variability in fecal coliform levels with the result being short periods of bed closures. For example, beds were reopened in the southern portion of the Inlet in 2013 after first having been closed in 2011. SC DHEC attributed the improvement in water quality to diminished rainfall in 2011-2012. (The 2013 shellfish report was based on data from 2010 to 2012). Thus any climate phenomenon that contributes to rain variability has the potential to influence fecal coliform levels, such as the El Niño–Southern Oscillation that has a periodicity of 2 to 7 years (MacMynowski and Tziperman 2007).

The weight-of-evidence approach used in this study provided sufficient confirmatory evidence, despite the inherent limitations of the NSSP data, to identify priority subwatersheds for remediation. The transport processes elucidated from the fecal indicator data analysis lead to selection of a suite of best management practices (BMPs) that comprise some of the action items in the watershed-based plan (Newquist 2014). These BMPs are directed at intercepting stormwater flows, dredging in-filled creeks, and reducing fecal sources on land, recognizing that the populations of urbanized wildlife are not natural.

Approval of the watershed-based plan by SC DHEC enabled the stormwater managers in Horry and Georgetown County to obtain US EPA 319 funding to implement some of these BMPs. Insights from the collaborative data analysis also lead to formulation of action items that recommend monitoring that is enhanced to better assess progress in remediating fecal coliform impairments in Murrells Inlet.

ACKNOWLEDGEMENTS

Technical support was provided by SC DHEC (M. Murphy, A. Bennett and M. Pearson), US EPA Region IV (K. Synder), Georgetown County Stormwater (T. Jones), Horry County Stormwater (D. Fuss and T. Garigen), and Earthworks, Inc. (S. Williams). Funding was provided by SCDHEC, Horry and Georgetown Counties.

LITERATURE CITED


Abstract. This case study describes the plan development process, implementation strategies and initial and future challenges to implementation for the Murrells Inlet Watershed Plan (WRCOG, 2014). The Plan was crafted by a group of key stakeholders with community support and guidance to address fecal coliform bacteria loading in shellfish harvesting waters in the Murrells Inlet Estuary along the northeastern South Carolina coast. Stakeholders debated the interpretation of the data analysis and ultimately concluded that the primary pollutant sources were non-human, namely wildlife and domestic animals. Stakeholders also concluded that the loads from these sources were being delivered to the estuary via a landscape that includes a network of surface drainage ditches and subsurface pipes so that water retention on the landscape has been largely short-circuited.

Armed with this information, plan participants devised management measures that encompass several strategies, including: (1) utilize an end-of-pipe/ditch solution that addresses pollution nearest the discharge point; (2) generally reduce volume and flow and/or increase retention/detention across the landscape to reduce the pollutant load; and (3) use education and outreach to achieve behavior change.

During both plan development and the implementation of management measures, the plan steering committee faced significant challenges. Initial challenges include: geographic and space limitations that make the use of large retention or detention devices impractical; lack of state or local requirements to use low impact development techniques to increase retention; and mounting outreach campaigns that cannot guarantee significant pollution reductions. Additional complications include mechanisms to sustain community support and involvement. As implementation progresses, the steering committee must track plan implementation and determine creative ways to evaluate the effectiveness of management measures. Local funding allocations must also be sought to leverage against potential grant funds to enable implementation.

INTRODUCTION

Watershed planning has become increasingly emphasized in a variety of disciplines, including stormwater management, resource conservation and stewardship, and water resource management. Granting and resource management agencies have widely adopted the watershed approach and have published guidelines and manuals to assist communities with watershed planning efforts.

For example, South Carolina Department of Health and Environmental Control (SC DHEC) has embraced the watershed planning concept by encouraging stakeholders in watersheds throughout the state to undertake the watershed planning process. This emphasis manifests itself in the publication of the South Carolina Simplified Guide to Developing Watershed-based Plans (SC DHEC, 2014b) and the offering of a designated Request for Proposals for watershed-based plan development within its Section 319 Grant Program. SC DHEC draws its Simplified Guide from the U.S. Environmental Protection Agency’s Handbook for Developing Watershed Plans to Protect and Restore Our Waters (US EPA, 2008).

These helpful documents, which provide needed structure and organization to the watershed planning process (Figure 1), belie the difficulties and challenges of explaining and managing water resources in the face of competing interests within human society. Furthermore, plan development is only part of the process. Implementation of watershed plans poses significant challenges to those tasked with carrying out plan recommendations and management measures. In urbanized areas, that responsibility falls to Small Municipal Separate Storm Sewer Systems (SMS4) such as Georgetown and Horry Counties.

In presenting this case study, we describe the plan development process, implementation strategies and initial and future challenges to implementation for the Murrells Inlet Watershed Plan (WRCOG, 2014). The strategies are intended to achieve reductions in the pollutant load of fecal coliform bacteria to shellfish harvesting waters.
BACKGROUND

The Murrells Inlet Watershed Plan (WRCOG, 2014) was crafted by a group of key stakeholders with community support and guidance. Murrells Inlet is a coastal community that strongly identifies with its salt marsh and its natural resources, particularly its finfish and shellfish fisheries as signified by the community’s nickname of “Seafood Capital of South Carolina.” The Murrells Inlet watershed encompasses 9,313 acres in Georgetown and Horry Counties (Figure 2) along part of South Carolina’s northeastern coast, which is known as the Grand Strand. The South Carolina Department of Health and Environmental Control (SC DHEC) estimates that the watershed contains 3,108 acres of habitat suitable for shellfish production. As of 2012, 2,217 acres (71%) of shellfish habitat was approved for shellfish harvesting based on water quality testing at 25 locations throughout the watershed (SC DHEC, 2014a).

Murrells Inlet is the most significant shellfish harvesting area in northeastern South Carolina, and it boasts a robust commercial fishing industry. In addition, the seafood restaurants that line the Murrells Inlet Marshwalk, spearheaded by the community preservation organization Murrells Inlet 2020) and the many recreational fishermen and nature lovers who use the Marshwalk for access to the marsh serve as symbols that the economic and cultural underpinnings of the community are inextricably linked to the salt marsh and its resources.

Yet, the salt marsh is exposed to fecal coliform bacteria that has resulted in some oyster beds being closed to harvesting for violations of water quality standards that are designed to protect the safe consumption of raw shellfish. Some SC DHEC water quality monitoring stations in Murrells Inlet Estuary were listed on the state’s 303(d) impaired waters list. As a result, SC DHEC issued a Total Maximum Daily Load report (TMDL) in 2005 that included pollutant load reductions allocated to SMS4s within the Murrells Inlet Estuary watershed, namely Georgetown and Horry Counties (SC DHEC, 2005).

The 303(d) list and the TMDL are both elements of the federal Clean Water Act that are designed to protect and restore water bodies with impairments linked to specific pollutants. The TMDL for Murrells Inlet Estuary generally identifies non-point sources as the main contributor of pollutants, but identifies neither specific pollutant sources nor strategies for mitigating pollutant loads. Those tasks are left to the local communities and require considerable effort, expertise, and financial support.

The State of South Carolina National Pollutant Discharge Elimination System General Permit for Stormwater Discharges from Regulated SMS4s that became effective January 1, 2014 now requires SMS4s to implement monitoring and management measures to address impairments for waters with approved TMDL reports and for those listed on the 303(d) impaired waters list. In an effort to address these impairments prior to the issuance of the new SMS4 permit, the Murrells Inlet community engaged in cross-jurisdictional watershed planning in 2012 with grant funding from the SC DHEC 319 Grant Program for Watershed-Based Plan Development.

The stakeholder-based planning process was led by the Waccamaw Regional Council of Governments and Murrells Inlet 2020, a community cultural and environmental preservation group. The project cooperators engaged as many community members as possible in order to gain a thorough understanding of the social, environmental and economic issues and the various perspectives and viewpoints that were represented in the community at large. Stakeholder activities and meetings attracted realtors, business owners, community activists, water quality monitoring volunteers, state park rangers, and seafood industry representatives. The involvement of a wide range of community members yielded valuable information about both the estuary and the community. Meanwhile, this stakeholder engagement also raised community expectations for on-the-ground environmental improvement to flow from the final watershed plan recommendations.

The effort lasted one and a half years and involved considerable debate and data analysis. Initially, an in-person and online mapping effort engaged stakeholders by asking them to identify possible pollutant sources based on their observations and knowledge of the watershed
landscape. While involving stakeholders is time-consuming, it is important to offer opportunities for participation in the process in order to gather expert and local knowledge and achieve an enhanced understanding of the issues and increased likelihood of consensus (Treby and Clark, 2004). These stakeholder observations were paired with detailed expert analysis of decades of water quality data and rainfall information in order to inform interpretations of analysis results and to discern correlations that might invite more thorough field investigation. This step led to additional field reconnaissance, which ultimately helped to inform the prioritization of locations for management measures that address pollutant sources.

While at times the planning process was confusing and contentious, stakeholders energetically debated the interpretation of the data analysis and field observations, and ultimately concluded that the primary pollutant sources were non-human, namely wildlife and domestic animals. Stakeholders also concluded that the loads from these sources were delivered to the estuary via a landscape that is characterized by a dense network of surface drainage ditches and subsurface pipes so that retention of storm runoff on the landscape has been largely short-circuited. Following more comprehensive and detailed investigation, human sources were eliminated as a significant contributor, with the exception of rare accidental discharges.

**PROJECT CHALLENGES**

During both plan development and the implementation of management measures, the plan steering committee faced significant challenges. Starting with the conclusions of the analysis of data and field investigations, project cooperators were confronted with the difficult task of devising Best Management Practices (BMP) that address both the major pollutant sources and the aggressive pollution reduction estimates established in the TMDL. Given that pollution sources in Murrells Inlet are widespread and are primarily delivered to the impaired receiving waters via a highly modified, dense drainage network that accompanies development, increasing retention on the landscape using either conventional BMP such as wet detention ponds or low impact development (LID) devices such as bioretention areas is appropriate. With a fairly high density of land use already on the landscape, however, geographic limitations make the extensive use of such devices impractical. Not enough space is available to accommodate the number of large detention/retention devices needed.

Furthermore, pollutant mitigation devices that target bacteria have been designed for use in specific geographic locations within small drainage areas with relatively low flows, such as catch basins and curb inlets. Extensive treatment with such devices was deemed impractical, expensive and unlikely to target all or even many of the pollutant contributors. Wildlife and domestic pets such as dogs are typically attracted to vegetated drainage ditches along the roadside or between lots. Anecdotal evidence suggests that the higher-flow pathways like large, vegetated outfall ditches actually serve to concentrate wildlife, so an attempt to treat any of these sources by using devices in upstream catch basins would fail to intercept the primary pollutant sources.

Pollution reduction estimates in the TMDL are significant, approximately 80%. With the constraints on the use of large BMP, it became clear that it would be difficult to sufficiently address the pollution reductions in the TMDL. A challenge for project cooperators was to identify innovative, specialized devices manufactured to target bacteria as a pollutant in stormwater runoff (i.e. nonpoint sources) and apply them within the landscape’s limitations, which likely include untested settings for the devices.

Low impact development devices are known to increase retention on the landscape. Collectively across the landscape, the use of small devices such as rain barrels and rain gardens on individual lots can amount to significant reduction in the volume of stormwater runoff. The challenge for implementing LID on a scale large enough to address the TMDL reduction requirements is that there are currently no state or local requirements for the widespread use of LID in new or existing development. While there is a guidance manual for using LID in coastal South Carolina (Ellis et al., 2014), the document has not been adopted by the South Carolina Department of Health and Environmental Control. There are also no specific state requirements for the use of LID. Likewise, at the local level in Horry and Georgetown Counties, instituting requirements to widely use LID in new developments has not been politically feasible.

The use of education and outreach campaigns to spread a watershed plan’s messages and change behaviors to address pollution reduction is typically a part of a watershed plan’s recommendations. By their voluntary nature, however, such campaigns do not guarantee compliance with the management measures and therefore may not translate into significant pollution reductions that meet the TMDL requirements. Designing and supporting such campaigns so that they will be effective will be a challenge. Furthermore, in general, sustaining community involvement and identifying designated funding mechanisms to implement the plan’s recommendations pose long-term challenges for the plan’s steering committee.

**INITIAL STRATEGIES**

In the face of these challenges and the conclusions from the data analysis, project cooperators chose to consider the following strategies: (1) utilize an end-of-pipe/ditch solution that addresses pollution nearest the discharge point to the estuary; (2) generally reduce volume and flow and/or increase retention/detention across the landscape to reduce delivery of the pollutant load; and (3) utilize education, outreach, and incentive programs to achieve behavior change.
Discharge Point Strategies

The first of the project strategies requires either radical modification (e.g. retrofitting) of the existing drainage system or application of manufactured BMP technology in untested, high-flow settings for which the technology was not originally designed. Retrofitting the drainage system is hampered by space limitations around existing buildings and structures, while the feasibility of untested technology across the landscape warrants pilot studies to prove efficacy.

Based on available research of existing bacteria removal methods, the project cooperators determined that the ideal strategy for bacteria removal is to maximize runoff retention time on the landscape by incorporating detention basins, particularly those with vegetation, into the current drainage system. Increasing retention time lengthens exposure to natural causes of bacteria mortality, such as sunlight and predators. The project cooperators chose to adapt the concept of deploying wetland systems for treating pollutants in wastewater (Iasur-Kruh et al, 2010; Karathanasis et al, 2003) to a setting for treating stormwater with shoreline vegetation or floating treatment wetlands in detention basins.

In the Horry County portion of Murrells Inlet, however, the drainage is handled primarily along the roadside ditch network which cannot physically accommodate detention basins due to space constraints and road construction standards. In Georgetown County, the drainage network primarily concentrates higher flows into larger canal-style ditches that pass between lots, many of which are already developed. With these geographic constraints, limited opportunities exist to incorporate detention basins into the landscape. One location in Georgetown County lends itself to the creation of constructed stormwater wetlands to serve as a detention basin. This project was identified as a priority in the watershed plan. Besides this location, however, the current landscape conditions limit retention options. This reality pushed the stakeholders towards the concept of utilizing technology in higher-flow conditions, although the technology was originally designed for and has only been tested in low-flow conditions.

Bacteria media filter socks have been deployed in roadside drainage ditches in Horry County (Figure 3) and are planned for deployment in canal-style drainage ditches between lots in Georgetown County. One challenge in Georgetown County is to acquire the easements needed to deploy the filter socks. Bacteria media filter socks have been shown to be effective in pollutant removal in controlled settings (Faucette et al, 2013), but have not been tested in ditches. As a pilot project, the filter socks are installed as a series of small check dams within the ditches (Figure 4) in an effort to create a series of micro pools to increase retention time and maximize contact with the media.

In addition, floating treatment wetlands and submerged colloidal filters will be installed in several in-line detention ponds (Figure 5) to intercept and reduce the pollutant loads. Much like constructed stormwater or wastewater treatment wetlands, floating treatment wetlands (Figure 6) have been successfully used to sequester pollutants such as nutrients, heavy metals, and suspended solids (Masters, 2012; Tanner and Headley, 2011). They will be used in Murrells Inlet to help capture sediments to which fecal coliform bacteria attach and move with stormwater runoff. Colloidal filters target sediment particles that transport fecal coliform bacteria. Both the floating islands and colloidal filters have a porous, extruded plastic matrix that maximizes the surface area for periphytic biofilm to form. Sediments to which pollutants adhere will tend to attach to the combination of vegetation and biofilm, which is the basis for pollution reduction (Tanner and Headley, 2011). In addition, by reducing turbidity caused by sediments in the water column, sunlight penetration should increase. Fecal coliform bacteria are susceptible to ultraviolet radiation from sunlight, which is a secondary benefit from this strategy.

Strategies to Increase Retention

The second strategy utilizes widespread implementation of Low Impact Development (LID) techniques to increase retention across the landscape. LID is designed to mimic natural hydrology by integrating practices across the landscape that reduce runoff close to its source (Ellis et al, 2014). Pollutant loads are reduced by reducing stormwater runoff volume. This strategy includes the use of devices such as bioretention swales, rain gardens or constructed wetlands, as well as rain barrels or cisterns.

Bioretention swales are linear features that use biological processes to sequester pollutants in storm runoff, preventing them from reaching adjacent waterways. Bioretention swales differ from simple vegetated swales in that the native soils are excavated and replaced with an engineered soil mix that is designed to infiltrate storm runoff and bind pollutants to soil particles (Ellis et al, 2014). Plantings in bioretention swales range from turf grass to shrubs and flowers with mulch. Rain gardens are similar to bioretention swales but are typically more compact in shape and more closely resemble landscaped beds. A bioretention swale is planned along an existing drainage ditch adjacent to a water quality monitoring station on the north end of the Murrells Inlet estuary.

Constructed stormwater wetlands are best management practices that use biogeochemical processes found in wetland systems to process pollutants (Ellis et al, 2014). They also increase retention time on the landscape. These devices typically have a range of habitats including permanent pools and wet meadows that are planted with native wetland species (Figure 7). They may be used as an alternative to wet detention ponds. A constructed stormwater wetland basin is planned for a location in Georgetown County on the south end of the Murrells Inlet estuary that is adjacent to an existing volunteer water quality monitoring location. The basin is being designed by faculty and students at Clemson University. This highly visible location is owned by Murrells Inlet 2020, the community preservation organization, and will take advantage of a high water table.

Collectively, rainwater harvesting using rain barrels (Figure 8) or cisterns at homes and businesses can reduce the volume of storm runoff flowing across the landscape (Ellis et
Increasing retention on the landscape is an effective way to reduce the pollutant load reaching adjacent waterways. Recent efforts to offer rain barrels at reduced rates, along with easy installation instructions, have been hosted by community organizations. This will continue to be a strategy.

The benefits of LID are best realized when these techniques are used throughout the landscape (Ellis et al., 2014). The lack of specific local or state requirements for using LID poses a complication for the widespread use of LID techniques. Faced with voluntary participation, education and incentives will need to be used cooperatively to establish interest and confidence in this approach among homeowners.

**Outreach and Education Campaigns**

The third strategy addresses education, outreach and incentive campaigns to change the behavior of target audiences. One example is a pet waste outreach and cleanup campaign, perhaps in concert with the establishment of pet waste ordinances. Many communities around the country have instituted this approach, including some of those along the Grand Strand. Dog waste has been shown to be a significant contributor in some subwatersheds of Murrells Inlet Estuary (WRCOG, 2014).

The Coastal Waccamaw Stormwater Education Consortium, supported by its member SMS4s, has been developing a pet waste cleanup campaign during the last two years. SMS4s and education partners have installed pet waste cleanup stations in numerous public spaces. This effort has been complemented by the common use of similar pet waste stations by homeowner associations.

While these stations offer the tools for cleanup, they do not guarantee compliance by pet owners. The intention is to continue to educate the public so that pet waste cleanup becomes the norm, rather than an uncommon occurrence. In addition to the inherent challenge of establishing a new norm, the fact that pet waste is only a partial contributor to the water quality problem means that such a campaign may be difficult to link directly with significant water quality improvements. Even if pet waste is completely eliminated from storm runoff, there remain other sources of fecal coliform bacteria that may cause water quality monitoring stations to remain in violation of water quality standards.
An additional example is the Inlet-friendly Business Program spearheaded by Murrells Inlet 2020. The program targets restaurants and other businesses that operate in Murrells Inlet, particularly those near the Marshwalk, which is a major draw for residents and tourists alike. Started less than a year ago, the program aims to encourage environmentally-friendly practices by recognizing program participants with window plaques and website acknowledgements. The businesses can use the recognition as a marketing tool to the discerning public that is increasingly seeking to patronize “green” businesses. Activities include committing to manage and maintain dumpster areas; safely disposing of grease and wash water; and avoiding “water brooming” of parking lots and storage or work areas, which are known as a non-point source for fecal coliform bacteria. The challenge for this program is to provide enough incentives to attract businesses’ interest in participation. Another challenge will be adapting this program to the residential sector to achieve pollution reductions.

FUTURE CHALLENGES

As watershed plan implementation moves forward, SMS4s will have to use a strategic approach to determine effectiveness of BMPs in addressing the water quality impairments. Continued financial support and expansion of existing monitoring programs conducted and overseen by Coastal Carolina University’s Environmental Quality Laboratory, including volunteer monitoring, will be needed to evaluate the impacts of BMPs. Generally, BMPs are targeted in areas where long-term monitoring data exists to be able to track trends. In addition to evaluating BMP effectiveness, the local governments operate under state permit requirements that call for monitoring to meet provisions in TMDL reports for impaired waterways. The TMDL for Murrells Inlet Estuary contains such provisions, which call for specific data collection methods that cannot be achieved with volunteer monitoring. An approach must be devised to use resources efficiently to best meet various monitoring needs, which may include additional volunteer and/or technical staff effort. To accomplish such an approach will require the commitment of significantly more resources than are currently allocated.

Sustaining community involvement over a ten to twenty year time period can be daunting. Early energy and enthusiasm tends to wane and implementation may fall to a few key individuals. A watershed plan implementation steering committee, composed of key stakeholders, will oversee and track plan implementation. The steering committee is intended as a vehicle for long-term community engagement. Periodic steering committee meetings are designed to keep stakeholders connected and interested. These meetings will also serve as a forum for determining project priorities as time passes. Support for the steering committee will need to come from key organizational partners, such as the

Figure 6. Schematic of floating treatment wetlands, showing floating matrix with biofilm for treatment (Credit: Midwest Floating Island, LLC).

Figure 7. Constructed wetlands at recreation center in Horry County.

Figure 8. Rain barrel capturing roof runoff at a home in Horry County.

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local and regional governments and Murrells Inlet 2020. Sustaining this interest and energy will require ongoing coordination and communication among the partners. To meet stakeholders’ expectations, water quality improvement projects will need to be consistently undertaken and water quality improvements will need to be demonstrated.

The challenge of funding plan implementation over decades is also a concern. Watershed plan implementation is a long-term endeavor that will require considerable financial and personnel commitments by SMS4s. Since there is no dedicated funding source for implementing these measures, annual budget allocations will be needed. Local funding sources may be leveraged against grant funds to boost implementation by evaluating pilot studies for BMPs or strategies that have not been tested widely, but grant funding remains scarce. This strategy is currently being used to administer a SC DHEC Section 319 implementation grant, which will allow the watershed plan steering committee to evaluate the effectiveness of BMPs before prematurely expending resources. Administering grants requires time, energy and expertise from organizational partners, so long-term commitments must be honored to achieve success.

CONCLUSIONS

Watershed planning is now widely accepted and encouraged. This case study described both the Murrells Inlet Watershed Plan development process that deeply involved the community and the implementation of its recommendations. The sometimes difficult process yielded a plan with community support that recommends measures to address the fecal coliform impairments in the estuary.

Due to the considerable investment of time and energy in the planning process, community stakeholders became committed to the plan. As a result, stakeholders also developed expectations that plan recommendations would be implemented following plan approval. Community support for the plan and its implementation is a critical part of the planning process and is largely generated through involvement in the process. It is important to recognize that it is not the outcome, but the process of active participation that engages stakeholders and engenders support for the elements of the watershed plan.

There are numerous challenges facing the stakeholders during the implementation phase. These range from geographic and space limitations to achieving voluntary adoption of low impact development techniques to mounting campaigns that aim to change behaviors. Additional complications include assessment of effectiveness of management measures, sustained community support and involvement, and dedicated funding for implementation measures. SMS4s must play a lead role in overcoming these challenges to achieve success in meeting pollution reductions in Murrells Inlet Estuary.

ACKNOWLEDGEMENTS

The Murrells Inlet Watershed Plan Steering Committee spent many hours planning, gathering information, evaluating data, and preparing and editing the document. Implementation would not be possible without their work. Of particular importance to completing the project were the co-leads for the watershed plan, Sue Sledz formerly of Murrells Inlet 2020 and Daniel Newquist of the Waccamaw Regional Council of Governments.

Technical assistance was provided by: Ray Davis and Lewis Roach of Floating Island Solutions; Rod Tyler, Britt Faucette, Jon Stewart and J.R. Stewart of Filtrexx International; Russ Britton and Steve Rodgers of EcoExpress; and Stephen Williams of The Earthworks Group.

LITERATURE CITED


Hydraulic Geometry Curves in the Pee Dee Watershed

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Abstract. Hydraulic bankfull geometry or regional curves are a useful metric for evaluating stream stability and planning stream restoration projects. Streams and tributaries within the Middle Pee Dee River Basin (MPDRB) in South Carolina drain an agrarian and forested landscape characterized by water conveyance structures, such as active and historical ditches which support forestry and agriculture. While streams in the region are generally stable, pockets of this landscape are beginning to face increasing pressure from development with signs of stream instability apparent in several locations as evidenced by streams in and around the urbanizing areas around Darlington and Florence, SC. In order to provide a foundation for potential stream restoration projects in the area, 15 sites in the MPDRB were selected on the basis of catchment area, in categories of small (<50 km²), small-medium (50-500 km²), medium (500-1000 km²), and large (>1000 km²). Bankfull geometries, channel substrate, flow and water temperature were measured at all the sites and a set of regional hydraulic geometry curves developed. The frequency of bankfull flows that occurred over the period of sampling were also estimated to document floodplain connectivity. Results suggest that bankfull dimensions in the MPDRB were well correlated with bankfull discharge and drainage area. The results showed that hydraulic geometry in the region were similar to those measured in a similar physiographic region in North Carolina. The study also shows that streams in the MPDRB experience bankfull exceeding flows much more frequently than streams in other parts of the country, but at a frequency that is comparable to streams in the coastal plains of North Carolina.

INTRODUCTION

Hydraulic bankfull geometry relationships are essential to the geomorphological characterization of stable streams that might potentially be subject to perturbations of flow and sediment regime. These perturbations could arise as a function of land use change (short term) or climate change (long term) and can significantly alter the fluvial form and function of stream channels. By establishing a reference condition for channel form and function based upon hydraulic geometry, one might potentially quantify the extent of departure from that stable state and possibly provide a basis for future restoration efforts (Sweet and Geratz, 2003).

The existence of hydraulic geometry in streams with topographically similar watersheds has been well documented and the relationship is referred to as regional curves or hydraulic geometry curves (Metcalf et al., 2009; Sweet and Geratz, 2003; Leopold, 1994; Dunne and Leopold, 1978). Hydraulic geometry curves have been developed for various regions across the United States and are generally represented in the form of a power equation (e.g. Dunne and Leopold, 1978). While Dunne and Leopold’s (1978) hydraulic geometry curves relied on a bankfull flow rate (Qbkf) as the independent term in a power relationship of the form \( W_{bkf} = a Q_{bkf}^b \), recent studies (Metcalf et al., 2009; Cinotto, 2003; Sweet and Geratz, 2003; Doll et al., 2002; Castro and Jackson, 2001) employ drainage area (Ac) as a predictor of hydraulic geometry \( W_{bkf} = a A_c^b \) as a consequence of the close correlation between drainage area and bankfull flow (Doll et al., 2002; Castro and Jackson, 2001).

The development of hydraulic geometry curves have been carried out within specific geographical boundaries, boundaries defined by ecoregion (Sweet and Geratz, 2003), physiographic province (Cinotto, 2003), and the regions with similar average yearly rainfall and runoff patterns (Metcalf et al., 2009). Initially reported by Dunne and Leopold (1978) and later modified by Leopold (1994), hydraulic geometry curves have since been developed across the country for various topographic regions. These include studies in the Pacific NW (Castro and Jackson, 2001), Pennsylvania and Maryland (Cinotto, 2003), northern Florida (Metcalf et al., 2009), Midwestern agricultural streams (Jayakaran et al., 2005) and the piedmont (Doll et al., 2002) and coastal plains (Sweet and Geratz, 2003) regions of North Carolina.

There has also been considerable interest in relating bankfull flow to a recurrence interval. Sweet and Geratz (2003) summarized several published studies (e.g. Castro and Jackson, 2001; Harman et al., 2000, 1999; Rosgen, 1996; Leopold, 1994; Dunne and Leopold, 1978) on streams across the continental U.S., the piedmont, and mountain
regions of North Carolina, reporting bankfull flows associated with a recurrence intervals ranging between 1.4 and 1.6 years. All those studies employed annual duration series (USGS, 1982) with several decades of flow record per study. However, in a study on streams in the coastal plains of North Carolina, Sweet and Geratz (2003) reported a bankfull flow recurrence interval of less than a year based on an annual duration series. However, recurrence intervals based on a partial duration series averaged only 0.19 years for those same streams. In other words, streams in the North Carolina coastal plain tended to overtop their bankfull elevation several times a year.

With the increase in stream restoration projects in neighboring states (Sweet and Geratz, 2003: North Carolina), it is likely that stream restoration projects in South Carolina will soon follow suit. However, to date no regional hydraulic geometry curves have been derived for streams in the MPDRB. As landscape and climate changes impact the streams that drain these watersheds and the need to restore potentially degraded reaches increase, the defining of hydraulic geometries that characterize stable streams in the region become critical. The objectives of this study were to derive bankfull curves for a coastal plain watershed using 15 sites in the MPDRB, as well as to quantify the annual average number of times bankfull exceeding events that took place over the period of available data.

PROJECT DESCRIPTION

Streams in the MPDRB are non-tidal low gradient coastal plain streams with bed substrates comprising a sand or sand-gravel mix. Study sites were selected to represent a wide range of watershed drainage areas, ranging from 17 to 1,718 km². Sixteen sites were selected on the basis of catchment area, in categories of small (<50 km²), medium-small (50-500 km²), medium (500-1000 km²), and large (>1000 km²). Only sites deemed geomorphologically stable based on visual surveys of channel bed, banks, and vegetation were chosen (e.g. Sweet and Geratz, 2003). The selection process also evaluated each possible site on the basis of land use within the watershed, ease of access and security of instrumentation. Study sites were all located within the Southeastern Plains EPA Level III ecoregion (Griffith et al., 2002) though some watersheds had upper sections of their catchment in the Piedmont Level IV ecoregion. At the Level IV scale, watersheds spanned six ecoregions: Atlantic Southern Loam Plains, Southeastern Floodplains and Low Terraces Sand Hills, Southern Outer Piedmont, Carolina Slate Belt, and Triassic Basins (Figure 1). Stream densities in all the study watersheds averaged 0.22 km of stream per square kilometer and varied between 0.13 and 0.37 km of stream length per square kilometer of catchment area (A). Six of the chosen sites utilized United States Geological Survey (USGS) flow monitoring gauges. These sites included Big Black Creek below Chesterfield (02130840), Black Creek near McBee (02130900), Black Creek near McBee (02130900), Black Creek near McBee (02130980), Jeffries Creek (02131110), Lynches River near Bishopville (02131500), and Little Fork Creek at Jefferson (02131320). Four sites were chosen in conjunction with the SCDNR’s fish monitoring program (Figure 1). One of the sixteen sites was subsequently abandoned as a colony of beavers built a dam just downstream of the site, impacting our ability to reasonably quantify flow rates. Ultimately, 15 sites were used to develop hydraulic geometry for the MPDRB.

METHODS

Stream Morphology

For the wadeable stream sites, a total station was used to measure channel pattern, profile, and dimension per Harrelson et al. (1994). Stream surveys ranged from 100 to 300 m along the stream profile depending upon the size of the stream including at least three representative cross sections. Cross sections were chosen based on the presence of a stable riffle with well-defined bankfull features. Depending on the size of the stream, cross sections ranged from 30 to a 120 m apart. Elevations for channel thalweg, water surface and bankfull features were also recorded. Bankfull features were identified, taking careful note of indicators of bankfull level, grade changes, changes in vegetation, significant changes in particle size, level of organic debris, and scour lines (Dunne and Leopold, 1978). Specifically, evidence of bankfull elevation included a significant change in grade (i.e. steep slope to mild slope), change in vegetation (bare soil to grasses, grasses to moss, or the line where woody vegetation begins), significant changes in particle size (gravel to sand, sand to silt, etc.), level of organic debris (i.e. leaf litter), and scour lines (Dunne and Leopold, 1978). Panoramic photos taken at each site helped to corroborate selection of bankfull stage and provided photographic documentation of each site. A weight of evidence approach was used based on the above parameters, and an estimate of bankfull elevation that satisfied as many indicators as possible was made.

For non-wadeable streams, stream pattern, profile, dimension and velocities were measured with a floating acoustic doppler current profiler (ADCP). To measure stream profile and pattern, the ADCP unit (River Surveyor M9 Sontek-YSI) with Real Time Kinematic positioning (RTK-GPS) was towed behind a slow moving boat several times along the stream centerline in both upstream and downstream directions. The RTK-GPS capability allowed for tracking ADCP position in three-dimensional space providing stream sinuosity, and water surface elevations. The profiling capability of the unit provided the elevations of channel bottom along the path of travel. To measure stream dimension and average stream velocity, the ADCP unit was slowly pulled several times from bank to bank across the stream cross section being measured while ensuring that the ADCP’s rate of travel never exceeded 10% of stream velocity. To ensure a complete characterization of stream morphology, total station topographic surveys were carried out to complete the above-water portions of the stream cross sections that were profiled with the ADCP.
Hydraulic Geometry Curves in the Pee Dee Watershed

Flow Monitoring

Streamflow data for the six USGS sites were obtained from the USGS real time water website (http://waterdata.usgs.gov/sc/nwis/rt); data availability ranged from 3 to 52 years. For the 9 remaining sites, flow was estimated from river stage data measured with logging pressure transducers (Solinst® Leveloggers) in conjunction with stage-flow rating curves developed for each site. Site specific stage-flow rating curves were based on estimated roughness coefficients developed using measured velocity readings at various flow depths, and estimating flow using the continuity equation $Q = A \times V$; where $Q$ = estimated flow, $A$ = wetted area, $V$= measured stream velocity. For non-wadeable streams, velocities were estimated using a floating ADCP unit per Mueller and Wagner (2009), while in wadeable streams, a two-dimensional flow velocity meter (YSI-Sontek Flow Tracker®) was used per John (2001). For above bankfull flow stages, a floodplain roughness coefficient was estimated using Chow (1959). Flow values were estimated for every stage sensor value on a 10-minute basis from July of 2009 through June of 2012.

Occurrence of Bankfull Flows

Bankfull discharges were calculated by estimating the amount of flow needed to fill the bankfull channel, based upon the slope and calculated roughness coefficient for each site. We also recorded the number of times flow in the stream exceeded calculated bankfull flow over the period of record. Frequency of bankfull flow exceedance enabled the calculation of an annual average bankfull occurrence rate, or simply, the average number of times in a year that flow in a stream exceeded bankfull flow. Two successive bankfull exceeding events occurred only if the stream level dropped below the bankfull elevation between the two events. Therefore multiple peaks that did not drop below the bankfull stage counted as a single bankfull exceeding event. Given that we only had 2.9 years of flow record at 9 sites (except USGS sites), we calculated bankfull occurrences per...
year and not a traditional recurrence interval as calculated by Sweet and Geratz, (2003) and others. The bankfull occurrence per year metric was simply a means to relate our temporally limited dataset with other published studies.

RESULTS

Most streams in the MPDRB were swampy, sluggish, and impeded by large woody material. Stream slopes ranged from 0.023% to 0.42% and calculated Manning’s roughness values ranged from 0.038 to 0.107. Hydraulic geometry for the MPDRB was based upon bankfull dimensions, in turn derived from measured cross-sections at 15 study sites with drainage areas that spanned three orders of magnitude. Given the broad range of watershed drainage areas, the four bankfull dimensions \( W_{bkf}, D_{bkf}, A_{bkf}, \) and \( Q_{bkf} \), also showed a broad range of values. Bankfull width ranged from 3.4 to 46.7 m, average bankfull depth ranged from 0.5 to 3.2 m, bankfull cross sectional area ranged between 1.5 and 148.0 \( m^2 \), and bankfull flow rate ranged between 0.5 and 68.1 \( m^3/s \). Bankfull dimensions and site parameters are summarized in Table 1.

Bankfull Occurrence

Bankfull occurrence ranged from 0.3 to almost 6.2 times per year with an average of 2.5 occurrences per year across all sites. In other words, flow rates on average met or exceeded bankfull discharge more than 2 times per year in the MPDRB.

Hydraulic Geometry

Bankfull related measurements such as bankfull width, average bankfull depth, bankfull cross sectional area and bankfull flow rate were closely correlated to the size of the contributing watershed (drainage area). Regression analyses yielded highly statistically significant relationships between all log transformed bankfull measurements and watershed drainage area values (predicted \( r^2 \) ranging from 0.85 to 0.95, \( p < 0.001 \)) with drainage area predicting bank flow the best, and bankfull depth the worst. The resulting regional curves, in the form of the modified power functions prescribed by Dunne and Leopold (1978), are presented in Figure 2.

DISCUSSION

Bankfull occurrences per year for the MPDRB tended to be much higher than documented occurrences in other studies (e.g. Metcalf et al., 2009; Wilkerson et al., 2008; Castro and Jackson, 2001; Wolman and Miller, 1960). Annual average bankfull occurrences reported here were more similar to values reported by Jayakaran and Ward (2007) and Sweet and Geratz (2003). In fact, the Sweet and Geratz (2003) study was based on Coastal Plain stream sites in North Carolina that were physiographically most similar to those studied in this project. Sweet and Geratz (2003) report an average of 5 bankfull exceeding flow events annually in the North Carolina (NC) coastal plain, a frequency much greater than the typical 1.5 year recurrence

<table>
<thead>
<tr>
<th>Site</th>
<th>Drainage Area (km²)</th>
<th>Bankfull Area (m²)</th>
<th>Bankfull Width (m)</th>
<th>Bankfull Depth (m)</th>
<th>Bankfull Flow (m³/s)</th>
<th>Manning's N</th>
<th>Slope (%)</th>
<th>Bankfull occurrences per year</th>
</tr>
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<tbody>
<tr>
<td>1</td>
<td>17.3</td>
<td>1.5</td>
<td>3.4</td>
<td>0.4</td>
<td>0.5</td>
<td>0.106</td>
<td>0.420</td>
<td>3.4</td>
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<td>2</td>
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<td>4.8</td>
<td>8.1</td>
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<td>2.1</td>
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<td>4.6</td>
<td>6.9</td>
<td>0.7</td>
<td>1.9</td>
<td>0.076</td>
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<td>0.029</td>
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<td>16.4</td>
<td>1.2</td>
<td>13.9</td>
<td>0.069</td>
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<tr>
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<td>11.2</td>
<td>0.7</td>
<td>19.2</td>
<td>0.022</td>
<td>0.113</td>
<td>1.5</td>
</tr>
</tbody>
</table>

Table 1. Bankfull dimensions and site characteristics for 15 sites used to develop hydraulic geometry curves in the MPDRB.
interval that reported by studies in other part of the United States. They hypothesized this high frequency of bankfull flow events to several characteristics that typify coastal plain watersheds in Southeastern United States. There are: high precipitation, low landscape gradient, large surface storage, high water table conditions, and low flushing rates. Given the similarities in hydrologic and physiographic conditions between MPDRB and the NC coastal plain, the similarity in bankfull flow frequency in this study to the Sweet and Geratz (2003) study, is to be expected.

The investigation of hydraulic geometry relationships in the MPDRB region showed that catchment area and bankfull dimensions were significantly related. The relationships that described hydraulic geometry had coefficients of determination (see Figure 2) that fell within the range reported in the literature. Previously published curves had coefficients of determination as low as 0.54 (Castro and Jackson, 2001) to as high as 0.99 (Metcalf et al., 2009). The highest coefficients of determination typically related bankfull area and flow rate to watershed area (Sweet and Geratz, 2003; Doll et al., 2002).

Figure 2. Hydraulic geometry relationships relating (a) bankfull discharge (b) bankfull area, (c) bankfull width, and (d) bankfull depth, to drainage area. Black and blue lines represent lines of best fit and the 90% prediction intervals, respectively. Light gray lines depict a hydraulic geometry relationships derived by Sweet and Geratz (2003) for coastal plain watersheds in North Carolina.
and the lowest coefficients of determination consistently related average depth to watershed area (e.g. Metcalf et al., 2009; Cinotto, 2003; Sweet and Geratz, 2003; Doll et al., 2002; Castro and Jackson, 2001). The hydraulic geometry curves derived by Sweet and Geratz (2003) reproduced here as gray continuous lines in Figure 2, lie within the confidence limits of the regression lines generated by this study. The slopes of the log transformed regression lines in this study were slightly greater than those derived by Sweet and Geratz (2003) for NC coastal plain streams, but those differences in slope were statistically insignificant.

The hydraulic geometry curves derived in this study provide critical insight into stream function, providing a model that scientists and engineers can use in the classification and restoration of streams in the Middle Pee Dee region. With increasing agricultural and commercial development in the region, stream systems subject to development typically undergo changes in stream morphology driven by changing flow and sediment regimes. These morphological changes are often expressed by stream bank erosion and increased sediment export to downstream receiving waters. Stream bank erosion can cause channel incision and widening that will result in a stream losing equilibrium and deviating from its stable channel geometry. This in turn could lead to flow confinement and a loss of floodplain connectivity resulting in infrequent bank overtopping flows. The negative impacts of development upon riparian functioning have been widely documented in various geographic settings and at multiple spatial scales. (e.g. White and Greer, 2006; Booth and Jackson, 1997; Schueler, 1994; Booth, 1990; Krug and Goddard, 1986; Martens, 1968) These hydraulic relationships provide a basis for stream restoration in the region, and add to an existing framework of hydraulic geometry relationships (Metcalf et al., 2009; Jayakaran et al., 2005; Cinotto, 2003; Sweet and Geratz, 2003; and Doll et al., 2002; Castro and Jackson, 2001; Leopold, 1994) that will likely continue to expand into many other regions. An expansion of this study into the lower and upper portions of the Pee Dee River watershed, as well as an investigation of neighboring ecoregions may illuminate the optimal regional boundaries for application of these hydraulic geometry curves.

ACKNOWLEDGMENTS

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LITERATURE CITED

Hydraulic Geometry Curves in the Pee Dee Watershed


Abstract. A warming climate leads to a moister atmosphere and more rapid hydrologic cycle. As such, many parts of the country are predicted to experience more total rainfall per year and more frequent extreme rainfall events. Most regions of the country have stormwater systems designed to a standard that matches outflow rates to pre-development values for specified return period storms. Increases in these return period storm depths, as predicted by many global climate models, will stress existing stormwater infrastructure. This paper examines how rainfall patterns will change over the remainder of the century across the state of South Carolina.

Rainfall simulations from 134 realizations of 21 global climate models were analyzed across the state of South Carolina through 2099. Results show that there will be increases in both annual total rainfall (ATR) and 24-hour design storm depth for a range of return period storms. Across South Carolina, ATR is predicted to increase by approximately 2.3-4.0 inches over the forecast period while the 100 year design storm depth is predicted to increase by 0.5-1.2 inches depending on location. However there are significant regional variations with the Savannah River Basin experiencing smaller increases in ATR compared to the rest of the state.

INTRODUCTION

Over the last century the average global temperature has risen 0.85 degrees Celsius (IPCC, 2014). Forecasting climate changes is important for preparing societies for possible impacts to food supply, water resources, infrastructure, ecosystems, and even human health. Temperature changes are only one aspect of the predicted changes the Earth will experience. Other changes include precipitation patterns and intensities, ice and snow cover, sea level, and ocean acidity. In 2001, the Intergovernmental Panel on Climate Change (IPCC) published strong conclusions in response to evidence of global climate change (IPCC, 2001). The 1990’s were reported to be the warmest decade, for the northern hemisphere, since adequate record keeping (IPCC, 2001). Trends in precipitation are increasing slightly, about 1% per ten years, and the number of severe precipitation events is also increasing (IPCC, 2001). The IPCC concluded that the warming that is being observed in the last century is not natural. Models that attempt to predict historical trends based on natural radiation perform less well compared to models that include increases in atmospheric greenhouse gas concentrations (IPCC, 2001).

The IPCC made its conclusions based upon a large variety of research and data. Specific to the United States, there has been trend analysis done for precipitation and temperature for major urban areas. Mishra and Lettenmaier (2011) found that there were significant increases in extreme precipitation events in 30% of urban areas from 1950-2009. Martinez et al. (2012) found increasing trends in temperature and decreasing trends for precipitation for the state of Florida for a similar time period.

In general, climate change models predict a warmer and moister atmosphere resulting in a more rapid hydrologic cycle and more extreme rainfall events. Stormwater systems, some of which are already overloaded, will be stressed even further with increased runoff. As a result water quality will decrease as sediment runoff and flooding will increase.

Current South Carolina stormwater regulations (DHEC, 2002), only regulate peak flows and not total runoff. As such, traditional stormwater designs have reduced infiltration and increased total runoff when compared to original site hydrology. Developing sites often requires significant downstream storm sewer infrastructure. With increased rainfall due to climate change, these design weaknesses will cause a disproportionate amount of the additional rainfall to directly become runoff. Responsible stormwater management is required to maintain the quality of surface water in a climate that will exhibit increased frequency and intensity of rainfall over time.

This paper presents the results of a detailed analysis of rainfall forecasts based on Global Climate Model (GCM) data.
archived through the Climate Model Inter-comparison Project - 5 (CMIP5). The data is analyzed to examine the change in annual total rainfall (ATR) and 2, 5, 10, 25, 50 and 100 year 24 hour storm depths between now and the end of the century (the storm depths selected are those used by various municipal and state agencies in their stormwater regulations).

Engineers and regulators will better understand the risk a changing climate will present to stormwater infrastructure as a result of this analysis. That is particularly true for state agencies with regulatory responsibilities for defining stormwater design events such as SC-DHEC and SC-DOT.

The remainder of the paper is structured as follows. The project description summarizes the main goals of the project and pertinent literature. The sources of data used and the analysis techniques are described in the methods section. The results section presents forecasts for the ATR and 2, 5, 10, 25, 50, and 100 year 24 hour storm depths for the entire state of South Carolina. Conclusions and suggestions for future work are presented in the discussion section.

PROJECT DESCRIPTION

As an increase of rainfall intensity and frequency is expected, the responsibility of designing stormwater systems to be effective for their entire design life lies with the designing engineer. However, in order to effectively plan for future rainfall patterns, data on expected changes is required. GCM’s typically produce low spatial resolution data that must be statistically downscaled for the purposes of local hydrologic trend analysis. There are a number of approaches to downscaling including Bias Corrected Constructed Analogs (BCCA) and Bias Correction and Spatial Disaggregation (BCSD) (Ahmed et al. 2013). The choice of downscaling technique depends on the application. Downscaling GCMs using Bias Corrected Constructed Analogs (BCCA) provides a higher temporal and spatial resolution (Barsugli, et al, 2009, Maurer & Hidalgo, 2008) and improved estimates of precipitation compared to other downscaling methods (Brown & Wilby, 2012). Using multiple GCMs removes the bias that a certain model may have and improves the estimation of variability that is typically under estimated by using a single downscaled data set (Brekke, et al., 2008). This study uses projected rainfall data from 134 realizations of GCMs with daily temporal resolution and 1/8o degree spatial resolution to explore long term trends in rainfall in South Carolina. These data sets include GCM model runs for all four Representative Concentration Pathways (RCPs). That is, they include model runs for a range of different long term atmospheric CO2 concentration levels. The choice of appropriate RCP would require a prediction of future public policy which is beyond the scope of this paper. As such, all four data sets were lumped together. The results, therefore, represent an average set of predictions of future rainfall patterns. This approach may underestimate the potential changes in rainfall patterns if global CO2 emissions are not curbed.

METHODS

Downscaled GCM data was analyzed for each of the locations of NOAA precipitation measuring stations, Figure 1, so that the projected rainfall data could be directly compared to historical data and posted 24 hour storm depths. Historical rainfall data is available for all of the stations through the National Climatic Data Center (NCDC) run by NOAA. While breaks in the data (no data recorded) exist in the data sets, they only exist for relatively short periods and are not accounted for in the analysis. The average data set for the historical data from 1950-1999 contained 41.6 years of data. The list of stations was edited to remove duplicate stations (occurring for stations that measured both hourly and daily values), stations located outside the projection grid (occurring for some coastal stations), or stations with region information not specified by NOAA (Bonnin, et al., 2006). BCCA downscaled CMIP5 daily hydrologic projections were downloaded for each station from an online archive (U.S. Department of Interior, 2014). The projections used 21 climate models with various combinations of four RCPs and different realizations creating a total of 134 different daily rainfall projections for a period of record (POR) from 2015-2099.

A precipitation frequency analysis had already been performed on the historical data by NOAA and was the computational method behind the Precipitation Frequency Data Server (PFDS), which gives the storm depths for different return periods and durations. The NOAA Atlas 14, Volume 2 is based on data from 13 states and covers precipitation frequency estimates for event durations of 5 minutes through 60 days at recurrence intervals of 1-year through 1,000 years. The method is based on converting annual maximum data to partial duration data series and then further “personalizing” by location through regionalization. The analysis herein focused on 24 hour storm depths due to their role in stormwater design regulations.

Figure 1. NOAA weather station locations in South Carolina for which observed data was collected and downscaled GCM data was analyzed.
After importing the data for each station, the maximum daily values were converted to 24-hour maximum values using

\[ P_{24\text{max}} = P_{\text{max}} \times t_{24} \]  

(1)

where \( t_{24} = 1.13 \) is the ratio between average daily maxima and average 24-hour maxima. This ratio is empirically derived from 86 stations that had 15 years of concurrent data. Comparing the conversion factors to past NOAA volumes and other studies finds that the conversion value is comparable if not the same. The 24 hour annual maximum depth data was then converted to partial duration data series using

\[ P_{\text{Amax}} = P_{24\text{max}} \times \frac{T_{\text{AMS}}}{T_{\text{PDS}}} \]  

(2)

The parameter \( T_{\text{AMS}} / T_{\text{PDS}} \) is equal to 1.58 and represents the frequency ratio between an annual maximum series and a partial duration series. This ratio allows for multiple large storms in a single year to be considered in the final value such as occurred in Clemson, SC in 2013. The partial duration series was averaged and converted into a set of 24 hour storm depths of specified return period using

\[ P_{n,\text{yr}} = P_{\text{Amax}} \times RGF_n \]  

(3)

where \( n \) is the return period in years. The Regional Growth Factor (RGF) for each return period depends on the location of the rain gauge and is given in the NOAA Atlas. Distribution of the regions for the RGF can be seen in Figure 2. For example, since the station in Clemson, SC (Station ID 38-1770) is assigned to NOAA Region 12, its RGFs for the 2, 5, 10, 25, 50, and 100 year return period storms are 0.907, 1.196, 1.429, 1.801, 2.148, and 2.272 respectively (Bonin, et al., 2006). Using the same frequency analysis technique employed by NOAA allows for direct comparison of the GCM precipitation frequency values to the precipitation frequency values reported by NOAA based on historical rainfall data.

RESULTS

Results are presented for changes in Annual Total Rainfall (ATR) and for the 24 hour storm depth for 2, 5, 10, 25, 50, and 100 year return period storms. Because much of the data presented is location specific, Clemson, SC was chosen as a case study and is represented in many of the figures herein to illustrate a typical location. There are also figures that summarize this data for the entire state of South Carolina.

Changes in annual total rainfall

For each NOAA precipitation gauge location the daily time series of historical rainfall data and each downscaled GCM data set was converted into an ATR time series. A plot of the 134 ATR time series from 2015-2099 along with the historical recorded data from 1948-2011 for Clemson, SC are shown in Figure 3. The data shows significant year to year variation in the historical recorded data and a similar level of variation across the different GCM data sets presented. There is also a steady increase in the GCM predicted ATR over time. This is seen more clearly in Figure 4 which shows the mean and standard deviation of the historical data along with the yearly mean and standard deviation from the 134 GCM data sets. Note that there is a slight jump in average ATR from the historical mean to the start of the GCM time series. However, this discontinuity is well within the range of variability observed in both the historical and GCM projected data.

The downscaled GCM data shows a clear increase in the ATR over time. However, a histogram of the ATR from 2089-2099 for each of the 134 GCMs shows only a slight increase in mean ATR compared to historical records (see Figure 5). To verify that the increase is statistically significant a T-test was performed to compare the historical data with the GCM data for the last eleven years of the century (2089-2099). The T-test showed that the difference in the means
The data and analysis above was for a single location, Clemson, SC. Similar analysis was conducted for each of the precipitation gauge locations throughout the state. All locations showed an increase in ATR between 2015 and the end of the century. However, the net increase in ATR historical mean and standard deviation in ATR was compared to the mean and standard deviation of the ATR for 2015 based on all 134 GCM realizations. These data varied across the state. There was also an offset between the predicted 2015 mean ATR based on 134 GCM data sets and the historical record. At each gauge location the ATR is compared to historical values, which are plotted in Figure 6 and Figure 7. Figure 6 shows a scatter plot of historical mean ATR versus 2015 GCM mean ATR. The offset between the historical mean and the 2015 mean varies by location though the 2015 GCM mean ATR is almost always larger than the historical mean ATR. This would be expected for a climate with increasing mean ATR as the historical record would average over a non-stationary data set and would, therefore, underestimate the current mean ATR. Figure 7 shows the standard deviation in the historical ATR versus the 2015 GCM ATR standard deviation. Again the difference varies with location though in this case the standard deviation is not consistently higher or lower for the GCM data. The historical data shows a greater range of standard deviations compared to the GCM data, though this is likely due to the smaller number of data points in the historical data sets used in this analysis (average 41 years of data, 14 year standard deviation) compared to the 134 data points for the 2015 GCM ATR standard deviation.

Given the variation in both mean offset and predicted standard deviation it might be somewhat misleading to simply present the difference between the historical mean and the mean averaged over the later years of the century. Instead, we present data for the projected change in ATR based on a linear curve fit through the mean ATR for the GCM data from 2015-2099. Straight lines were fitted through the mean GCM ATR for each location. The slope of this line (with units of in/year) was then multiplied by 84 years (the GCM POR) to give a projected change in ATR over the remainder of the century. The data from each station was then entered into ArcGIS by ESRI where the geographic data information...
was interpolated using a tensioned spline method to create contour surfaces. A tension spline interpolation results in a surface that is less smooth but more closely constrained by the inputted data. This contour plot is presented in Figure 8.

Figure 8 shows significant variation in ATR change from 2.3 in for certain parts of the Savannah River basin to over 3.5 in in the coastal region, especially Charleston and Horry County. Much of the upstate and the length of the Savannah River Basin are all predicted to see lower levels of ATR increase compared to the rest of the state. The exception to this is the northern section of the border between Greenville and Spartanburg counties which will see ATR increases of around 4 in.

Changes in 24-Hour Design Storm Depths

Stormwater design in South Carolina is generally based on the 2, 10, and 100 year return period storms (DHEC 2002). Therefore, it is important to see how these design storm depths change over time, especially in comparison to the current NOAA return period data. In a changing climate the idea of a return period storm is not clearly defined. However, given 134 annual time series per year it is possible to get reasonable estimates of 2, 5, 10, 25, 50, and 100 year return period 24 hour storm depths for each year in the GCM POR and analyze how they change over time. A sample plot of the variation in storm depth for Clemson, SC is shown in Figure 9 along with the current NOAA values for the same return periods.

As with the ATR, the 24-hour storm depths are also seen to increase over time for each return period. However, there is also a difference between the historical record and the 2015 GCM projection for the each return period storm. In this case, the 2015 GCM data is lower than the NOAA value for the 2 year storm and higher than the NOAA value for the 100 year storm. In general the 2015 GCM projections for the 100 year storm were higher than current NOAA values though not always. Figure 10 shows a histogram of this difference for the 101 precipitation gauges analyzed as part of this study. The vast majority of locations have a difference of less than 1 in though some exhibit differences of up to 4 in. Twenty stations had 2015 GCM 100 year 24 hour storm depths lower than the current NOAA data. Regardless of the offset between 2015 GCM predictions and current NOAA data there is a clear upward trend in all six return period storm depths. Therefore, as with the ATR data, the projected change in depth is reported. Lines were fitted through the yearly return period depths for each return period and each precipitation gauge. The slope of these lines was then used to calculate the projected increase in storm depth by the end of the century across the state.

As with the mean ATR, there is significant uncertainty in the calculated values of 24 hour storm depth for a given return period. As such, NOAA reports the calculated depth and the depths at the extremes of the 90% confidence interval. For each rain gauge location, the projected year at which
that average total annual rainfall will increase across the state and that the GCM calculated storm depth exceeded the upper range of the 90% confidence interval for the historical data was calculated. Histograms of this year for each of the calculated return period storms are shown in Figure 11.

The data shows that there is a larger change in the longer return period storms. For example, most locations will not see the 2-year storm depth exceed the current NOAA 90% confidence interval value until well into the next century whereas most locations will have 100-year storm depths that exceed the current 90% confidence interval in the next few years. The year in which the GCM trendline exceeds the current 90% confidence interval is sometimes greatly outside the simulation period of record and should, therefore, not be taken as predictive. However, the data clearly shows that longer return period storms will exceed the current 90% confidence interval sooner than smaller storms.

The linear fits for each location and each return period were used to create contour plots of the total change in depth predicted over the GCM POR. The slope of each line was multiplied by 84 (the number of years in the POR) to calculate a change in depth. This approach is the same as that used for calculating changes in mean ATR over the GCM POR and ignores any offset between the 2015 GCM data and historical data. This offset is discussed below. A contour plot of the projected depth change for each return period storm is shown in Figure 12. The GCM data projects that the 100 year storm depth will increase by between 0.5 in and 1.2 in over the next 84 years whereas the 2-year storm depths only increase by between 0.2 and 0.5 in. As with the ATR data there is significant variation across the state with the largest increases in similar regions to those that were predicated to have the largest increase in ATR.

One possible explanation for the 2015 GCM 100 year storm depth being different, and typically deeper, from the current NOAA data is that the climate has already been changing over time. If this is the case, and the extreme event depths have been increasing over time, then there should be a correlation between the GCM 2015 to NOAA difference and the projected change in 100 year storm depth as plotted in Figure 12. Figure 13 shows a contour plot of the GCM 2015 to NOAA difference for the entire state. Visual comparison between Figure 12 and Figure 13 indicates that the regions of higher storm depth growth (darker regions of Figure 12) correspond to regions of greater initial difference in depth (darker regions of Figure 13). Further evidence of this relationship is shown in Figure 14 which shows scatter plots of the initial difference versus projected change for each of the return periods considered. Again, a clear correlation is observed between the offset and the projected rate of increase in storm depth.

CONCLUSION

The projected increases in both average annual total rainfall and design storm depths have the potential to stress existing stormwater infrastructure. The increases may also require regulatory agencies to re-visit their published design storm depths. One possible approach to mitigating the impact of these changes is to require new developments, as well as re-developments and retro-fits, to more closely replicate the predevelopment site hydrology. This could be done through the use of low impact development (LID) best management practices (BMP) to encourage infiltration and on-site runoff management. Such an approach has the potential to make new development more resilient to the projected changes in rainfall patterns.
Figure 11. Histograms of the year in which the 24 hours storm depth will exceed the current NOAA 90% confidence interval upper limit using the GCM trendline equation. Reading from top and left to right, 2, 5, 10, 25, 50, and 100 year return period storms. The vertical red lines represent the GCM simulation POR.
Figure 12. Contour plot of the GCM prediction of the change in 24 hour design storm depth (inches) over the forecast period. Reading from top and left to right, 2, 5, 10, 25, 50, and 100 year return period storms.
Figure 13. Contour plot of the offset between the 2015 GCM 100 year storm and the current NOAA data.

Figure 14. Scatter plot of the offset between the 2015 GCM 24 hour storm depth and the current NOAA data versus the projected growth in storm depth over the next 84 years. Reading from top and left to right, 2, 5, 10, 25, 50, and 100 year return period storm.


Model Results and Software Comparisons in Myrtle Beach, SC Using Virtual Beach and R Regression Toolboxes

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Abstract. Utilizing R software and a variety of data sources, daily forecasts of bacteria levels were developed and automated for beach waters in Myrtle Beach, SC. Modeled results are then shown for beach locations via a website and mobile device app. While R provides a robust set of tools for use in forecast modeling, the software has an extensive learning curve and requires skilled statistical interpretation of results. The Environmental Protection Agency (EPA) created the “Virtual Beach” software package to address these concerns. To evaluate the utility of the more user-friendly Virtual Beach modeling toolbox, predictive models were developed and model results were analyzed using the two software suites. Recommendations were made based on ease of use and several performance measures. Model results indicate the two software toolboxes yield comparable outputs. However, Virtual Beach tends to create more robust model forecasts, while R provides more options for model setup and outputs.

INTRODUCTION

As more people live, work, and play in coastal areas, an increasing need exists to provide robust and timely measures of potential illness risk from fecal water pollution, while ensuring that local economies are not harmed by unnecessary beach closures and advisories. To help accomplish this goal, new forecast tools were developed through the collaborative efforts of the University of South Carolina (USC) Arnold School of Public Health, University of Maryland Center for Environmental Science (UMCES), and National Oceanic and Atmospheric Administration (NOAA). Eight beaches (Figure 1) in the Myrtle Beach Grand Strand area of South Carolina now have daily forecasts for bacteria concentration in swimming waters. Radar-based rainfall estimates and coastal ocean observing system platforms provide real-time environmental data used in these new tools. Enterococci concentration estimates are provided in near real-time. These estimates (forecasts) are then uploaded to a database linked to a website and mobile device application. From here, bacteria concentrations and swim advisories can be seen and compared to EPA water quality criteria for swimming safety.

Previous research and bacterial estimates relied on weekly monitoring program results and a network of rain gauges (Johnson 2007; McDonald 2006). The near real-time models analyzed here offer many advantages and advances over existing monitoring and assessment approaches. First, remote sensing allows rainfall data to be collected and averaged over watersheds. According to Kelsey et al. (2010), areally averaged rainfall values provide more predictive capability for bacteria concentrations than point estimates obtained from rain gauges. Second, remotely sensed data products can be collected, collated, and processed in automated fashion. Computed bacteria concentration estimates can be provided daily and without the need for costly and maintenance intensive rain gauges.

Figure 1. Locations of sampling sites and model areas.
Alternative technologies and software tools have been utilized to model bacteria in coastal waters. EPA’s Virtual Beach (VB) software suite was developed for beach recreation areas. This software package provides many statistical tools needed for beach modeling including several of the tools used in previous Myrtle Beach forecasting efforts. In conjunction with the EPA, a need was identified to compare the performance of the existing Myrtle Beach models with those derived from VB.

The purpose of this project was to compare and contrast R and VB modeling software packages in terms of model development procedures and performance results. The Virtual Beach software package is designed to be relatively simple to use by those without statistical background. If the models developed using VB had similar predictive power to those developed using a more manual process in R, it would suggest that VB is a useful tool for developing predictive models for beach bacteria. Bacteria prediction results and the processes used to derive them were analyzed quantitatively and qualitatively when developing new predictive models in the Grand Strand.

METHODS

Data for this analysis were previously collected and summarized as part of a beach water quality prediction project. Data were collected in 2006, 2007, and 2009. These data represented many input and survival factors (Figure 2) necessary for the propagation of bacteria in marine waters. They were collected weekly and were representative of a wide variety of climate and environmental conditions. A common set of data (bacteria concentration, remotely sensed, modeled, and observing system data from varied sources [Table 1]) were included in the models. Enterococci bacteria concentration (culture forming units [CFU]) data were collected approximately weekly from the mid-May to mid-October beach swimming season. These data were compiled into a single .csv file for use in the following modeling processes. In both modeling efforts, multiple linear regression (MLR) was used to analyze multiple explanatory variables.

R Model Development

R, a free statistical software suite, is command-line oriented and must utilize the R language, similar to the S coding of S-Plus. R is open-source and supported and documented by a large user-base (Revolution Analytics 2015; R Core Team 2013).

In R, all potential parameters/predictors for the dependent variable were utilized. The dependent variable, Enterococci concentration, was log transformed to approximate a normal distribution and facilitate further standard statistical analysis. Data were imported via the common .csv file. Sample stations were reassigned as categorical variables so they could be analyzed as potential predictors. To compare results, the “relevel()” command in R was used in the categorical analysis of station location. This allowed the same sample stations to be used for model development in R and VB. No other data pre-processing was performed.

Models were then developed for each of the eight beach regions using linear regression. These locations were delineated based on South Carolina Department of Health and Environmental Control (SCDHEC) sampling station groupings. A backwards, manual selection process was used. The lm, or linear model, function in R was employed. Model “lm is used to fit linear models. It can be used to carry out regression, single stratum analysis of variance and analysis of covariance…” (R Core Team 2013). Variance inflation, parameter p-value, and model Bayesian Information Criterion (BIC) were used in selecting the models with the highest

Figure 2. Input and survival factors for bacteria (Kelsey et al. 2010).
predictive power. Because many of the predictors were related (e.g., rainfall averages of different length), variance inflation was evaluated. By deleting parameters with high Variance Inflation Factor (VIF) values (> approximately 10) in the model, unpredictable variance was kept to a minimum. Model selections proceeded by systematically removing parameters from the model until parameter p-values were approximately less than 0.10. BIC was used to evaluate remaining model parameters by removing parameters individually and exploring their effects on BIC. A lower BIC value was more desirable than a higher one. Final models retained parameters with variance inflation values less than 10, p-values generally less than 0.05, and lowest possible BIC values.

Virtual Beach Model Development

The EPA developed Virtual Beach 3 as a decision support tool incorporating suite of statistical software (Cyterski et al. 2013). The tool allows decision-makers and beach managers to predict fecal bacteria concentration using linear relationships between independent and dependent parameters. VB provides a list of model outcomes for the user to analyze (Cyterski et al. 2013).

VB 3 and 2.2 Users’ Guides (Cyterski et al. 2013; Cyterski et al. 2012) were utilized as outlines for developing models in VB. The same .csv data file used to develop models in R was analyzed. Dummy variables were created to test whether sample location, a categorical variable, was significant in model predictions. Data were imported and “validation” procedures were performed. Blank columns, rows, columns with missing data, or non-numeric records were deleted. Next, study sites were located along their respective beaches. A map feature, using Google Earth, was provided and an orientation box was created. From this box, an angle was generated which allows a wind, wave, and/or current component to be calculated and used in the modeling process. Since wind speed and direction were collected in the initial dataset, a wind component was generated for wind values perpendicular to the shore (O) and along the shore (A).

Multiple linear regression options were run on both standard and transformed (independent variables) datasets. The standard dataset included raw data with only wind components added. The transformed version contained independent variables that were transformed (e.g., Log10, ln, inverse, square, square root, quad root, polynomial, and exponential functions) and included if they met a 25% threshold for the Pearson correlation coefficient with respect to the dependent variable.

Using the MLR tab, independent variables were chosen in the variable selection tool under model settings. Model fitness can be analyzed using any one of ten model evaluation criteria (e.g., R2, adjusted R2, AIC [Akaike’s Information Criterion], BIC, Sensitivity, etc.) under the Control Options tab. BIC was chosen because it tends to limit over-fitting, keeping the number of variables in the model small (Cyterski 2013). Then, VIF levels were set to a maximum of 10 (VB can monitor this automatically). By checking the “Run all combinations box” under the manual option for linear regression modeling and clicking the “Run” button, VB evaluates models generated with all possible combinations of predictors. VB then automatically selects the 10 models with the best performance as determined by the evaluation criterion. The best model, having the lowest BIC (and, in general, the highest adjusted R2), was selected for further evaluation and comparison to the models developed in R.

Performance Metrics

AIC, BIC, adjusted R2, cross validation Mean Square Error of Prediction (MSEP), and Receiver Operator Characteristic curve (ROC) area under the curve (AUC) were used to compare performance of the models developed in R and VB. AIC, BIC, and adjusted R2 values help determine if additional parameters add predictive capacity to the model given the uncertainty introduced by adding an additional predictor. Cross validation allows evaluation of a fixed set of parameters in the final model; it uses random subsets of the original data set to develop parameter estimates and uses the remaining data to validate and compare observed values to the values predicted by the model. ROC curves (like those displayed Figure 3) were utilized to compare true positive to false positive values generated by the model. Curves like those seen in Figure 3 with high true positives (high sensitivity), low false positives (high specificity), and a steep transition are desired. Curves are compared by calculating the AUC. A perfect model would have an AUC=1, and a model with no predictive capability would have an AUC=0.5 (Morrison et al. 2003). In Figure 3, 2.02 represents the log10(104), where 104 is the Enterococci concentration guideline for recreation. The color code and the right scale represent the false positive and true positive rates at a particular decision point. Red represents a decision point approaching 2.7, where false positive and true positive rates are both 0. Blue represents the false positive and true

![Figure 3. ROC curve for the MB1 site.](image-url)
positive rates approaching decision point 0.75, where false positive and false negative rates are 1. This can be used to determine the decision rule at an acceptable false positive and false negative rate.

Following evaluation of all model criteria (AIC, BIC, R², adjusted R², MSEP, and ROC area) a matrix was generated to compare performance metrics for models at all locations developed in R and VB (Table 2). Each model was given a score of 0, 0.5, or 1 based on a comparison of performance metric values. A score of 1 was given to the most desirable metric value, while the least desirable was scored 0. Where two models tied for the most desirable metric value, a score of 1 was given to both while the remaining model was given a score of 0. Scores for each set of models were tallied. The model with the highest overall point value would represent the model with overall best performance.

A qualitative assessment of the modeling process was also performed. Overall software utility and methodology were evaluated. Ease of use, flexibility, utility of inputs/outputs, etc. were evaluated for R and VB. Each software package was analyzed for simplicity, learning curve required, flexibility of input data and output results, and the overall usefulness of the software.

RESULTS

Results and performance metrics for each model are summarized in Table 2. When first run in R, values for AIC, BIC, and cross validation were very different from VB. This was likely a result of the pre-processing step that VB uses to remove records with missing values for any potential parameters. In R, missing values were removed systematically, only removing records that have missing values for the parameters used in the model. To standardize comparisons, the dataset generated by the pre-processing step in VB was also used in R, resulting in identical data inputs. Model scores were generally highest for the VB model developed with transformed data, next highest for the models generated in VB with non-transformed data, and lowest for the models generated in R. Based on Table 2, VB transformed had a summed score of 37, VB was 21, and R was 16.5. The table also shows the VB transformed column having more green (highest point value) than either of the other two columns, while the R column had more red (no point value) than the other columns.

DISCUSSION

For investigations of Enterococci bacteria in beach applications, VB and R software can be useful for regression analysis and bacteria predictions for differing reasons; each has its strengths and weaknesses.

Quantitative Comparisons

Performance comparisons suggest that VB can generate more robust models than the simple linear regression manual selection techniques used in R for this assessment. The features of transforming variables and model comparisons using all potential prediction combinations used in VB can somewhat be reproduced in R, but is probably unnecessary,
as these features are built in to the current version of VB. Most importantly, the quantitative comparisons suggest that model development can be improved by using input data sets with predictors that are transformed to create linear relationships with the dependent variable, and using a model selection technique that evaluates all potential combinations of the model parameters.

Qualitative Comparisons
VB and R offer many benefits to potential users. While model results were somewhat comparable, the manner in which model predictions were derived is different. VB enables users to create robust models by running all possible variable permutations. It provides options for transforming independent variables and/or calculating wind A/O values. The VB tool also has an easy to learn graphical user interface (GUI) that utilizes self-explanatory tabs for major functions. VB requires no programming skill and is fairly easy to learn. VB provides users with a no-cost option to expensive commercial-off-the-shelf software tools.

In comparison, R requires use of a command-line programming language and scripting ability. To become proficient in R, time and resources are necessary and would be required to replicate some of the VB options employed here (e.g., calculating potential predictor permutations, transformation of independent variables, etc.). However, R provides some flexibility and options that are currently not available in VB, including automating data input/output, direct linkage to databases, and flexibility in generating descriptive visuals and graphical output. Additionally, predictive models can be developed using a variety of advanced methods in R, and many others are developed every year. Currently, MLR, partial least squares (PLS), and gradient boosting machine (GBM) options are the only options available in VB.

Contributions to the Field
Over the last fifteen years, predictive models for Escherichia coli and Enterococci concentrations have been developed for fresh and marine waters (respectively). Francy et al. (2013) showed that relationships between bacteria concentrations and environmental variables could produce models for use in making near real-time forecasts at inland beaches. Work conducted by Paule et al. (2014) and Francy et al. (2006) utilized MLR analysis to model bacteria from environmental, water, and hydrological data. MLR was utilized by Paule et al. (2014) to determine which hydrogeological factors impacted indicator bacteria concentrations most. Francy et al. (2006) indicated MLR allowed for the determination of beach-specific explanatory variables. Employing similar MLR procedures to evaluate the best variables for bacteria concentration predictions, we also found explanatory variables are unique to beach location. Bacterial models were even developed by Frick et al. (2008) utilizing the VB toolset. Here, weather and environmental data were processed by VB’s MLR tool (similar to our efforts in Myrtle Beach) to yield now-casts and forecasts of bacterial concentrations for Huntington Beach, Lake Erie (Frick et al. 2008). Additional modeling efforts incorporated PLS techniques to predict bacteria concentrations and produced similar results to regression efforts (Brooks et al. 2012). The Brooks et al. (2012) study even led to the incorporation of its PLS techniques in VB. The bacterial modeling field continues to expand its statistical modeling tools in an effort to increase accuracy, functionality, and usefulness of predictions for forecasts.

The results of this study are not shocking or ground-breaking. They do, however, reaffirm the importance of making accurate and timely estimates of bacteria in beach waters where permanent swimming advisories may not be in place (e.g., Florida beaches, where sampling is utilized to monitor bacteria levels) to ensure public safety. In SC, these results suggest that SCDHEC could remove permanent advisories and use the model results to determine when advisories should be issued for a particular site. The methodologies and comparisons highlighted in this study can certainly be applied in other beach areas. By utilizing VB, R, MLR, etc., accurate and precise forecasts can be employed by beach managers to ensure public health is impacted minimally. These tools and methodologies can be added to and extend the capabilities of any beach manager’s toolbox.

CONCLUSION
Overall, VB is recommended for model development in situations where programming skill is limited. If descriptive graphics and multiple input/output functions are needed, R software should be utilized. To match R’s automated data integration, additional programming, support, and funding of VB are recommended to increase tool functionality. The geographic footprint and ensemble modeling approach used here continues to expand; most notably with freshwater bacterial modeling recently completed in the Lower Saluda River of South Carolina and Enterococci concentrations currently being modeled in southwest Florida.

ACKNOWLEDGMENTS
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LITERATURE CITED


Groundwater Recharge Rates in Isolated and Riverine Wetlands: Influencing Factors

Chenille Williams\textsuperscript{1} and Dan Tufford\textsuperscript{1}

\textbf{Abstract.} Isolated wetlands and riverine wetlands have been shown to have similar groundwater hydrology despite their difference in topography and surface water hydrology. The current study aimed to address the impact of topography and surface water hydrology on groundwater hydrologic behavior by comparing the groundwater recharge rates of several isolated and riverine wetlands in the Coastal Plain of South Carolina. Study sites contained an isolated wetland, a riverine wetland, and an upland that bisected the two wetland types. Shallow water tables and sandy soils, allowed a rapid response to precipitation to be clearly visible. Soil characteristics, water table fluctuations, and precipitation data from January 2012-September 2012 were evaluated and from that data mean recharge rates were calculated using an adapted version of the water table fluctuation method. During the study period, it was observed that the frequency of precipitation (storm events) and saturated zone soil type were more impactful on water table movement than topography, surface soil type, and surface water hydrology. One significant finding of this research is that the isolated wetlands in this study did, in fact, recharge groundwater, which implies that their presence increases the opportunity for groundwater replenishment.

\textbf{INTRODUCTION}

One of the many functions of wetlands is the ability to capture stormwater runoff and recharge groundwater (Richardson, 1994; van der Kamp and Hayashi, 1998). Studies have suggested that riverine wetlands and geographically isolated wetlands may share that hydrologic capability (SEIWA, 2011), but further research into isolated wetland groundwater hydrology is needed.

Isolated wetlands are located throughout the United States, with characteristics that vary with geographic location, climate, and geomorphology. These microhabitats are called depressional wetlands, as they have a slightly depressed topography surrounded by an upland area. Most notably, isolated wetlands have no immediate surface water connection - a direct contrast to riverine wetlands, which often serve as riparian zones. One component of the water budget of both wetland systems is groundwater recharge - the addition of water to a subsurface aquifer. This type of input is valuable because it functions as a water source during low river flows and low precipitation, and its abundance affects human, animal, and plant populations (Richardson, 1994; Achayra and Barbier, 2000). Groundwater recharge rates have implications for shallow groundwater quality and those rates can be impacted by many factors including climate, topography, soil saturation, and soil texture.

With groundwater being a drinking water source for rural residents and an irrigation water supply for agriculture activity, groundwater hydrological processes are considered when assessing the water budget of an ecosystem and accounting for groundwater supply replenishment. Because groundwater is such a valuable resource, it is important to understand factors that may affect recharge processes. The objective of this study was to explore the groundwater hydrology of isolated and riverine wetland systems, compare their recharge rates, and assess factors that influence their recharge capabilities.

Over time, it has been recognized that riverine wetlands provide recharge opportunities; however little research specifically on recharge in isolated wetlands has been conducted in the Southeastern United States. Findings of this nature often become the basis of conservation laws, for which there may be a need of in many states. When making decisions, land managers and owners may not always have an interest in groundwater resources. Thus, it is up to state regulation to provide directives on groundwater protection. Knowing what factors affect groundwater supply (and potentially surface water quality) can be advantageous when making land disturbance permitting decisions.

Isolated wetlands are located throughout the United States, with characteristics that vary with geographic location, climate, and geomorphology. These microhabitats are called depressional wetlands, as they have a slightly depressed topography surrounded by an upland area. Most notably, isolated wetlands have no immediate surface water connection - a direct contrast to riverine wetlands, which often serve as riparian zones. One component of the water budget of both wetland systems is groundwater recharge - the addition of water to a subsurface aquifer. This type of input is valuable because it functions as a water source during low river flows and low precipitation, and its abundance affects human, animal, and plant populations (Richardson, 1994; Achayra and Barbier, 2000). Groundwater recharge rates have implications for shallow groundwater quality and those rates can be impacted by many factors including climate, topography, soil saturation, and soil texture.

While there is an overall variation in the topography of isolated and riverine wetlands, the hydropatterns of both systems create the opportunity for the development of hydric soils. Soil profiles vary regionally and the presence of a hydric soil has to be made based on the evaluation of the soil in each specific location. Pore size within the texture of a hydric soil determines the speed at which the pore pressure equilibrates (Williams, 1978). As a result, soil textures with
large pores allow water to move more readily than soil textures with small pores. Little research has been conducted to directly assess the similarity between the soil profiles of isolated and riverine wetlands within close proximity of one another - a factor that may influence the similarities between their recharge capabilities.

Until recently, most of the isolated wetland research has focused on prairie potholes in the Midwestern United States. Although that research provides insight on general isolated wetland behavior, the same behavior cannot be expected of wetlands in the Southeastern US, such as Carolina Bays and pocosins, due to the different climate, geomorphology, and wetland type. Since 2010, several studies have focused specifically on the hydrology of isolated wetlands in the southeastern region of the United States. Callahan et. al. studied the groundwater recharge rates of several isolated wetlands in South Carolina (2012), while the Southeastern Isolated Wetland Assessment (SEIWA, 2011) and the Hydrologic Connectivity, Water Quality Function, and Biotic Criteria of Coastal Plain Geographically Isolated Wetlands study (IWC, 2013) both assessed the surface water quality, groundwater quality, and groundwater nexus between isolated and riverine wetland systems. Additional research will increase the current body of knowledge about isolated wetland systems and how their functions compare to riverine wetland systems.

METHODS

In this study, recharge was defined as a change in water table height as caused by water percolating through the vadose zone to the zone of saturation (Lerner et. al., 1990; Devries and Simmers, 2002). The sites used for this study were within wildlife management areas in Marion County (Site MA and Site MF) and Horry County - both located in the Coastal Plain of South Carolina. Each of the three study sites contained two wetlands - one isolated and one riverine - and an upland that bisected the two wetlands types (Figure 1).

Groundwater Monitoring

At each site, a transect of groundwater monitoring wells was installed in the surficial aquifer from the isolated wetland to the riverine wetland (Figure 1). Each well location was identified with a “sub-site” based on its placement within the site. Isolated wetland (IW) indicated the edge of isolated wetland. Upland identified the upland area between the two wetlands. Connected wetland (CW) identified a location at the edge of the riverine wetland. Riverine wetland (RW) referred to a location in the riverine wetland that is closer to the surface water. Across all three sites, a total of 13 wells were installed and outfitted with pressure transducers whose accompanying software translated water and air pressure measurements to changes in water table depth. Water level

![Figure 1. Layout at LB site in Horry County, SC. IW indicates edge of isolated wetland, Upland indicates upland area, CW indicates the edge of the riverine wetland and RW indicates a location in the riverine wetland closer to the surface water.](image)
loggers were programmed to record hourly temperature and depth to water from the top of the well’s casing. Logger data was downloaded every two months from January 2012 - September 2012. During each download event, a discrete water level measurement was taken using an electronic water level meter. This data was used to establish an initial depth to water measurement from a designated measuring point at the top of the well casing (from which the logger was calibrated), and to correct for electronic drift of the loggers. Differential level surveys were also conducted to determine the elevation above sea level at the top of each well casing. Continuous monitoring data for each site was then compiled into hydrographs to analyze the water table’s behavior.

**Soil Profiles and Precipitation**

During the time of well construction, soil profiles were created to note changes in texture and/or color with depth. The observed profiles were compared to soil data from the Natural Resources Conservation Service for continuity. From the recorded data, stratigraphy maps for each site were created in order to display the underlying soil layers along the transect. Tipping bucket-style rain gauges that measured hourly air temperature and amount of precipitation were installed at each site in an open area to prevent overhead interception. Because of the sparsely interrupted overhead vegetation at the MF site, one rain gauge was used for both Marion County sites. Data from the rain gauges were also downloaded every two months during the same time the logger data was downloaded and the discrete water level measurements were taken.

**Recharge Calculation**

Recharge rates at each sub-site were calculated using the water table fluctuation (WTF) method, which is best used for unconfined aquifers (Healy and Cook, 2002) with shallow water tables that have a rapid response to precipitation (Moon et. al., 2004). The WTF method uses a water table budget to assume that a rise in the water table, as measured by an increase of water level height in a surficial groundwater well, is caused by recharge (Healy and Cook, 2002; Crosbie et. al., 2005). In an equation adapted by Callahan et. al. (2012), recharge is measured as:

\[ R = [S_y (h_u - h_a)] / \Delta t \]  

(1)

where \( R \) is the rate of recharge [cm/day] from the maximum water table depth \( (h_u) \) [cm] to the minimum water table depth \( (h_a) \) [cm], \( S_y \) [dimensionless] is the specific yield, and \( \Delta t \) is the duration of the recharge event [days] (Scanlon et.al., 2002; Healy and Cook, 2002; Callahan et. al., 2012).

Equation 2 was used to account for natural groundwater recession rate in the absence of precipitation in order to determine \( h_a \). The equation, which was originally used by Zhang and Schilling (2006) and adapted by Callahan et. al. (2012), is written as:

\[ h_a = h_i + h_d / (1 - e^{-\alpha t}) \]  

(2)

where \( h_a \) [cm] is the projected water table depth at the end of the recession period, \( h_i \) [cm] is the water table depth at the beginning of the recession period, \( h_d \) [cm] is the observed maximum water table depth at the end of the recession period, \( \alpha \) [d-1] is the recession coefficient, and \( t \) [d] is time.

Using a sub-set of the water level data, \( S_y \) values were calculated using a formula established by Williams (1978) and adapted by Callahan et. al. (2012). In the formula:

\[ S_y = P / \Delta h \]  

(3)

\( S_y \) is specific yield [dimensionless], \( P \) [cm] is precipitation, and \( \Delta h \) [cm] is the change in hydraulic head prior to the water table rise.

Using sub-sets of the data collected \( S_y \) and \( h_a \) were calculated. Those results were then used in Equation 1 to calculate the rate of recharge in response to designated rain events. Qualifying rain events had to fall within a certain range of duration, amount of precipitation, and time frame in order to be used. These restrictions were created to ensure a rise and fall could be attributed to a specific rain event.

**RESULTS**

**Soil Profiles**

All of the study sites were located in the Coastal Plain of South Carolina and underlain by sandy soils. Both the IW (Well 1) and RW (Well FL) at the LB site in Horry County contained silty loam topsoil (Figure 2). As shown in Figure 3, the topsoil at the MA site in Marion County contained a silty loam and loam at the CW (Well 3) and IW (Well 1) sub-sites, respectively. The topsoil at the MF, shown in Figure 4, site contained a loam and clay loam at the RW (Well 4) and IW (Well 1) sub-sites, respectively. The upland areas at each of the study sites contained a soil texture with a higher percentage of sand than that of either of the wetland sub-sites. Despite their different locations, and varying topsoil textures between the upland and wetlands sub-sites, each site was underlain by a sandy soil approximately 2.0 m in depth wherein the water table was located.

**Analysis of Recharge Rates**

In comparing the rates across all the study sites, the fastest rates were observed at the RW sub-sites in both Marion County sites (MA=5.73 cm/day, MF=5.90 cm/day), and the CW sub-site at the Horry County site (LB=5.22 cm/ day), as shown in Table 1. When the rates displayed in Table 1 are averaged, the riverine wetlands have an overall faster rate at 4.73 cm/day than the isolated wetlands at 3.29 cm/day. Because the calculated mean recharge rate does not indicate...
Table 1. Mean recharge rates± standard deviation (cm/day) per sub-site type

<table>
<thead>
<tr>
<th>Site</th>
<th>IW</th>
<th>Upland</th>
<th>CW</th>
<th>RW</th>
</tr>
</thead>
<tbody>
<tr>
<td>LB</td>
<td>3.32±4.05</td>
<td>3.11±3.11</td>
<td>5.22±3.52</td>
<td>2.56±1.87</td>
</tr>
<tr>
<td>MA</td>
<td>2.73±3.23</td>
<td>1.55±1.43</td>
<td>1.64±2.09</td>
<td>5.73±4.70</td>
</tr>
<tr>
<td>MF</td>
<td>3.81±2.34</td>
<td>2.97±2.88</td>
<td>-</td>
<td>5.90±6.18</td>
</tr>
<tr>
<td>All</td>
<td>3.29±0.54</td>
<td>2.54±0.86</td>
<td>3.43±2.53</td>
<td>4.73±1.88</td>
</tr>
</tbody>
</table>

* not all sites have the connected wetland (CW) sub-site

a significant different in rates between sub-sites (or sites), a MANOVA statistical test was run using land type (i.e. IW, CW, upland, RW) as a factor to determine if different wetland sub-sites produced different recharge rates. Based on the Wilks’ Lambda p-values (α = 0.10), there was no significant difference in the mean recharge rate between the different sub-sites within each site (LB=0.162, MA=0.157, MF=0.349). In other words, for each of the three sites, there was not a significant difference between the rates of all the sub-sites between the sites (e.g. the rate from collective IW data from all the sites was no different from the same data set from the RW collective data).
Analysis of Storm Events

A qualitative observation made during the hydrograph analysis was a difference in water table recession as the occurrence of storm events increased during the study period. Although the South Carolina State Climatology Office had declared a drought status during the early portion of this research, the study period was too short to infer that the observed changes were caused by climate variability. A distinction between the “wet” and “dry” periods was made based on the precipitation frequency, or frequency of storm events. For the Marion County sites, the dry period was from January - April 2012 and the wet period from May - June 2012. For the Horry County site, the dry period was from January - March 2012 and the wet period from April - September 2012. The dry and wet periods were also determined based on the variation in water table responses to change in precipitation frequency as observed from the hydrographs (Figure 5, Figure 6, Figure 7).

The change in precipitation appeared to be significant enough to impact the water table’s natural recession rate; as a result, a second MANOVA statistical test was run using precipitation frequency as a factor with the recharge events being categorized as occurring in either the dry or wet period. The Wilks’ Lambda p-values ($\alpha = 0.10$) for that analysis indicated that changes in precipitation frequency elicited a statistically significant impact on mean recharge rates at the LB site ($p=0.048$), MA site ($p=0.042$), and MF site ($p=0.103$). Although the type of wetland did not impact the rates, the amount of precipitation within a given period did.

DISCUSSION

The results of this study concluded that there was not a statistically significant difference in the mean recharge rates of the isolated and riverine wetlands used in this study. However, as the occurrence of storm events increased throughout the duration of the study period, there was a change in recharge rates observed at each of the wetland types. This change was noted as causing a statistically significant difference. Ultimately, weather patterns impacted groundwater recharge rates more than the type of wetland at which the recharge occurred.

The responses to weather patterns were based on the wet and dry periods established during the study period, and not necessarily not climate. Although the South Carolina State Climatology Office had declared a drought status during the early portion of this research, the study period was not long enough to definitively attribute any changes in weather to overall climate patterns. However, as the occurrence of storm events increased, the soil moisture and the hydraulic movement of subsurface water were impacted. Studies by Nolan et. al. (2007) and Callahan et. al. (2012) stress the relevance of considering deeper soil textures when analyzing groundwater behavior because hydrogeologic characteristics and water movement in the saturated zone contribute to the recharge rates in the unsaturated zone. The saturated zone at each of the study sites contained a sandy soil texture throughout each well transect. That persistent soil texture presumably drove the similar hydraulic movement of groundwater at each well location (in either an isolated wetland, upland, or riverine wetland area) and resulted in the similar recharge rates despite variation in wetland type and surface soil texture. There was a potential difference in infiltration and percolation rates due to the variation in surface soil textures, but the subsurface soil texture was more of a driving factor for groundwater behavior.

While an impact on rates was not observed for the different wetland types, an impact was noted for an increase...
Groundwater Recharge Rates in Isolated and Riverine Wetlands

The difference in recharge rates between the dry and wet periods may be a result of soil moisture content and the water table's ability to fluctuate. As the occurrence of storm events increased, the amount of available soil moisture also increased. In turn, the soils were more likely to be saturated throughout the soil profile, which would impact the water table's ability to fluctuate upon receiving percolating water. Less precipitation means less available water capacity, decreased soil moisture, and freedom for the water table to fluctuate as a result of the empty pore spaces. Additionally, each of the three study sites were underlain by sandy soils, through which water flows easily and resulting in a more dramatic change in water table movement. Soil type, particle size, pore size, and soil moisture appear to dictate groundwater movement. Those four variables are affected by the amount of precipitation in a given amount of time and potentially the climatic conditions.

One of the objectives of this study was to compare the recharge rates of isolated and riverine wetlands. While the wetland types in this study did not have different recharge rates, the isolated wetlands did, in fact, recharge groundwater. The influence of isolated wetlands on the groundwater of an ecosystem is not to be overlooked, nor is the suggestion that isolated wetlands recharge groundwater to same degree as riverine wetlands. As locations of recharge, the presence of isolated wetlands increases the capability for an area to replenish groundwater resources. One could even argue that because infiltrating water collects in the depression and surrounding groundwater follows the downward slope of the depression and remains in the depression, as opposed to discharging into a flowing surface water body, isolated wetlands recharge more groundwater than uplands or riverine wetlands. Decreasing the aforementioned opportunities to replenish groundwater should be considered by regulatory agencies when making permit decisions. It would be beneficial to further pursue this line of research to increase the knowledge about additional similarities or differences between wetland systems in the South Carolina Coastal Plain. It would also be valuable to expand the research to comparing different wetland systems in other regions of the Carolinas, such as the Piedmont or the Blue Ridge.

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LITERATURE CITED


Figure 7. Hydrograph of water table fluctuations and hourly precipitation at the MF site (Marion Co., SC) from January 2012 to September 2012.


Abstract. The expansive tidal salt marshes of South Carolina support a unique and sensitive ecosystem providing environmental and economic value to the coastal community. These tidal ecosystems are often altered by sea level rise through various processes, including the lesser-known stress of saltwater intrusion in groundwater systems. The goal of this research was to measure the baseline groundwater dynamics of an undeveloped tidal saltmarsh. Groundwater wells were installed along transects from the upland into the marsh and a culminating water budget of the watershed was developed. Analysis of water table dynamics showed that in the upland zone, evapotranspiration and precipitation were the dominant processes, whereas in the marsh zone and the uplands directly adjacent to the marsh, water table fluctuations were dominated by tides. An influencing feature for the site was the large tidal creek (Big Bay Creek), which is a tributary of the South Edisto River. The cut bank of Big Bay Creek was adjacent to the south end of the study site where tidal influence on the shallow groundwater was observed. The location of an ephemeral stream through the site was considered as a potential pathway for saltwater intrusion into the uplands, yet this was not confirmed. Groundwater response rates were likely influenced by the presence of fine-grained, well-drained sandy soils. Application of this research will assist coastal resource managers identifying pathways of marsh migration as driven by future sea level rise.

INTRODUCTION

Salt marshes support a collection of unique and sensitive ecosystems providing environmental and economic value to the coastal community. Storm protection, carbon sequestration, nutrient transformation, and fisheries support are a few of the benefits provided by healthy tidal salt marshes (Kirwan and Megenigal, 2013). However, their ecological viability may be threatened by sea level rise and land-use stressors such as coastal development. Furthermore, saltwater intrusion resulting from sea level rise may disrupt the hydrologic balance between the salt marsh and fresh upland groundwater system.

PROJECT DESCRIPTION

The groundwater system studied at this site was the surficial aquifer within the South Carolina Lower Coastal Plain region. This aquifer is unconfined so it is mainly subjected to infiltration of precipitation and areal recharge, as well as atmospheric pressure effects (SC DNR, 2009). Due to this exposure of the surficial aquifer to the surface, anthropogenic land-use practices are a defining threat to this groundwater system. Although a majority of groundwater systems contain fresh water, surficial aquifers in close proximity to tidal systems may contain saltwater (SC DNR, 2009). This study
focused on the area of marsh known as the MTU, which is similar to the high marsh, classified as only being flooded during very high tides twice a month from new and full moon phases (NOAA Ocean Service Education, 2008). Additionally, this study spotlighted the upland maritime forest bordering the marsh. In order to understand the relationship of groundwater movement between the marsh-upland zone, groundwater monitoring wells were installed in a triangulated network. The use of groundwater monitoring wells in the maritime forest and MTU zone allowed for data collection of various groundwater variables over an 11-month time period to highlight the monthly and seasonal dynamics, as well as to capture storm events. The primary objective of this research was to calculate the water budget for the watershed, which illustrated the influence of the surficial aquifer on the upland and marsh interface.

Additionally, the main goal of this study was to describe the groundwater dynamics that occur in the surficial aquifer at this marsh-upland interface. In order to satisfy this goal, the relationships among topography, potential evapotranspiration, precipitation, tidal amplitude and duration were identified. It was hypothesized that groundwater dynamics would mimic the topography of the watershed and salinity would decrease with increasing distance from the saltwater source, Big Bay Creek. Furthermore, the water budget in the upland zone of this coastal site of a maritime forest and adjacent tidal salt marsh should be dominated by water demand for evapotranspiration and precipitation, whereas in the marsh zone, tidal forcing should control the water budget.

METHODS

Study Area

The study site for this project is located in a maritime forest and adjacent undeveloped tidal saltmarsh along Big Bay Creek at Edisto Beach State Park within the ACE Basin, South Carolina. The marsh bordering Big Bay Creek is tidally dominated and the vegetation along the marsh study zone is characterized by *Spartina alterniflora*, *Salicornia virginica* (glasswort), and *Juncus roemerianus*. The upland portion of the study site is proximal to the marsh, and the topographic relief of the uplands to the marsh is about 2.5 m. The upland flora is consistent with a southern maritime forest. The maritime forest at this location is classified as a near-coast forest whose plant community is influenced by salt spray and typically is characterized by live oak, cabbage palmetto, Southern magnolia, red bay, yaupon, American holly, sparkleberry, wax myrtle, and saw palmetto (Whitaker et al., 2009). A distinguishing physical feature at this site is an ephemeral stream running perpendicular to Big Bay Creek.

The depth of the shallow surficial aquifer being studied at the site ranges from approximately a meter below mean sea level (BMSL) to 15 meters BMSL (Park, 1985). Beneath the surficial aquifer lies the Cooper Formation from 15 to 115 meters BMSL and the Santee Limestone/Floridan Aquifer from 107 to 189 meters BMSL (Park, 1985).

Field Study Collection

Wells were installed in a triangular pattern to determine the direction of groundwater movement and hydraulic gradient. Three wells were located in the uplands (North, Middle, and South) and three in the MTU (T5, T2 Shallow, and T2 Deep) (Figure 1). The T2 wells were coupled at varying depths in order to indicate whether there was a difference in groundwater readings based on the depth or the presence of a freshwater lens. Each of the wells consists of a solid PVC pipe connected to a screened PVC pipe to allow groundwater to flow through the bottom of the well. A bentonite seal was applied above the well screen to guarantee water was being monitored from the screen depth and not infiltrating from the surface. The well depths were dependent on the depth of the water table at each of the sites to guarantee a continuous groundwater supply in the wells. The varying lengths of the wells and screen depths are displayed in Table 1. Solinst levelogger instruments were deployed in each well using braided fishing line measured as string length (Table 1). The Solinst levelogger instruments allowed for 30-minute data collection of water temperature (C), electrical conductivity (μS/cm), water level (cm), and barometric pressure (kPa) from June 6, 2013 to May 5, 2014. For the purposes of this study, all electrical conductivity readings were converted to salinity (ppt) and groundwater data were compensated for pressure and temperature.

Mapping the topography of the study site was important to delineate the watershed and also to understand the

Figure 1. Site Map including NERRS Boundaries and well locations.
relationship between groundwater levels in the wells to relative elevation (AMSL). In order to determine the upland and marsh elevation for the well sites, traditional surveying was performed using an RTK Global Positioning System (GPS). By relating the elevation of ground surface of each well to height above mean sea level (AMSL), the water levels were established and related by use of a common datum at each well site. The watershed was delineated using ArcGIS from a digital elevation map constructed from LIDAR. The ground elevations and coordinates of each of the wells AMSL are displayed in Table 2.

Following the Solinst Levelogger Series User Guide-Version 4, water level inside each well (A) was calculated by the equation:

\[ A = L - B \]  

where (A) = actual water column height; (B) = Barometric pressure; (L) = levelogger total pressure reading. Water level readings were also temperature compensated using in-situ readings (Solinst, 2013).

In order to observe potential tidal influences from adjacent Big Bay Creek, water level and salinity data were retrieved from the NERRS CDMO. Additionally, soil characterization at each well site was also determined during well installation by grab samples every half-meter. Determining the soils and topography helped uncover the groundwater pathways within the watershed.

Additionally, vegetation surveys were carried out in order to more thoroughly analyze the type of vegetation affecting evapotranspiration conditions and to determine basal area. Monitoring basal area determines how much of an area is made up of tree stems (Walsh, 2010). The basal area per tree was summed for each site to determine the total basal area per well location. In order to carry out the basal area study, a 200 m diameter was plotted around each well and specimens were characterized at circumference breast height (CBH) and then converted to diameter breast height (DBH) by genus and species. The vegetation was broadly grouped by oak trees, pine trees, holly trees, dwarf palmetto, sabal palm, black gum, bald cypress, green ash, and red bay. The equation for determining basal area is:

\[ \text{Basal area per tree (sq. ft)} = 0.005454 \times (\text{DBH})^2 \]  

where 0.005454 converts inches into square feet and is called the “forester’s constant”; and DBH is equal to diameter at breast height per tree (Mississippi Wildlife, Fisheries & Parks, n.d.).

### Weather Data Collection

In order to calculate the water budget, precipitation and air temperature data were retrieved from a nearby weather station at Bennett’s Point, SC through the NERRS Centralized Data Management Office (CDMO) and converted into total daily readings. The Bennett’s Point weather station is located in an open field allowing for the collection of total precipitation with no threat to loss of rainfall from the tree canopy. However, because the Edisto well site is located in a forested upland, throughfall at this site is less than Bennett’s Point due to greater interception rates.

Throughfall was calculated for the dominant vegetation types: Eastern hardwood forests (Oak trees) and Southern pine forests (Loblolly Pines) to determine the amount of precipitation reaching the forest floor and the uncertainty of the total precipitation data. The throughfall equation for the Eastern hardwood forests during the growing season is:

\[ \text{Th} = 0.901 \times (P) - 0.031 \times (n) \]  

where Th is throughfall (in); P is total precipitation (in); and n is number of storms (Helvey and Patric, 1965). The equation used for the Southern pine forests for Loblolly Pine is (Roth and Chang, 1981):

\[ \text{Th} = 0.930 \times (P) - 0.0011 \times (P)^{2.61} \]  

The throughfall results were converted to millimeters and compared to the total precipitation amount. Precipitation compensated for throughfall of the Eastern hardwood forests was used for the calculation of the water budget.

Potential evapotranspiration (PET) was calculated using the Hamon model and an adjusted Hargreaves-Samani (H-S) model. In order to achieve a more accurate PET based on available weather inputs, an averaged PET of the two models...
was used in the water budget calculation. The Hamon model for potential evapotranspiration is:

\[ PET = 0.1651 \times Ld \times RHOSAT \times KPEC \]  

(5)

where PET is equal to zero when temperature is less than zero; Ld is the daytime length (x/12 hours); RHOSAT is the saturated vapor density; and KPEC is the calibration coefficient, which is 1.2 as determined from studies of the southeast United States (Lu et al., 2005).

Dai et al., 2013 successfully used an adjusted Hargreaves-Samani equation for their study at the Santee Experimental Forest in South Carolina by adding a coefficient to the original H-S equation (0.408) to convert extraterrestrial radiation from megajoules/sq. m./day into water evaporation depth at mm/day. An additional coefficient of 0.0021 was used in the coastal North Carolina region (Amatya et al., 2000). The adjusted H-S model supported by Dai et al. (2013) and Amatya et al. (2000) used is:

\[ PET = 0.408 \times 0.0021 \times Ra \times T^{0.5} \times (T + 17.8) \]  

(6)

where PET equals daily PET in mm/day; T equals daily mean air temperature (°C); Ra equals extraterrestrial solar radiation in MJ. m-2. day-1; TD equals the daily difference between maximum and minimum air temperature (oC).

Water Budget Calculation

In order to effectively characterize the groundwater flow in this system, a water budget must be determined. A water budget characterizes the inputs and outputs of water flow over a system. Water budgets are useful tools in identifying key pathways that water infiltrates, flows, and exits through a study site. The water budget is a measurement of the processes of the hydrologic cycle, which include precipitation, evapotranspiration, groundwater infiltration, and surface runoff (SC DNR, 2009). In this study, precipitation, groundwater inflow/outflow and evapotranspiration were included in water budget calculations. Runoff was not a factor due to the lack of impervious surfaces and flood inducing storms, as well as highly-permeable soils at the site. The water budget was calculated for over weekly and monthly timescales using the formula:

\[ \Delta S = P - PET + \Delta G \]  

(7)

where \( \Delta S \) is change in storage, P is precipitation, PET is potential evapotranspiration, and \( \Delta G \) is change in groundwater. Runoff was not included in this calculation due to the presence of sandy soils at this site and the lack of flood-inducing storms and impervious surfaces.

The change in groundwater (\( \Delta G \)) was calculated on a monthly timescale by obtaining daily 1:00 am readings for each well and subtracting the water table depth at the end of the month by the beginning of the month. The change in groundwater depth was additionally normalized for specific yield of the soil and sediments, that is, the available pore space for infiltrating water to fill. Specific yield was determined from five storm events that caused a rapid rise in water table depth (Table 3). Precursor conditions for these storm events included: (A) water level depth below ground surface could not be greater than 100 cm; and (B) a precipitation event larger than 15 mm caused the water level change. Specific yield (Sy) was calculated as:

\[ Sy = P / \Delta WT \]  

(8)

where P is the total amount from a precipitation event (mm), and \( \Delta WT \) is the change in water table depth (mm) subsequent to the precipitation event (Harder et al., 2007). The average specific yield was calculated from the five events and then multiplied by the change in water table depth to get the resulting change in groundwater (\( \Delta G \)) that was used to complete the water budget.

In order to understand the flow of groundwater across the site, Darcy’s Law was used to estimate groundwater flux for the upland area. The one-dimensional form of Darcy’s Law is:

\[ q = K(\Delta h/\Delta L) \]  

(9)

where q (m/day) is groundwater flux, K is hydraulic conductivity (m/day), \( \Delta h \) (m) is the difference in head between sites, and \( \Delta L \) (m) is the well separation distance (Fitts, 2013).

Hydraulic conductivity was estimated from the typical values of hydraulic conductivity based on sediment type from Davis (1969) and Freeze and Cherry (1979). The highest (103 m/day) and lowest (10-1 m/day) values for hydraulic conductivity for sandy soils were used to capture the range of possible conditions at this site. The \( \Delta h \) (m) also included both the highest and the lowest difference in head values between the north and middle upland wells, and also the same ranges between the south and middle upland wells in order to approximate groundwater flux toward the ephemeral stream channel where the middle well was located.

RESULTS

Groundwater Dynamics Per Well

The groundwater hydrograph analysis and water budget results showed that groundwater position over time was affected by both direct and indirect influences. Evapotranspiration, precipitation, and semidiurnal tidal
Figure 2. Groundwater and atmospheric dynamics over a 7-day period for the Middle Well. Night is shown as the dark vertical bars. Evapotranspiration-driven groundwater drawdown occurred during the day while groundwater recovery occurred at night.

Figure 3. Groundwater dynamics at the South Well compared to Big Bay Creek surface water level over a 7-day period.

Figure 4. Groundwater and atmospheric dynamics at the North Well occurring over a 7-day period. Nighttimes are the dark bars. Evapotranspiration-driven groundwater drawdown occurs during the day while groundwater recovery occurs at night.

Figure 5. Groundwater dynamics in the North Well compared to Big Bay Creek surface water level over a 7-day period.

Figure 6. Water table comparison among the three upland wells referenced to AMSL.

Figure 7. Close-up of T5 groundwater and salinity dynamics compared to Big Bay Creek surface water.

Figure 8. Close-up of T2 Shallow groundwater signature and salinity compared to Big Bay Creek surface water.

Figure 9. Close-up of T2 Deep groundwater signature and salinity compared to Big Bay Creek surface water.
signals directly influenced the upland groundwater wells, whereas lunar phases, topography, and seasonal variations in the tides indirectly influenced the groundwater. The main freshwater input to the aquifer for the three upland wells was precipitation-driven infiltration. Over monthly and seasonal timescales, groundwater dynamics were indirectly influenced by lunar phases and landscape position showing recharge under high elevation well sites and discharge at lower elevation sites. In particular, the middle well was the most sensitive to precipitation inputs and diurnal evapotranspiration outputs at a daily rate at the ephemeral stream (Figure 2). The south well was clearly influenced by a delayed tidal signature in the uplands (Figure 3), while the north well lacked a clear evapotranspiration or tidal signature over short term daily analyses (Figure 4 and Figure 5). Groundwater depth in the middle well occasionally reached close to the surface but generally remained around 70cm below the surface. The middle well also had the most dynamic groundwater flux, whereas the north and south well remained about 150cm to 300cm below the ground surface.

The upland groundwater data were converted from depth below ground to mean sea level to enable a comparison of water-level dynamics amongst the three wells. The results of the upland well comparisons showed that all three wells followed the same general long-term trend (Figure 6). The middle well deviated from the north and south wells by responding more dramatically to rain events and lacking an obvious tidal signal. The south, north, and middle wells differed in groundwater depth in that order from deepest to shallowest. The average groundwater elevation for the south well was 843cm, north well was 776cm, and middle well was 730cm.

The MTU wells were mainly influenced by tidal signals and to a lesser extent by precipitation and evapotranspiration, as evidenced by increased salinity readings in the fall and winter months when precipitation rates were low. In particular, the T5 well located in the northern marsh was primarily influenced by semidiurnal tidal patterns although there was a slight lag (1.0 to 1.5 hours) in groundwater highs and lows compared to the surface water of Big Bay Creek (Figure 7). Groundwater patterns at the T2 Shallow and T2 Deep coupled wells, located in the southern marsh, both were dominated by semidiurnal tidal patterns (Figure 8 and Figure 9). The groundwater highs and lows for the coupled wells occurred nearly simultaneously to those in the surface water.

Water levels in the T5 well generally remained at about 15cm below ground, but frequently rose above ground due to high tides and rain events. Water levels in the T2 Shallow and Deep wells were generally 35cm and 85cm below ground, respectively. Water levels in the T2 Shallow well infrequently rose above the surface, whereas the levels in the T2 Deep groundwater never did. The groundwater level in the deep well was typically 50cm below that of the shallow well. This difference in groundwater depth reflects a positive (downward) hydraulic gradient between the shallow and deep T2 MTU wells, which is partly due to the greater length of the deeper well and the lower depth of its screen below ground.

The water table elevation graph for the MTU wells referenced to AMSL, showed that all three marsh wells tend to follow the same tidal-driven groundwater pattern (Figure 10). T2 Deep and T2 Shallow were closer in water table elevation. During the first half of the study period, the water table patterns between T2 Deep and T2 Shallow were similar, showing more dramatic gains and losses compared to T5. However, during the second half of the study, during the spring and summer months, all three marsh wells showed clear water table gain and loss patterns.

Rain Event Response
Precipitation in the upland wells was a clear groundwater input factor, as evidenced by the August 14, 2013 rain event accumulating 56.4mm of precipitation (Figure 11). A snapshot of this rain event showed that the middle well rise in groundwater level occurred the same day that the rain event transpired, rising twice as fast in comparison to the other two wells over the same 90-minute period. The north and south wells showed a less dramatic increase in groundwater level during this rain event coming to a peak two days after the initial storm. All three wells then showed a gradual decline in the water table level indicating groundwater infiltration after the rain event. The ground elevations relative to sea level for the south, north, and middle are 1,071, 1,037cm, and 761cm, respectively.

A closer look at the groundwater response in the marsh wells during and following rain events can be seen in Figures 12 and 13. Figure 12 shows the response of the T2 deep and T2 shallow wells to the August 14, 2013 rain event (the T5 well did
The 11 Shallow and Deep water levels increased by about 40 cm over a five and six hour period while continuing to show a tidal signal. The ground surface elevations above sea level for the well locations were 390 cm (T5) and 260 cm (T2 deep and shallow). An additional rain event of 44.4 mm on November 26, 2013 highlights the response of T5 to rain events (Figure 13). This rain event showed that a general tidal signal was present for both T5 and T2-shallow wells until the rain event signal was diminished. The gain in groundwater level from this rain event was 22 cm in an 11 hour period for T2 shallow and 13 cm over an 8.5 hour period for T5. At this time period, the T2 Deep well did not have a functioning datalogger.

Salinity Variations

Although it was hypothesized that salinity would decrease with increasing distance from the creek, the upland salinity graph shows that this may not be the only contributing factor (Figure 14). In fact, the middle well had the highest salinity level at 30x greater than the north and south wells, although it was the furthest from Big Bay Creek. The middle well salinity was brackish in the earlier time of the study period. The north and south upland wells were considered freshwater groundwater systems since they were within the 0 - 0.5ppt salinity range. The salinity for the north and south wells also showed different patterns, particularly evident during the time periods of mid-October 2013 to February 2014.

The salinity variations in the marsh wells were relatively similar to each other (Figure 15). Both T5 and T2 shallow had similar increasing patterns although they were on opposite ends of the study site. This may be due to their comparable well depths. The salinity of the T2 deep well was more stable and could be due to the fact that the well was slightly deeper. The T2 shallow well had a salinity pattern that mimics the tidal signal seen in the groundwater level at this site. It is also clear that compared to the upland wells, the marsh wells’ salinity changed seasonally. The summer and spring months showed a generally lower salinity than the fall and winter months.

Main Input/Output Trends

Precipitation and PET were considered the main input and output factors affecting the water budget at this site. In general, precipitation was the greatest in the summer months (June-August) at 434 mm and lowest in the winter (December-February) at 85.5 mm. The seasonal precipitation pattern was typical of the South Carolina coastal areas (SC DNR, 2009).

In order to generate more accurate results for precipitation to use in the water budget model, throughfall was calculated for the dominant vegetation types: Eastern hardwoods and Southern pines (Figure 16). Throughfall was calculated...
using the total precipitation for the study period (892.7 mm). Throughfall totals were calculated for the Eastern hardwood (734.24 mm) and Southern pine forests (696.46 mm). The amount of precipitation that reached the forest floor, as calculated by throughfall, was 82.25% for hardwoods during the growing season and 78.02% for loblolly pine trees. Therefore, about 18% and 22% of total precipitation was intercepted by tree canopies for the Eastern hardwood and Southern pine forest types. The total precipitation was adjusted using monthly throughfall rates from the Eastern Hardwood Forest and was used in the water budget calculation to provide accurate site-specific results. It was apparent that the greatest difference between the original and adjusted precipitation occurred in the summer months (June - August) (Figure 16).

The Hamon model and adjusted Hargreaves-Samani model for PET were averaged on a daily and monthly scale to more accurately represent PET rates over the study period (Figure 17). This averaged PET was used as the PET input for the water budget calculation. Potential evapotranspiration comparisons showed that it generally followed the precipitation pattern: greatest rates were found in spring and summer and the lowest in the fall and winter (Figure 17). This pattern coincides with the hottest and coolest months of the years, as well as the growing and dormant vegetation periods, respectively. During the late spring and summer (June to September), potential evapotranspiration averaged about 126 mm/month then decreased in the fall and winter, eventually reaching the lowest PET value in January (34 mm/month). Evidence for the impact of evapotranspiration was seen in the diurnal groundwater level fluctuations in which the water table decreases during the afternoon due to peak drawdown and then rises to the surface at night or the early morning (Figure 2).

Water Budget

Precipitation corrected for throughfall, monthly averaged potential evapotranspiration, and groundwater storage (ΔG) were used to calculate the water budget (mm) on a monthly basis. The overall results of the water budget showed a water deficit, specifically from June to July, September to October, and January to March (Figure 18). Periods of balanced water storage conditions occurred during August, November, and December. April was the only month that had a water surplus for all well locations. The greatest change occurred in April when all six wells experienced a 70 mm increase in water storage (Table 4). The month of July 2013 is not representative of completed monthly results for the T2 deep and shallow wells which started recording water level on July 12. Additionally, due to datalogger malfunctions, the water budget could not be calculated for the T2 Deep location from November 2, 2013 - February 9, 2014.

Overall, the north and south wells maintained similar monthly changes in water storage. The middle well varied monthly with storage changes sometimes comparable to the upland (north and south) or marsh wells. The T2 wells had similar monthly changes in water storage throughout the entire study period and the T5 well only varied slightly from the T2 wells in storage change.

A monthly water budget of the middle well was chosen to represent the water storage along a groundwater discharge zone. (Figure 19). The groundwater table was close to the surface at this site and during a precipitation event, groundwater discharge and infiltration directly contributed to the change in water storage. During periods where the groundwater showed a water surplus, this may have indicated ponding at this discharge zone (Figure 19). In April there was a precipitation event, which caused a water surplus at the middle well. Based on the water deficit period over the preceding months, the antecedent water level was low and the large amount of precipitation in April caused the water to rise near the surface indicating the rapid response of groundwater level to water inputs (Figure 19).
Table 4. Monthly surplus(+)/deficit(-) in mm. N/A: wells not yet installed.

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**Topography and Groundwater Flow**

Upon analyzing the LiDAR DEM, it appeared that the coastal morphology was made up of historic dune ridges, causing the rise and fall of the elevation in a uniform northwest direction perpendicular to the Atlantic Ocean. The LiDAR DEM showed that the northern edge of the study area (north well) maintained a fairly high elevation around 10-15 meters AMSL and the south side of the site reached elevations of 8 to 10 meters AMSL (Figure 20). The middle well is located at a lower elevation (7 meters AMSL) adjacent to an ephemeral stream that discharges into Big Bay Creek and on its upstream side, reaches northeast outside of the study site.

Groundwater flow paths were determined from the LiDAR DEM because water generally moves from high to low elevation areas. Therefore, it was deduced that a majority of the groundwater is flowing from the uplands into the lower elevation ephemeral stream and along the topographic break downslope from the uplands to the MTU. Figure 20 also shows that a portion of the groundwater flows away from the site, particularly along the northern watershed boundary.

The results from Darcy’s Law calculations suggest that groundwater flow occurred at a faster rate from the south well to middle well as compared to the flow from the north well to the middle well. This is due to the slope of the hydraulic head across these sites. The groundwater flux from the south to middle well ranged from $1.48 \times 10^4$ to $3.75$ m/day. The north

![Figure 19](image-url)  
**Figure 19.** Monthly surplus/deficit of middle well over the study period. Negative: water deficit; positive: water surplus.

![Figure 20](image-url)  
**Figure 20.** LiDAR digital elevation model (DEM) map of Edisto Beach State Park. Elevation is provided in meters above mean sea level.
to middle groundwater flux ranged from $5.16 \times 10^4$ to 3.64 m/day. The differences in flow reflect differences in hydraulic conductivity (K) and head value (Δh).

**Soils and Vegetation**

Soil samples taken at each of the well sites were analyzed and classified by soil type. It was determined that the site is made up of fine-grained clean sand and loamy sand with a surface layer of organic material. There were also iron deposits found in depths reaching anoxic conditions on the north and south sides of the study site. The Natural Resources Conservation Service Web Soil Survey (n.d.) provided soil classifications that matched the general field classifications. The predominant soil type is Wando loamy fine sand (WnB), making up 76% of the area of interest while Capers silty clay loam is present only in the ephemeral stream.

The basal area was calculated at each site and showed that the middle well (0.30 sq. meters) and T5 marsh well (0.25 sq. meters) sites had the lowest basal area coverage. The north well (0.89 sq. meters) site had the greatest basal area coverage, followed by the south well (0.37 sq. meters) and the T2 marsh wells (0.35 sq. meters). Species dominance for each well site was also determined. The south well was dominated by two species of oak trees (Quercus falcata and Quercus nigra) making up 65% of the basal area at the site. Loblolly pine trees (Pinus taeda) were the dominant species at the north well making up 90% of the basal area despite stem count dominance from oak trees. The middle well basal area was dominated by sabal palm trees (Sabal palmetto) that comprised 63% of the total basal area. Oak species (Quercus virginiana, Quercus laurifolia, and Quercus nigra) dominated the T5 well site’s basal area coverage (78%) despite stem count dominance of pine trees. The dominant species contributing to basal area coverage at the T2 wells was a sabal palm (Sabal palmetto) (42%).

The basal area findings were dependent on the surrounding well locations measured out along the site. For this reason, sites that were located within a clearing or depression did not have as many trees to measure for basal area and therefore may not have been representative of their settings. For example, the marsh wells (T2 deep and shallow and T5) lacked measurable specimens for half of the site because of the well position along the upland-marsh bank. The middle well location also limited the availability of measurable specimens due to its location in a sparse depression. It is apparent from these 200 sq. meter quadrants, which well sites have the greatest tree density immediately around the well site.

**CONCLUSION**

It was proposed that (A) groundwater level dynamics would mimic topography and salinity would decrease with increasing distance from Big Bay Creek; and (B) upland groundwater patterns would mimic evapotranspiration while the marsh groundwater patterns would reflect a tidal influence. The results of this study showed that other types of groundwater dynamics occur and are likely due to differences in environmental and topographic conditions across marsh-upland ecosystems. For example, groundwater patterns at the middle well (evapotranspiration dominance) and marsh well locations (tidal dominance) supported the hypothesis. However, the groundwater level at the south well was mainly influenced by tidal forcing patterns and not evapotranspiration patterns, despite the well being located at the highest elevation. This is likely due to its close proximity to the cut bank of Big Bay Creek. Therefore, proximity of the uplands to a tidal water body was shown to affect groundwater patterns more than elevation. The hypothesis that the upland groundwater will show a dominant evapotranspiration pattern did not stand regarding the south well. Alternatively, salinity levels at the north and south wells were related to the proximity of Big Bay Creek where groundwater was characterized as fresh, and at the marsh wells where groundwater was saline.

Additional evidence of alternative groundwater conditions showed that at the middle well, the highest salinity reading was recorded for the upland wells despite it being located furthest away from Big Bay Creek. The topography at the middle well may explain the uncharacteristic groundwater and salinity readings at this site. This well is located in a lower elevation slough which extends to the creek, and perhaps allows for surface water to enter into the slough. However, it was further questioned whether contamination affecting the salinity readings at the middle well occurred from the bentonite seal installation. The bentonite seal was applied around the same intersection of the middle well as the mean groundwater level. Previous studies found that contamination of groundwater from bentonite seals occur with a peak in contamination over the first 100 to 500 days of installation, as witnessed in the middle well hydrograph (Remenda and Kamp, 1997). Future research at this site may confirm this assumption through the installation and monitoring of a well at the slough-creek outlet. Beyond those findings, the hypothesis that groundwater would mimic topography was supported by the groundwater elevation graphs showing that the highest elevation locations also had the highest water table elevations AMSL.

The results of this study can be expanded to determine how sea level rise may affect the tidal salt marsh and upland habitats. In general, the lower elevation locations and those adjacent to the cut bank are at the greatest risk for future sea level rise. This can be seen in the northern high marsh (T5 well) where saltwater flooding events are already occurring (Figures 10 and 15). Despite these saltwater flooding events, the northern marsh acts as a net freshwater discharge area as evidenced by seasonal salinity variations at the T5 well which show lower salinity levels in the wet months (spring and summer) and higher salinity levels in the dry months (fall and winter). If saltwater intrusion continues into the upland north well, the amount of freshwater discharging would be diminished and could upset current marsh ecology.

Topographic variations at the site, as illuminated by the Lidar DEM (Figure 20), also indicate areas at risk for sea level rise. The topographic slope between the marsh
and uplands determines marsh sediment accumulation and therefore the marsh’s ability to retreat into the uplands in response to sea level rise. At this site, the topographic slope is gradual at the northern side and steep on the southern side. Therefore, despite current flooding occurring along the north MTU, the ability for the marsh sediment to accumulate and expand into the uplands is greater on the northern end of Edisto Beach State Park. Furthermore, dense Spartina alterniflora communities along the northern marsh will assist in sediment accumulation. Sediment accumulation at a rate greater than sea level rise will allow for the success of the marsh by retreating into the marsh-upland border.

MTU – upland areas with steeper slopes, such as the southern marsh, are at risk because areas of the MTU that are rarely flooded have slower vertical accretion rates since sediment is not constantly being deposited and settled out at the same rate as the lower marsh (Kirwan and Megonigal, 2013). Therefore, due to the higher elevation and infrequent flooding events, sediment may not accumulate at a rate that can keep up with sea level rise. In addition, the steep topographic gradient between the marsh and uplands at this site may make it difficult for the marsh to retreat into the uplands. This southern site is also at risk for saltwater intrusion as evidenced by the tidal signal apparent in the south upland groundwater hydrograph (Figure 3). This signal is believed to be a result of tidal forcing from Big Bay Creek. The geomorphology of the creek in the presence of the cut bank adjacent to the south end of the site allowed for propagation of tidal energy into the shallow freshwater aquifer. Therefore, the southern side of the marsh is clearly at risk for saltwater intrusion. This phenomenon is illustrated by the model of Schultz and Ruppel (2001) shown in Figure 21 in which the tidal signal loses amplitude as it migrates through the sediment further away from the creek. Saltwater intrusion from Big Bay Creek may also be occurring at the middle well although it is located furthest away from the creek. The middle well recorded high salinity levels and is adjacent to an ephemeral stream perhaps allowing saltwater from Big Bay Creek to enter into the uplands from this topographic low.

However, another groundwater input process may be simultaneously occurring as well. Groundwater from the uplands is likely flowing horizontally into the depression and recharging the middle well due to the decrease in elevation surrounding the middle well. This process was seen in the water budget following a rain event in mid-August when the groundwater of the north and south wells showed a water surplus at the end of the month and the middle well remained around the antecedent water level indicating discharge over the month (Figure 18). The location of the middle well as a discharge area and the north and south wells as recharge areas may explain the differences in water storage among the upland wells.

Additional groundwater trends that were revealed through the water budget analysis showed that the north and south upland wells did not differ much despite their distance. This may be due to their similar topographic and groundwater levels. The marsh wells generally followed similar water storage patterns although the T5 location had slightly greater water storage change. This may indicate that

![Figure 21. Diagram showing tidal flow pattern from the creek into the water table. The diagram shows the tidal amplitude lessening as it flows through the sediment (Schultz and Ruppel, 2001).](image)

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