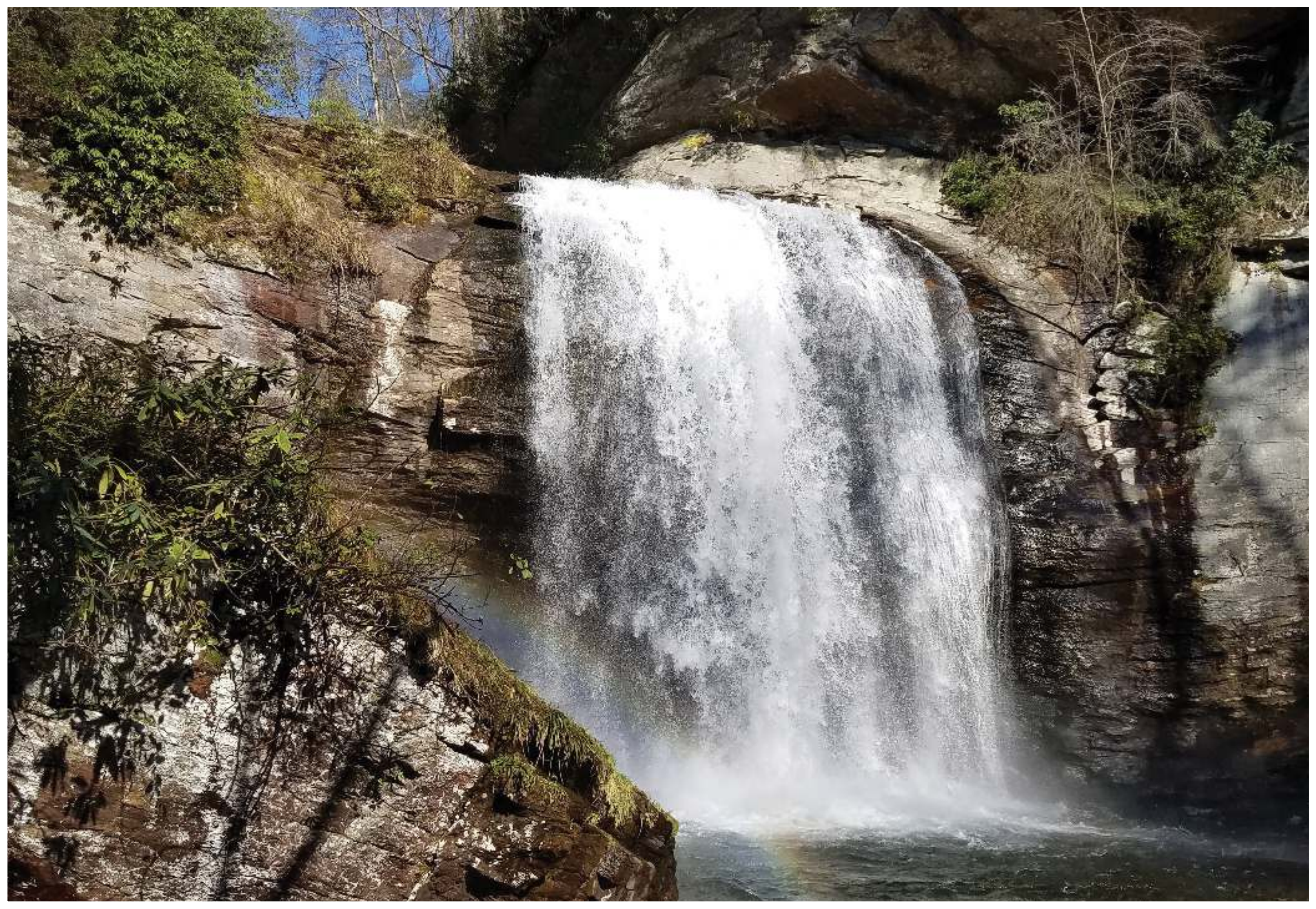
Journal of Contemporary

Water Research & Education

Issue 166
April 2019



UCOWR
UNIVERSITIES COUNCIL ON WATER RESOURCES

A publication of the Universities Council on Water Resources with support from Southern Illinois University Carbondale

JOURNAL OF CONTEMPORARY WATER RESEARCH & EDUCATION

Universities Council on Water Resources
1231 Lincoln Drive, Mail Code 4526
Southern Illinois University
Carbondale, IL 62901
Telephone: (618) 536-7571
www.ucowr.org

ISSN 1936-7031

Cover photo: *Looking Glass Falls, Pisgah National Forest, NC*, Credit: Jackie Crim
Back cover photo: *Summertime in Snowbird,* Credit: Snowbird
Inside back cover photo: *Aerial Tram*, Credit: Snowbird

Subscription Information: The *Journal of Contemporary Water Research & Education* is published by the Universities Council on Water Resources. The annual subscription rate is $60 (domestic) and $100 (international). Prices per copy for past issues are $20 (domestic) and $40 (international). Members of UCOWR receive access to the *Journal of Contemporary Water Research & Education* as a part of their membership.

Printed by the authority of the State of Illinois. April 2019. 400 copies. Order Number XXXXX.

Journal of Contemporary Water Research & Education

Issue No. 166 April 2019

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 1-11, April 2019

A Survey of Public Perceptions and Attitudes about Water Availability Following Exceptional Drought in Texas

*Drew M. Gholson[1], Diane E. Boellstorff[2], Scott R. Cummings[3], Kevin L. Wagner[4], and Monty C. Dozier[2]

*[1]Mississippi River Valley Alluvial Aquifer Water Center, Delta Research and Extension Center, Mississippi State University, Stoneville, MS; [2]Texas A&M AgriLife Extension Service, Dept. of Soil and Crop Sciences, Texas A&M University System, College Station, TX; [3]Texas A&M AgriLife Extension Service, Dept. of Agricultural Leadership, Education, and Communications, Texas A&M University System, College Station, TX; [4]Oklahoma Water Resources Center, Oklahoma State University, Stillwater, OK; *Corresponding Author*

Abstract: This study examines the results of a random sample survey of Texans evaluating citizen awareness, attitudes, and willingness to adopt water conservation practices. The study investigates changes in public attitudes following the most intense one-year drought on record in Texas by evaluating public perception of water availability, assessing Texans' attitudes and perceptions regarding drought conditions, and comparing the number of Texans adopting practices to conserve water before and after the drought of 2011. Almost 70% indicated that the likelihood of their area suffering from a prolonged drought was increasing. More than 61% of respondents have changed the way their yard is landscaped and 62% have also adopted new technologies in an effort to conserve water. Overall, responses indicated that Texans are concerned with water availability after experiencing, in 2011, the worst one-year drought on record, and that the majority of respondents are taking personal action in an effort to conserve water for the future.

Keywords: *survey, perception, water conservation, drought, attitudes*

Texas experienced its worst single-year drought on record in 2011 (Nielsen-Gammon 2012), affecting people in many ways. While farmers may have been more directly affected by drought, city dwellers also were impacted by expectations for compliance with municipal drought contingency plans and water restrictions. For some citizens, public supplies came within days of running out of water and a few systems were supplied by neighboring utilities. Reservoir levels dropped and reached record lows for storage, while aquifer levels also dropped and some wells went dry. The 2011 drought caused a record loss of $7.62 billion to Texas agriculture (Fannin 2012). Most water supply systems implemented mandatory water restrictions. The severity of the drought captured the attention of Texans from all regions of the state.

In addition to the pressures of periodic, extreme drought, the Texas Water Development Board (2017) estimates that the Texas population will increase more than 70% from 2020 to 2070, and water demand will increase by 17%. Texas' rapidly growing urban areas will lead water consumption for the state. By 2070, 30% of the total water volume included in management strategies proposed in the State Water Plan will involve demand management to reduce needs for additional water through water conservation and drought management (Texas Water Development Board 2017).

Public perceptions and attitudes toward water issues will play an important role in whether Texans choose to adopt water conservation practices. Water conservation by Texas residents will play a pivotal role in meeting water supply demands the state will face in the future. Previous research links attitudes and perceptions to water use behaviors (Campbell et al. 2004; Clarke and Brown 2006; Jorgensen et al. 2009; Willis et al. 2011). The public's attitudes regarding water supply also can be linked to experiences in longer-term drought conditions (Delorme et al. 2003; Casagrande et al. 2007; Adams et al. 2013; Evans et al. 2015).

Texas A&M AgriLife Extension Service, in conjunction with a national needs assessment project initiated through the Pacific Northwest Regional Water Program, facilitated two mailed random sample surveys of Texans to evaluate citizen awareness, attitudes, and willingness to act on water issues (Mahler et al. 2013). The first survey was conducted in 2008 at the beginning of a relatively mild drought. The drought intensified through 2009-2012 when much of the state was categorized as enduring exceptional drought. The original survey was re-issued to another random sample of Texans in 2014, resulting in an opportunity to investigate changes in public attitudes following exposure to one of the most intense one-year droughts in Texas. The objectives of this study are to: 1) evaluate the public's perception of water availability, 2) evaluate Texans' attitudes and perceptions regarding drought conditions, and 3) compare the frequency of Texans adopting practices to conserve water before and after the drought of 2011.

Materials and Methods

A state-wide survey was developed to assess Texans' perceptions and attitudes about water resources within the state. The questionnaire is one of the survey components comprising the National Integrated Water Quality Program Needs Assessment Survey project initiated in 2002. The present survey is based on the 2002 template developed by water quality coordinators in the Pacific Northwest region, with input from other participating Land Grant Institution (LGI) water quality coordinators for the Southern, Mid-Atlantic, Northwest, Northeast, and Caribbean Island Regional Water Programs (Mahler 2010). The survey was mailed to 1,275 randomly selected Texas residents in August 2008 following methods described in Boellstorff et al. (2010); 419 surveys (33%) were completed and returned. Minor modifications were made to the template survey to adapt it to Texas' water management agencies and organizations, and to modernize particular questions before the survey was re-issued in 2014. The survey questionnaire included 59 questions addressing water resources, water quality, and other environmental issues. The study population consisted of the adult residents of Texas.

In April 2014, the questionnaire was sent via direct mail survey to 1,800 randomly selected residences in Texas following the tailored survey design method of Dillman (2000), and as the Texas population had increased, recalculating the number of mail outs necessary as described in Boellstorff et al. (2010). As in the 2008 survey, randomly selected addresses were purchased from Survey Sampling International, Fairfield, CT and, individuals were mailed a paper copy of the survey instrument, a cover letter, and a self-addressed, stamped envelope. Twenty days later, individuals were sent a reminder postcard. Twenty days after the reminder postcard was sent, another survey instrument, cover letter, and self-addressed, stamped envelope were mailed. Twenty days later, a final reminder postcard was mailed to participants.

Individuals returning the survey or indicating that they did not want to participate in the study were removed from the mailing list so that they were not re-contacted. Taking into account the number of 1) surveys "returned to sender for incorrect address," 2) recipients requesting to not participate, and 3) recipient death, the effective number of mailed questionnaires in 2014 was 1,655 and the return rate for the completed survey questionnaires was 29%. Survey responses were coded and entered into a spreadsheet. Missing data were excluded from analyses.

This study investigates the relationship of water quantity perceptions to water conservation actions. Responses to the following five questions in Table 1 for both 2008 and 2014 along with socio-demographic information requested by the survey are the focus of this article.

Additionally, this study assesses the change in public attitudes and perceptions regarding water resources and actions taken to conserve water, using data from surveys administered in 2008 and 2014, and examines the change in rate of adoption of water saving practices regarding survey year and associated socio-demographics.

The Statistical Package for Social Sciences (SPSS) Version 23 was used for data analyses.

Table 1. Question wording and response set.

Question	Response Set
1) Do you regard water quantity (having enough water) as a problem in the area where you live? (Mark one answer)	a. Definitely not b. Probably not c. I don't know d. Probably e. Definitely yes
2) The likelihood of your area suffering from a prolonged drought is:	a. Increasing b. Decreasing c. Staying the same d. No opinion
3) The likelihood of your area having enough water resources to meet all of its needs 10 years from now is:	a. High (likely enough) b. Medium c. Low (likely not enough water) d. No opinion
4) Have you or someone in your household done any of the following as part of an individual or community effort to conserve water or preserve water quality? (Mark all that apply)	a. Changed the way your yard was landscaped b. Changed how often you water your yard c. Changed use of pesticides, fertilizers, other chemicals d. Pumped your septic system (if you have one) e. Adopted new technologies (low flow showerheads, high-efficiency washing machines and dishwashers, etc.)
5) Do you think that the amount of rainfall in your area will change as a result of global warming?	a. Yes, a significant increase in rainfall b. Yes, a slight increase in rainfall c. No, no change in rainfall d. Yes, a slight decrease in rainfall e. Yes, a significant decrease in rainfall d. I don't know

Descriptive summary statistics were calculated for socio-demographic variables (Table 2). The null hypothesis that the response frequencies are the same for the various answer options and socio-demographic variables was tested using Pearson's chi-squared and logistic regression analyses. A multinomial logistic regression analysis was used to predict the likelihood of adopting water conserving actions such as: changing yard landscaping, changing lawn watering, and adopting water conserving technologies, based on socio-demographics and responses from 2008 and 2014 surveys.

Further, the potential differences in the influence of water availability perception on water management behaviors before the exceptional drought (2008 survey) and responses after the exceptional drought (2014 survey) were evaluated. Pearson's chi-squared test ($p<0.05$) was applied to determine significant differences in responses before or after the 2011 Texas drought and for demographic variables.

Table 2. Demographics of respondents for surveys conducted in 2008 and 2014.

Category		Year 2008 % (n)	Year 2014 % (n)
Gender	Male	63.9 (262)	48.7 (185)
	Female	36.1 (148)	51.3 (195)
Years lived in Texas	All my life	47.9 (197)	46.6 (180)
	More than 10 years	40.6 (167)	45.6 (176)
	5 to 9 years	7.1 (29)	4.4 (17)
	Less than 5 years	4.4 (18)	3.4 (13)
Size of residence community	> 100,000	48.1 (190)	53.5 (238)
	25,000 to 100,000	21.3 (84)	19.6 (87)
	7,000 to 25,000	12.2 (48)	11.2 (50)
	3,500 to 7,000	8.6 (34)	5.8 (26)
	<3,500	9.9 (39)	9.9 (44)
Education	Less than or some high school	5.4 (22)	3.5 (16)
	High school graduate	16.4 (67)	12.6 (58)
	Some college	31.5 (129)	27.9 (129)
	College graduate	25.4 (104)	33.5 (155)
	Advanced college degree	21.3 (87)	22.5 (104)
Age	18 - 24	1.2 (5)	0.5 (2)
	25 - 34	6.9 (29)	4.2 (16)
	35 - 49	25.3 (106)	18.9 (72)
	50 - 64	28.4 (119)	40.8 (155)
	65 years old or older	38.2 (160)	35.5 (135)
Residence location	Inside city limits	73.5 (302)	72.8 (337)
	Outside city limits, not farming	22.6 (93)	22.7 (105)
	Outside city limits, farming	3.9 (16)	4.5 (21)

Results

The 2014 water issues survey achieved a response rate of 29.4% (491 out of 1,671 surveys) with 327 respondents coming from the first mailing, and 164 from the second mailing. Demographic characteristics regarding residence for 2008 and 2014 were not significantly different. As shown in Table 2, 48.1 and 53.5% of survey respondents lived in communities of more than 100,000 in 2008 and 2014, respectively. In addition, 73.5% of survey respondents in 2008 and 72.8% in 2014 lived inside city limits. A total of 71% of respondents from both surveys resided in communities of 25,000 or more people. Twenty-nine percent lived in small communities of 7,000 people or fewer. These demographic results are similar to those reported by the 2010 U.S. Census effort, which indicated that 84.7% of Texans reside in urban areas (U.S. Census Bureau 2010). A large majority, more than

90%, of respondents for both surveys had lived in Texas for more than 10 years or for all their lives.

Respondent gender distribution differed between the 2008 and 2014 surveys; with 2014 more closely reflecting the actual demographics of the state: 48.7% male and 51.3% female ($p<0.0001$). Respondents of both surveys were somewhat better educated and older than the general Texas population (U.S. Census Bureau 2013, 2015).

Water Quantity

Respondents were asked, "Do you regard water quantity (having enough water) as a problem in the area where you live? (Mark one answer)." From the response set, respondents could choose: definitely not, probably not, I don't know, probably, or definitely yes. In 2008, 22.5% of respondents believed water quantity to be a problem where they lived (Figure 1) and 47.9% believed that water quantity definitely or probably was a problem in their area. In comparison, 37.2% from the 2014 survey responded that water quantity is a problem where they live (likelihood ratio test, $p<0.0001$), and a sum of 61.6% believed water quantity definitely or probably was a problem in their area. Furthermore in 2008, 15.1% of the respondents agreed that water quantity was definitely not a problem where they lived, while only 6.8% agreed water quantity was definitely not a problem in the 2014 survey ($p<0.0001$). A combined 44.2% of respondents indicated that there was definitely not or probably not a water quantity problem in their area, and that fell to 28.2% in 2014. Multinomial logistic regression analysis of responses from the 2014 survey indicated no statistical significance with socio-demographic variables of gender, community size, age, residence location, years in Texas, and education.

Likelihood of Prolonged Drought

Similar responses to the water quantity question were given when survey respondents were asked to evaluate the likelihood of their area suffering from a prolonged drought. In 2008, 51.6% of respondents believed that the chance of a prolonged drought in their area was increasing, while in 2014, 69.2% responded that the chances of a prolonged drought in their area was increasing ($p<0.0001$). The number of Texans responding that the likelihood of a prolonged drought in their area staying the same decreased from 37.9% in 2008 to 22.1% in 2014 ($p<0.05$; Table 3). Fewer responses in the "staying the same" category were likely the result of about 40% of Texas experiencing some level of drought in August 2008, while about 66% of Texas was in a drought in April 2014 when the survey was re-issued. In April of 2014, more than 16 million Texans lived in areas categorized as in moderate or more extreme categories of drought (U.S. Drought Monitor Map Archive, Fuchs 2014). Multinomial

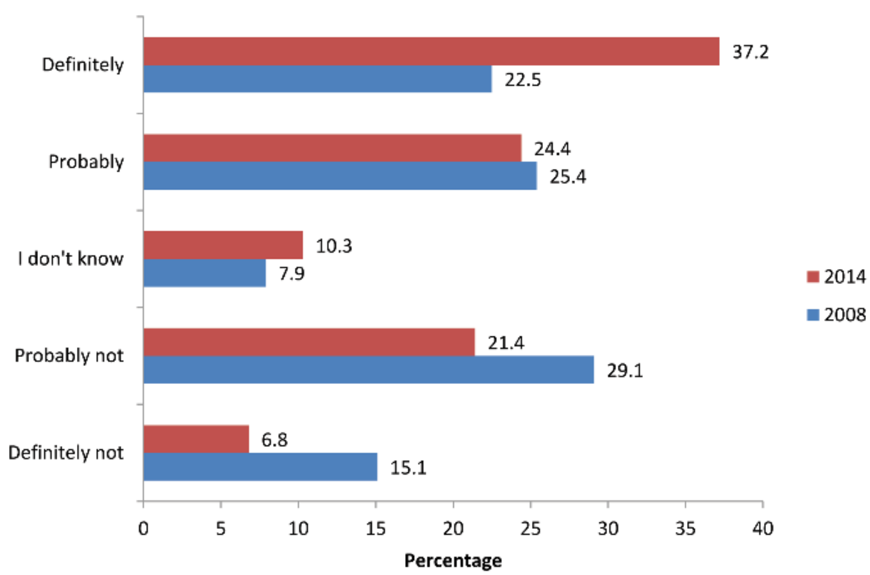


Figure 1. Is water quantity a problem where you live?

Table 3. The likelihood of your area suffering from a prolonged drought is:

		-------% Respondents-------		Percentage Point Change
		2008	2014	
Prolonged drought affecting your area	Increasing	51.6[a]	69.2[b]	17.6
	Staying the same	37.9[a]	22.1[b]	-15.8
	Decreasing	2.4[a]	2.1[a]	-0.3
	No opinion	8.1[a]	6.6[a]	-1.5

Superscript indicates significance at the 0.05 level.

logistic regression analysis of responses from the 2014 survey indicated no statistical significance of the response to the likelihood of a prolonged drought with socio-demographic variables of gender, community size, age, residence location, years in Texas, and education.

Likelihood of Enough Water to Meet Area Needs

Respondents were asked to evaluate the likelihood of their area having enough water to meet its needs 10 years from now. In 2008, 30.2% of the survey respondents believed that there would not be enough water in their area to meet all of its needs in 10 years (Figure 2). In 2014, the responses for low likelihood (likely not enough water) increased to 52.8% ($p<0.0001$). Additionally, 20.0% of survey respondents in 2008 replied that the likelihood of enough water in their area was high (likely enough water) to meet needs in 10 years, compared to only 7.1% in 2014. Multinomial regression analysis of the responses for the 2014 survey indicated respondents having more education ($p<0.001$) were more likely to believe that there would not be enough water in their area to meet needs in 10 years. Other socio-demographic variables showed no significant relationships.

Behavior Changes Protecting Water Quality or Water Quantity

Landscaping. As shown in Figure 3, respondents from the 2014 survey were more likely to have changed the way they landscaped their yards than 2008 survey respondents ($p<0.001$). Multinomial logistic regression analyses of the 2014 responses with socio-demographic variables indicated gender differences were significant ($p<0.05$). Female respondents were more likely than males to have changed the way they landscape their yard.

Watering. Surprisingly, there was no significant difference between 2008 and 2014 respondents regarding whether homeowners had changed how often they watered their yards, perhaps because municipal drought restrictions had already been commonly imposed during the drought in 2008 (chi-square). For 2014, gender ($p<0.05$) and number of years lived in Texas ($p<0.05$) were significant regarding whether respondents had changed how often they watered their yard. Females and respondents living in Texas longer were more likely to have changed the way they watered their yard.

Adopt New Technologies. Respondents in 2014 were more likely than those in 2008 to have adopted new technologies to conserve water quantity or quality (chi-square, $p=0.001$). Again, gender was the only significant predictor for adopting new technologies in an effort to conserve water (multinomial logistic regression, $p<0.006$). Females were more likely to adopt new technologies in an effort to conserve water than were males.

Rainfall Change as a Result of Global Warming

Responses to the question, “Do you think that the amount of rainfall in your area will change as a result of global warming?” differed significantly between survey years (chi-square, $p<0.001$). From the 2008 to the 2014 survey, an increased percentage of respondents (+12.4%) believed that rainfall would decrease significantly (Table 4); however, approximately one-third of respondents for both the 2008 and 2014 surveys answered that

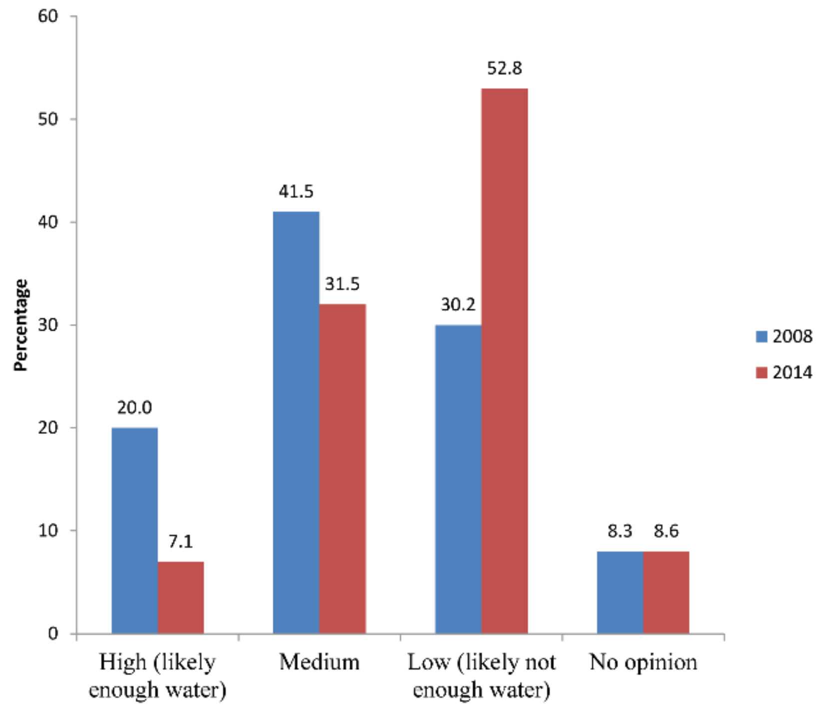


Figure 2. The likelihood of your area having enough water resources to meet all of its needs 10 years from now is:

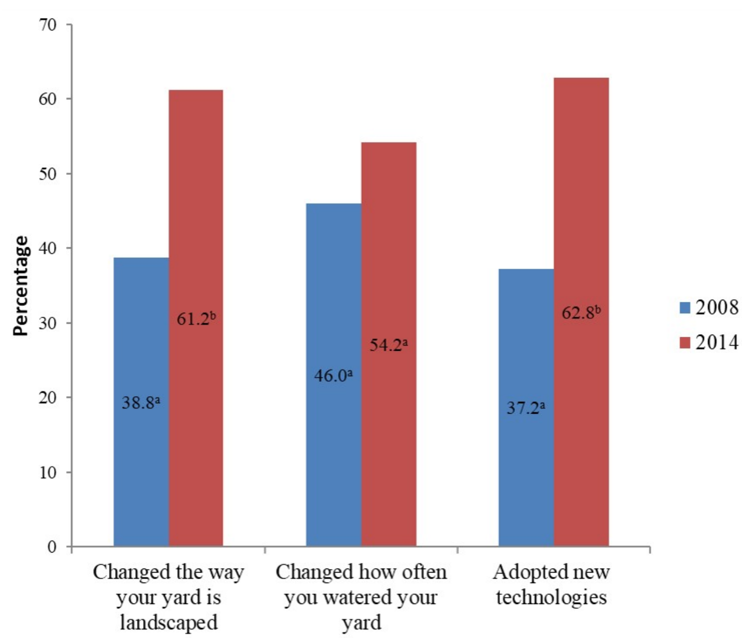


Figure 3. Have you or someone in your household done any of the following as part of an individual or community effort to conserve water or preserve water quality? Different letters indicate a significant difference at the 0.05 level.

they do not know if the amount of rainfall in their area will change.

Multinomial logistic regression of socio-demographic variables indicated that education plays a role in the perception of rainfall changes that might occur as a result of global warming (p=0.001). Those with more education were less likely to respond that rainfall will increase as a result of global warming (R2=0.06).

Discussion

Using data from surveys administered in 2008 and 2014, this study assesses the change in public attitudes and perceptions regarding water resources and actions taken to conserve water. The questionnaire is a component of the National Integrated Water Quality Program Needs Assessment Survey project initiated in 2002 (Mahler et al. 2005). The focus of this study was on the year of the survey (before or after a historical drought) and responses to questions related to current water availability issues and Texans' perceptions of future water availability. Additionally, change in rate of adoption of water saving practices was assessed regarding survey year and associated socio-demographics. The results of this study indicate that recent drought experience strongly influences public perception of current water quantity issues as well as perception of future water availability. Evans et al. (2015) similarly reported that perceptions of local drought conditions significantly affected public attitudes and awareness regarding water supply. Specifically, the public is more concerned about water resources and climate change during periods of extreme drought. Evans et al. (2015) also showed that length of residency significantly affected the perception of water availability, with respondents living in the state longer being less likely to be concerned with water supply. With the exception of how often respondents watered their yard, length of residency was not a statistically significant predictor in their adoption of water conservation practices in the present study, perhaps because the drought was exceptional in intensity and duration. Additionally, few respondents had lived in Texas for less than 10 years. News coverage of drought typically increases when drought intensifies, enhancing the awareness of extreme drought (Dow 2010).

As shown in Figure 1 and Table 3, perception of future water availability shifted significantly following the period of extended exceptional drought, at its worst in 2011, with 2014 respondents indicating more concern than did 2008 respondents. Texans have become more concerned with having enough water within 10 years to meet their needs, with 53% believing supply will not be adequate.

Table 4. Do you think that the amount of rainfall in your area will change as a result of global warming?

		----------- Year----------- 2008 % (n)	2014 % (n)	Percentage Point Change
Do you think that the amount of rainfall in your area will change as a result of global warming?	Yes, increase significantly	6.0 (24)	2.7 (12)	-3.3
	Yes, increase slightly	7.2 (29)	2.9 (13)	-4.3
	No change	26.3 (106)	17.8 (80)	-8.5
	Yes, decrease slightly	17.1 (69)	17.3 (78)	0.2
	Yes, decrease significantly	13.2 (53)	25.6 (115)	12.4
	I don't know	30.3 (122)	33.8 (152)	3.5

Almost 70% felt that the likelihood of their area suffering from a prolonged drought was increasing. More than 61% of respondents have changed the way their yard is landscaped in efforts to conserve water. Furthermore, more than 62% have also adopted new technologies in an effort to conserve water.

Perceived importance of water resources is a significant factor that drives water conservation (Adams et al. 2013). Efforts initiated during drought periods to conserve water by changing the way a yard is landscaped or adopting new technologies (low flow showerheads, high efficiency appliances, etc.), can become long-term behavior changes. Adoptions of more permanent changes, rather than temporary or short-lived actions, represent positive behavior modification likely to continue even during normal rainfall periods. Additionally, intensifying public concern regarding water supplies during drought conditions creates unique opportunities for Extension and other water resource management organizations to deliver timely and valued water conservation information.

Perception that the amount of rainfall in their area will change as a result of global warming increased from 2008 to 2014 with a jump (+12.4%) in respondents believing rainfall will significantly decrease. However, despite frequent media reports regarding climate change, respondents indicating that they did not know what rainfall changes would occur increased slightly from 30.3 to 33.8%. Udayakumara et al. (2010) reported that environmental awareness is influenced by education. Similarly, the present study found that increased education influenced perception that rainfall would decrease as a result of global warming. Kleinberg and Colby (2014) and Leiserowitz (2005) reported that some citizens believe that climate change will not affect them as individuals or as communities, but is rather more a global or national problem. The findings of these studies may support the contention that further climate change research and/or outreach education is necessary before more of the public feel they can draw an informed conclusion.

Conclusion

Overall, responses indicate that Texans are concerned with water availability and believe that there are concerns for water resources in the future after experiencing, in 2011, the worst one-year drought on record. Results also indicate that with citizen concern, the majority of respondents are taking personal action in an effort to conserve water for the future.

This study provides useful information in support of water conservation outreach programs. Texans tend to be more concerned with water availability during and after drought, providing a timely opportunity to highlight drought conditions and teach appropriate responses and actions for citizens through outlets such as state agencies, Extension services, news outlets, and groundwater and utility districts. It may also be effective to remind the public of the extreme droughts they have experienced when conducting an outreach program. As this study indicates, Texans are more willing to make changes to their landscape during and after droughts. Outreach programs with information including best management practices for lawn irrigation, drought tolerant landscapes, and new water conservation technologies should be made available through appropriate sources. The study further supports the idea that investment in education during critical environmental events, such as drought, when audiences are seeking information is especially effective. Cohen et al. (2006) suggested that adoption during extreme events frequently results in permanent behavior changes that continue to conserve water resources when more typical weather returns.

Regional and state-wide surveys are important tools for assessing public perception and attitudes regarding water availability issues. Survey evaluations can document changes in perception and adoption of best management practices, as well as identify opportunities for expanded outreach and research efforts.

Acknowledgements

This work was made possible in part by the United States Department of Agriculture National Institute of Food and Agriculture under Grant Number 2008-51130-19537, also through a Clean Water Act §319(h) Nonpoint Source funding from Texas State Soil and Water Conservation Board and the United States Environmental Protection Agency under Grant Numbers 10-04 and 13-08.

Author Bio and Contact Information

Dr. Drew M. Gholson (corresponding author) is an Assistant Extension/Research Professor with Mississippi State University specializing in water conservation and irrigation with the Mississippi River Valley Alluvial Aquifer Water Research Center located at the Delta Research and Extension Center in Stoneville, Mississippi. Outreach and research includes developing and delivering educational programs involving supporting Mississippi growers with their irrigation needs, improving irrigation application efficiency, adoption of the best management practices, and promote conservation. He can be contacted at drew.gholson@msstate.edu or via mail at P.O. Box 197, 82 Stoneville Road, Stoneville, MS 38776.

Dr. Diane Boellstorff serves as associate professor in the Department of Soil and Crop Sciences and Texas A&M AgriLife Extension water resource specialist. Her primary areas of focus are developing educational and applied research programs involving water resource management, including groundwater and private water well protection, water conservation, water quality, and watershed protection planning. Outreach includes developing and delivering educational programs involving supporting source water protection for residents using water wells, residential water conservation, and facilitation of community-driven watershed planning efforts. She can be reached at dboellstorff@tamu.edu or via mail at 2474 TAMUS, College Station, TX 77843.

Dr. Scott Cummings currently serves as the program leader for the Organizational Development Unit of the Texas A&M AgriLife Extension Service. His areas of work include development, evaluation, and accountability of community education programs. He can be reached at s-cummings@tamu.edu or via mail at 128C AGLS, 2116 TAMU, College Station, TX 77843-2116.

Dr. Kevin Wagner is the Director of the Oklahoma Water Resources Center and holds the Thomas E. Berry Professorship in Integrated Water Research and Management at Oklahoma State University. He provides leadership and administration of the Water Center's programs, leads university efforts to increase engagement with the water resources community across Oklahoma and the nation, and facilitates development of inter-disciplinary teams to address high priority water resources issues in the region. Dr. Wagner has 23 years of experience in watershed management, stakeholder engagement, and conservation practice research and education. He may be contacted at kevin.wagner@okstate.edu.

Dr. Monty Dozier holds a PhD in Agronomy from Texas A&M University. He has been employed by Texas A&M AgriLife Extension for 33 years serving in county, state, and leadership positions including currently with the Governor's Commission to Rebuild Texas. Academically, Dr. Dozier has authored or co-authored nine journal articles, 39 Extension publications, 53 abstracts, and secured over $3.3 million in grants. He has served on 17 graduate student committees and with 11 international Extension programming efforts. He can be contacted at mdozier@ag.tamu.edu or via mail at 2150 TAMUS, Bryan, TX 77807.

References

Adams, D.C., D. Allen, T. Borisova, D.E. Boellstorff, M.D. Smolen, and R.L. Mahler. 2013. The influence of water attitudes, perceptions, and learning preferences on water-conserving actions. *Natural Sciences Education* 42: 114-122. doi:10.4195/nse.2012.0027. Accessed January 31, 2019.

Boellstorff, D.E., M.L. McFarland, and C.T. Boleman. 2010. Water Issues in Texas: A Survey of Public Perceptions and Attitudes about Water. Available at: https://www.agrilifebookstore.org/A-Survey-of-Public-Perceptions-and-Attitudes-p/b-6219.htm. Accessed January 31, 2019.

Campbell, H.E., R.M. Johnson, and E.H. Larson. 2004. Prices, devices, people, or rules: The relative effectiveness of policy instruments in water conservation1. *Review of Policy Research* 21(5): 637-662. doi:10.1111/j.1541-1338.2004.00099.x. Accessed January 31, 2019.

Casagrande, D., D. Hope, E. Farley-Metzger, W. Cook, S. Yabiku, and C. Redman. 2007. Problem and oppportunity: Integrating anthropology, ecology, and policy through adaptive experimentation in the urban U.S. Southwest. *Human Organization* 66(2): 125-139. doi:10.17730/humo.66.2.h5277q2u27354358. Accessed January 31, 2019.

Clarke, J.M. and R.R. Brown. 2006. Understanding the factors that influence domestic water consumption within Melbourne. *Australian Journal of Water Resources* 10(3): 261-268. doi:10.1080/13241583.2006.11465301. Accessed January 31, 2019.

Cohen, S., D. Neilsen, S. Smith, T. Neale, B. Taylor, M. Barton, W. Merritt, Y. Alila, P. Shepherd, R. Mcneill, J. Tansey, J. Carmichael, and S. Langsdale. 2006. Learning with local help: Expanding the dialogue on climate change and water management in the Okanagan Region, British Columbia, Canada. *Climatic Change* 75(3): 331-358. doi:10.1007/s10584-006-6336-6. Accessed January 31, 2019.

Delorme, D.E., S.C. Hagen, and I.J. Stout. 2003. Consumers' perspectives on water issues: Directions for educational campaigns. *The Journal of Environmental Education* 34(2): 28-35. doi:10.1080/00958960309603497. Accessed January 31, 2019.

Dillman, D.A. 2000. *Mail and Internet Surveys: The Tailored Design Method*. John Wiley & Sons, Inc., New York, NY.

Dow, K. 2010. News coverage of drought impacts and vulnerability in the US Carolinas, 1998–2007. *Natural Hazards* 54(2): 497-518. doi:10.1007/s11069-009-9482-0. Accessed January 31, 2019.

Evans, J.M., J. Calabria, T. Borisova, D.E. Boellstorf, N. Sochacka, M.D. Smolen, R.L. Mahler, and L.M. Risse. 2015. Effects of local drought condition on public opinions about water supply and future climate change. *Climatic Change* 132(2): 193-207. doi:10.1007/s10584-015-1425-z. Accessed January 31, 2019.

Fannin, B. 2012. Updated 2011 Texas Agricultural Drought Losses Total $7.62 Billion. Available at: http://today.agrilife.org/2012/03/21/updated-2011-texas-agricultural-drought-losses-total-7-62-billion/. Accessed January 31, 2019.

Fuchs, B. 2014. U.S. Drought Monitor: Texas. Map Archive. Available at: https://droughtmonitor.unl.edu/Maps/MapArchive.aspx. Accessed January 31, 2019.

Jorgensen, B., M. Graymore, and K. O'Toole. 2009. Household water use behavior: An integrated model. *Journal of Environmental Management* 91(1): 227-236. Available at: http://dx.doi.org/10.1016/j.jenvman.2009.08.009. Accessed January 31, 2019.

Kleinberg, C.T. and C. Colby. 2014. Perceptions of climate change. *Journal of Environmental and Resource Economics at Colby* 01(01): 1-20. Available at: http://digitalcommons.colby.edu/jerec/vol01/iss01/4. Accessed January 31, 2019.

Leiserowitz, A.A. 2005. American risk perceptions: Is climate change dangerous? *Risk Analysis* 25(6): 1433-1442. doi:10.1111/j.1540-6261.2005.00690.x. Accessed January 31, 2019.

Mahler, R.L. 2010. Information sources, learning opportunities, and priority water issues in the Pacific Northwest. *Journal of Extension* 48(2): 1-9. Available at: https://joe.org/joe/2010april/rb2.php. Accessed January 31, 2019.

Mahler, R.L., R. Simmons, and F. Sorensen. 2005. Public perceptions and actions towards sustainable groundwater management in the Pacific Northwest Region, USA. *International Journal of Water Resources Development* 21(3): 465-472. doi:10.1080/07900620500036604. Accessed January 31, 2019.

Mahler, R.L., M.D. Smolen, T. Borisova, D.E. Boellstorff, D.C. Adams, and N.W. Sochacka. 2013. The National Water Survey Needs Assessment Program. *Natural Sciences Education* 42(1): 98-103. doi:10.4195/nse.2012.0025. Accessed January 31, 2019.

Nielsen-Gammon, J.W. 2012. The 2011 Texas drought. *Texas Water Journal* 3(1): 59-95. Available at: https://journals.tdl.org/twj/index.php/twj/article/view/6463. Accessed January 31, 2019.

Texas Water Development Board. 2017. 2017 State Water Plan. Water for Texas. Available at: http://www.twdb.texas.gov/waterplanning/swp/2017/doc/2017_SWP_Adopted.pdf. Accessed January 31, 2019.

Udayakumara, E.P.N., R.P. Shrestha, L. Samarakoon, and D. Schmidt-Vogt. 2010. People's perception and socioeconomic determinants of soil erosion: A case study of Samanalawewa watershed, Sri Lanka. *International Journal of Sediment Research* 25(4): 323-339. Available at: http://dx.doi.org/10.1016/S1001-6279(11)60001-2. Accessed January 31, 2019.

U.S. Census Bureau. 2010. 2010 Census Urban and Rural Classification and Ubran Area Criteria. Available at: https://www.census.gov/geo/reference/ua/urban-rural-2010.html. Accessed January 31, 2019.

U.S. Census Bureau. 2013. Age and Sex Composition in the United States: 2013. Available at: https://www.census.gov/data/tables/2013/demo/age-and-sex/2013-age-sex-composition.html. Accessed January 31, 2019.

U.S. Census Bureau. 2015. Educational Attainment of the Population 18 Years and Over, by Age, Sex, Race, and Hispanic Origin: 2015. Educational Attainment in the United States: 2015 - Detailed Tables. Available at: https://www.census.gov/data/tables/2015/demo/education-attainment/p20-578.html. Accessed January 31, 2019.

Willis, R.M., R.A. Stewart, K. Panuwatwanich, P.R. Williams, and A.L. Hollingsworth. 2011. Quantifying the influence of environmental and water conservation attitudes on household end use water consumption. *Journal of Environmental Management* 92(8): 1996-2009. Available at: http://dx.doi.org/10.1016/j.jenvman.2011.03.023. Accessed January 31, 2019.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 12-23, April 2019

Should Contact Recreation Water Quality Standards be Consistent across Hydrological Extremes?

*Lucas F. Gregory[1], Anna Gitter[1], Stephen Muela[1], and Kevin L. Wagner[2]

[1]Texas A&M AgriLife Research, Texas Water Resources Institute
[2]Oklahoma Water Resources Center
*Corresponding author

Abstract: Water quality standards are developed to protect and define when waterbodies support their designated uses including public water supply, recreational use, aquatic life use, and others. Recreational use categories include various activities that typically do not occur under similar hydrologic conditions making protection of all uses challenging. This paper presents a case study where *Escherichia coli* concentrations were grouped by flow rate to demonstrate potential effects of developing use-specific water quality standards for contact recreation. Adopting this approach requires a shift from current water quality policy which applies to all hydrologic conditions; however, it also requires additional data collection on actual usage types and occurrence before it can be implemented. This paper demonstrates that implementing an alternative water quality standards approach can still reasonably protect human health while minimizing taxpayer cost to restore impaired waterbodies.

Keywords: *E. coli, human health, risk*

Safeguarding water quality is essential to protect public health worldwide. Globally, the UN estimates that 780 million people do not have access to clean water, and another 2.5 billion do not have adequate sanitation (UNICEF and WHO 2012). Deficient water treatment and natural phenomena can cause infectious doses of pathogens to be present in surface waters. When consumed, these pathogens can potentially cause water borne illnesses. Pathogen presence estimates commonly use fecal indicator bacteria (FIB) concentrations such as *Escherichia coli* due to cost considerations; however, tools including molecular markers and quantitative microbial risk assessment are evolving and provide additional options for future water quality assessments (Pachepsky et al. 2018). Despite such advances, many locales continue to rely on simple FIB concentrations in water quality standards application.

Escherichia coli and associated pathogens arrive in streams through direct deposition (point sources or defecation into the stream) or indirectly via runoff (nonpoint source pollution). Nonpoint *E. coli* sources undergo various fate and transport processes before arriving in streams (Ferguson et al. 2003), thus affecting *E. coli* and pathogen quantities entering the stream. Regardless of transport mechanism, sediment provides an environmental niche where *E. coli* can persist for extended periods of time (Garzio-Hadzick et al. 2010) and potentially grow (Solo-Gabriele et al. 2000; Stocker et al. 2018). This challenges water managers, as extended persistence and growth can yield *E. coli* populations that may not be associated with recent contamination events (Anderson et al. 2005), thus diminishing potential relationships between *E. coli* concentration and human health risk. It may also lead to impaired waterbody statuses and significant financial investments to correct perceived pollution issues (Wagner et al. 2016).

Known flow rate effects on sediment transport further confound this issue. Research has demonstrated normal and high streamflow induced streambed bacteria releases. In southeast Texas, up to 90% of observed instream *E. coli* load was derived from sediment under baseflow conditions (Brinkmeyer et al. 2015). This deviates from conventional thought that resuspension only occurs during high-flow events (Jamieson et al. 2005). Using artificial floods, Muirhead et al. (2004) and Stocker et al. (2018) demonstrated roughly two order of magnitude increases in *E. coli* concentrations that directly resulted from flow rate induced sediment resuspension. This is not surprising, considering that a literature review by Pachepsky and Shelton (2011) noted that *E. coli* concentrations can be 1 to 2,200 times greater in sediments than in the water column. However, they found that correlations between *E. coli* concentrations in overlying water and sediment are typically very weak. Regardless of correlation, inclusion of high-flow influenced samples in water quality assessments can affect results.

Surface water quality standards are established to protect designated waterbody uses and provide the basis for permitting, compliance, and assessments. Standards include defined designated uses, water quality criteria, and antidegradation policies which largely influence water quality management decisions. Therefore, appropriately developing and applying standards is critical as future management actions and financial resources they require can be significant (Wagner et al. 2016).

Water quality standards established for contact recreation uses based on long-term FIB concentrations aim to protect human health during contact recreation. In work conducted by USEPA (1986) and reaffirmed in 2012 (USEPA 2012), gastrointestinal (GI) illnesses contracted by swimmers at defined bathing beaches were correlated to *E. coli* concentrations. Increased *E. coli* concentrations resulting from recent fecal contamination (point source discharges of treated wastewater effluent) related to a quantified human health risk. Their results formed the basis for development of primary contact recreation standards in many states and countries (Ishii and Sadowsky 2008).

Water quality standards are often applied to flowing water bodies and all flow conditions (TCEQ 2010), although watershed-scale has been reported to effect *E. coli* concentrations (Harmel et al. 2010). Various flow conditions present different inherent risks to engaging in contact recreation. Rational thinking suggests that activities such as swimming, wading by children, and tubing should not occur during high-flows due to increased drowning risks; however, whitewater activities such as kayaking, canoeing, and rafting commonly occur during these conditions. Whitewater recreation is inherently risky and increased flow rates that occur during or shortly after storms greatly increase these recreation opportunities in areas where whitewater streams are not common (Daniel 2004). The existence of these activity types has justified maintaining contact recreation standards at all flow conditions. However, arguments can be made that applying water quality standards at high-flows (floods) is not appropriate due to the natural pollutant flushing that occurs and the inability to effectively manage pollutant sources during these conditions. Further, Dorevitch et al. (2011) found that kayakers typically consume 35-40% less water than swimmers. Thus, an opportunity exists to evaluate other water quality assessment and standards development approaches that could minimize potential financial burdens to society without substantially affecting human health risks. This paper evaluates an admittedly small data set to demonstrate the potential effects of considering *E. coli* samples collected during high-flow events differently in water quality assessment results and discusses policy implications of flow rate and risk-based water quality standards. Results and conclusions are by no means meant to reflect an ubiquitous solution, but rather provide hypothetical evidence that the illness threat to the public may not be considerably different under varying flow regimes and water quality standards if the level and type of use change due to flow condition.

Methods

Site Description

Water quality monitoring was conducted on the Navasota River in east central Texas, USA (Figure 1) from December 12, 2014 through August 30, 2016. The Navasota River spans approximately 200 km from its headwaters to its confluence with the Brazos River. Average annual precipitation in the watershed ranges from 864 to 1,118 mm.

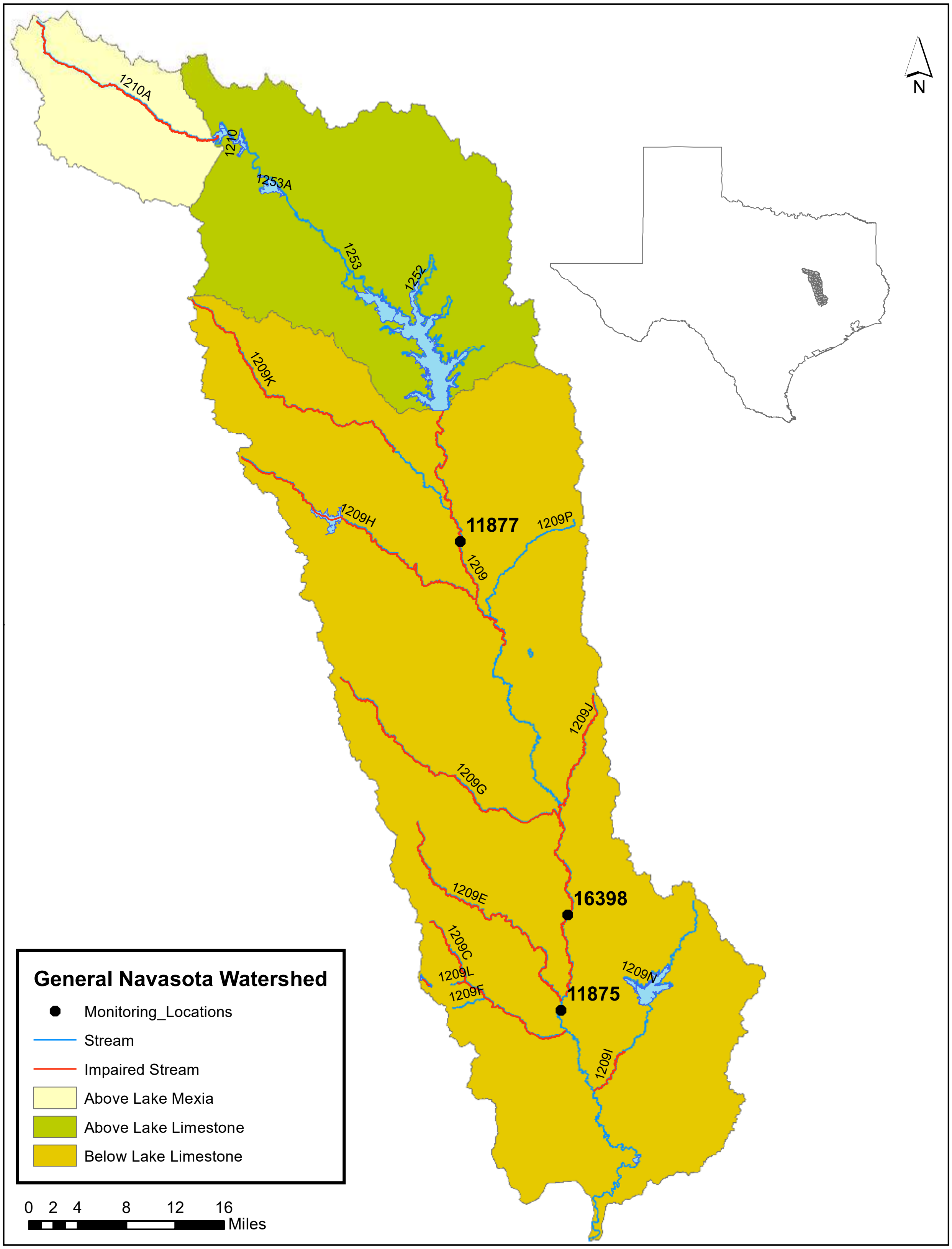


Figure 1. Navasota River Watershed in Central Texas, USA.

Cool, wet winters and hot, dry summers typify local conditions. The watershed is predominantly rural with undeveloped land encompassing >92% of the land area. Grazing land and forests are the dominant land covers. Flood control and water supply are provided by three reservoirs impounding the river in its upper reaches. Lake releases mostly occur in response to rainfall runoff thus making it difficult to distinguish between the effects of dam releases and precipitation/runoff (Gregory et al. 2015).

Three monitoring sites were selected based on geographic location, accessibility, and availability of historic data at each point. For the assessment presented here, only data collected from station 11877 were utilized. This site is located in the upper portion of the river approximately 27.4 km downstream of the largest reservoir. All sites were upstream of urban areas. U.S. Geologic Survey stream gage 08110500 is co-located at this site and records water levels at 15-minute increments. Monitoring occurred biweekly except when high-flows created hazardous sampling conditions or prevented station access. Approximately 25 storm events occurred during the monitoring period. Flow rates above 28.3 m^3/s (bankfull condition) produced hazardous conditions and monitoring was postponed. Missed events were rescheduled as soon as possible. Monitoring techniques followed procedures required by the Texas Commission on Environmental Quality (TCEQ 2012). Large storm events routinely produced discharges of ~300 m^3/s, which are considered major flood events.

Flow volume was recorded using a Sontek ADV (Acoustic Doppler Velocimeter) Flowtracker® or a Sontek RiverSurveyor® M9 Doppler boat. Concurrent pH, water temperature, DO (dissolved oxygen), and specific conductance measurements were recorded with a YSI EXO1 Multiparameter Sonde. Water samples were collected from the centroid of flow at approximately 0.3 m depth and were placed into sterile 200 mL WhirlPak® Thio-Bags®. Samples were transported in ice within six hours to the Soil and Aquatic Microbiology Lab at Texas A&M University for *E. coli* quantification using the EPA 1603 method, a modified thermotolerant membrane filtration approach. Turbidity was determined using a HACH 2100Q field turbidity unit.

Statistical Analysis

Differences in median *E. coli* concentrations between "safe," "unsafe," and "all flow" conditions were evaluated using the non-parametric Mann-Whitney and Kruskal-Wallis tests. Data were non-normally distributed according to Kolmogorov-Smirnov testing. Significance for all analyses was determined using $\alpha=0.05$, thus p values ≤ 0.05 were considered statistically significant. All statistical analyses were conducted using Minitab 17 software (Minitab Inc., State College, PA).

Risk Assessments

Probable human health risks due to potential pathogen exposure during recreational activity was evaluated using two approaches. The first technique applied the linear regression equation developed by Dufour and Ballentine (1986) that was reevaluated and modified for illness type by USEPA (2012) to relate potential swimmer illness rates to *E. coli* geometric mean values. This equation provides the basis of many recreational water quality standards, including those currently applicable in Texas. For this assessment, the below equation was used to estimate expected illness occurrence for differing number of recreators under varying flow conditions.

Illness rate per 1,000 swimmers =

[[Log(E. coli geometric mean) – 1.249]/0.1064]∗4.5

Quantitative microbial risk assessment (QMRA) was performed to estimate human health risks associated with exposure to specific pathogens. Similar approaches have been frequently used in recreational water settings (Schoen and Ashbolt 2010; Soller et al. 2010, 2014, 2015, 2017; McBride et al. 2013; Sunger et al. 2018) and we apply a simple version of these approaches. A point-value QMRA calculation was conducted to provide a rough estimate of the potential human health risks for a GI illness under both safe and unsafe flow conditions and assumed differences in fecal pollution source. FIB concentrations were used to develop a pathogen dose in similar fashion to other assessments in recreational waters (Schoen and Ashbolt 2010; Soller et al. 2010, 2014, 2015; Sunger et al. 2018). Norovirus was selected as the reference pathogen for this "back of the envelope" risk calculation since the pathogen is considered to be the primary agent for GI illnesses in recreational waters (Eftim et al. 2017). The QMRA methodology used in Schoen and Ashbolt (2010) and Soller et al. (2010) was applied for this calculation using the dose equation listed below and assuming input variables presented in Table 1.

Ingested dose of reference pathogen norovirus =

$$\left[\frac{C_{FIB}}{(D_{FIB}*100)}\right]*D_{NoV}*V$$

where C_{FIB} = the concentration of *E. coli* using a culture method in the waterbody (cfu/100mL); D_{FIB} = the density of *E. coli* in wastewater (either raw sewage or treated effluent) (cfu/L); D_{NoV} = the density of norovirus in wastewater (either raw sewage or treated effluent) (genome copies/L); and V = volume of water ingested (mL).

The calculated ingested dose for the reference pathogen is used in a dose-response model to estimate the risk of infection for a specific health endpoint, such as a GI infection. Further, a morbidity ratio can also be used to assess the risk of illness following infection from the pathogen. There are several dose-response models for norovirus in the literature, but the model used (Table 1) assumes viral aggregation of norovirus in the environment and has been recommended for studies assessing health risks in recreational waters (Soller et al. 2017; Van Abel et al. 2017; Sunger et al. 2018).

Table 1. Parameters used in QMRA risk assessment calculation.

Parameter	Use	Value	Units	Assumptions	Reference
E. coli concentration	Safe flow conditions	106.4	cfu/100 mL	Geometric mean	Gregory et al. 2015
	Unsafe flow conditions	510.4	cfu/100 mL	Geometric mean	Gregory et al. 2015
Ingestion rates	Swimming for adults/ children	18.5	mL	Geometric mean (assuming one hour of exposure)	USEPA 2010
	Canoeing/kayaking/ rowing/boating	4.55	mL	Arithmetic mean (includes capsizing during activities and assuming one hour of exposure)	Dorevitch et al. 2011
E. coli density	Secondary treated wastewater	4	log10 cfu/L	Maximum observed value	Rose et al. 2004
	Raw wastewater	8	log10 cfu/L	Maximum observed value	Rose et al. 2004
Norovirus density	Secondary treated wastewater	2.1 log10 removal	log10 GC/L	Average log10 removal for conventional wastewater treatment	Lodder and de Roda Husman 2005; Chaudhry et al. 2017
	Raw wastewater	4.9	log10 GC/L	Upper 95% of the mean; NoV genogroup GII	Eftim et al. 2017
Dose response	Norovirus	P=0.72; μ= 1106	NA	Aggregated; Fractional Poisson (Probability of Illness= P[1-e(-d/μ)])	Messner et al. 2014
	Morbidity ratio	0.6	NA	NA	Soller et al. 2017

Results

In order to recognize instances in which sediment resuspension and nonpoint sources are the likely cause of elevated *E. coli* concentrations, flow events were separated into safe and unsafe conditions for swimming and wading by children (Table 2). Based on recorded flow velocity and stream depths, a discharge of 2.12 m^3/s at the monitoring location was assumed as the upper flow-volume limit that allows for safe swimming and wading (TCEQ 2012). Biweekly monitoring and sampling during the two year study captured *E. coli* concentrations and flow volumes for multiple storm events and baseflow conditions. All data were aggregated into an all flows category for evaluation to represent the current assessment approach.

Statistically, median *E. coli* concentrations were not equal between the safe and unsafe flow categories ($p=0.001$). Between individual categories, safe and unsafe conditions were found to be significantly different ($p<0.001$), but safe conditions and all flows combined were not ($p=0.205$). The presence of several outlier *E. coli* concentrations during high-flow events strongly influenced the median and geometric means in each group (Figure 2), but these could not be excluded as they represent natural occurrences in *E. coli* concentration that sometimes arise from storm events (Figure 3) or unexplained sources that are also commonly observed during baseflow conditions (Muirhead and Meenken 2018). Despite the limited size of the data set, the evaluation suggests that there are potentially different human health risks under safe and unsafe flow conditions. These differing scenarios present an opportunity to create or apply multiple recreation water quality standards on the same waterbody that are based on flow condition and/or the amount and type of recreation that occurs.

Policy Implications

A singular numeric water quality standard for *E. coli* that a waterbody must meet to support recreation uses during all flow conditions may not be practical. In Texas, this was acknowledged and addressed by developing specific standards for different waterbody uses that are as follows:

- Primary contact 1 (126 cfu/100mL): uses presumed to involve a significant water ingestion risk including children wading, swimming, diving, surfing, water skiing, tubing, and whitewater kayaking, canoeing, or rafting.
- Primary contact 2 (206 cfu/100mL): uses are the same as primary contact 1 but are less frequent due to physical limitations of the waterbody and limited access.
- Secondary contact 1 (630 cfu/100mL): common activities with limited body contact including fishing, canoeing, kayaking, rafting, sailing, and motor-boating.
- Secondary contact 2 (1030 cfu/100mL): uses are the same as secondary contact 1 but are less frequent due to physical limitations of the waterbody and limited access.
- Non-contact (2060 cfu/100mL): contact is prohibited by law, or activities with no presumed water ingestion risk including hiking, biking, and birding.

Although this is an improvement from a singular standard, the definition of primary contact recreation includes disparate activities not likely to occur in a waterbody under similar flow conditions. Whitewater sports require much higher flow velocity than swimming, wading by children, or diving. The latter are likely to occur under normal or low-flow conditions, while the former occur during high-flow and flood conditions on all but a few Texas streams that have whitewater year round. Therefore, a logical assumption can be made that water quality may be worse when whitewater sports are likely to occur.

Whitewater sports are inherently dangerous due to adverse hydrologic conditions. Researchers documented whitewater kayaking fatality rates from 3 to 6 deaths per 100,000 kayaking days and injury rates at 4.5 per 1,000 kayaking days. They noted that self-guided paddling trips are significantly more dangerous than commercial trips (Fiore and Houston 2001; Schoen and Stano 2002). Insurance companies also acknowledge the increased risk by routinely increasing policy premiums by \$2 to \$10 per \$1,000 of coverage for frequent extreme sports participants. These persons assume increased risk for bodily harm and

Table 2. *E. coli* concentration descriptive statistics by flow category.

E. coli Concentrations cfu/100mL	N	Median	Standard Deviation	Geometric Mean
Safe flows	32	110	163.1	106.4[a]
Unsafe flows	9	290	1835.7	510.4[a]
All flows	41	124	978.9	150.1[a]

[a] ±36% uncertainty assumed in reported values due to potential influences of sample collection, storage, and analysis for 'good practices' in near surface sampling (Harmel et al. 2016).

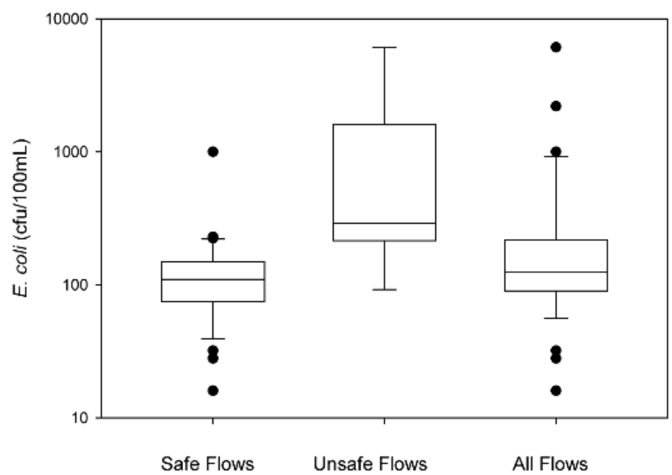


Figure 2. *E. coli* concentrations by flow condition.

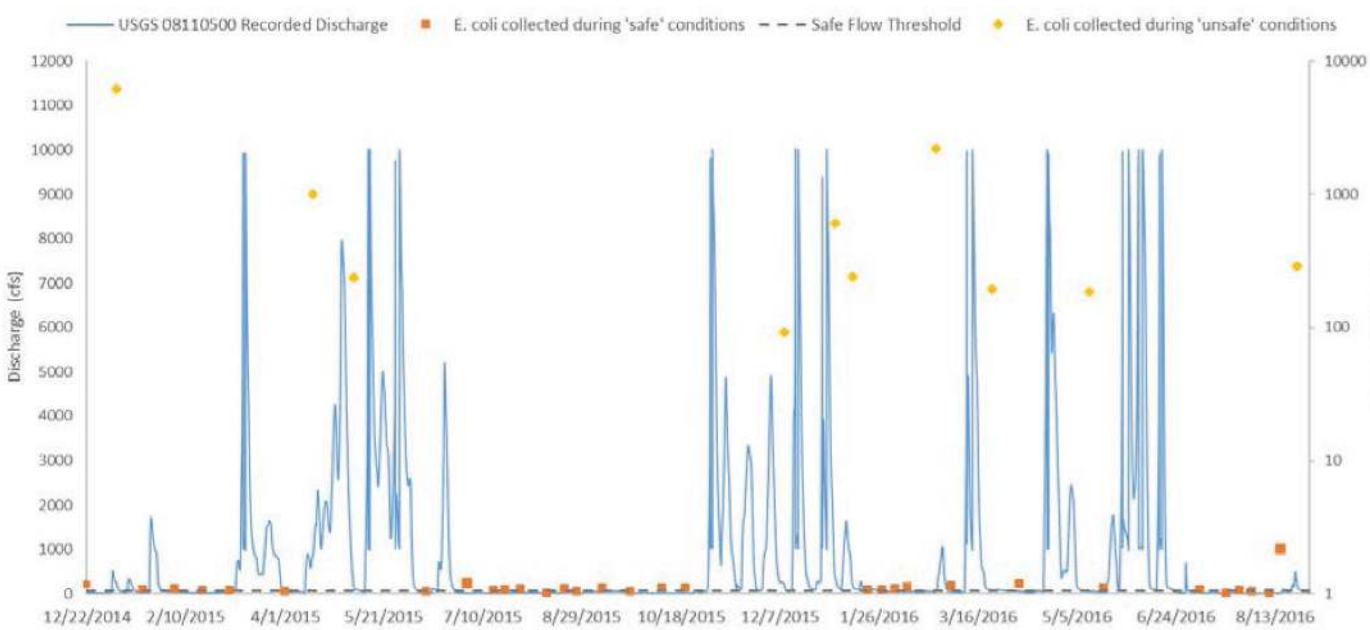


Figure 3. Hydrograph and *E. coli* concentrations at the monitoring station.

death during the activity, thus logic suggests that a slight risk increase for contracting a GI illness is not inappropriate. Implementing less restrictive water quality standards during natural high-flow conditions is likely to adequately protect human health without imparting excessive financial burden to have surface waters meet the most stringent standards under all flow conditions.

A practical option for establishing an alternative contact use category that is applicable for more dangerous flow conditions combines flow rate-based thresholds and risk-based approaches. This will necessitate site-specific criteria establishment but allows more appropriate water quality standards to be selected based on actual use. Utilizing site-specific criteria requires detailed analysis of recreational uses of a waterbody, which is not currently conducted. This is an additional data collection burden required before site-specific criteria could be established or implemented. Waterbodies also change throughout their course, thus it makes sense to evaluate standards at refined scales within streams to ensure that standards are individually relevant and not overly broad. Flow rate-based standards can be used in situations where multiple uses occur at varying flow conditions. Under normal or safe flow conditions, primary contact uses may occur; but under higher flow conditions, these uses become unsafe and are replaced by extreme uses like whitewater sports. Site-specific knowledge can be used to determine a flow threshold where swimming and wading become unsafe. In Texas, surface water quality monitoring procedures prohibit wading in streams where depth multiplied by velocity is ≥ 10 ft^2/s (TCEQ 2012), thus an assumption can be made that flows generating area velocities higher than this threshold are not safe for swimming or wading. Once this threshold is established, the primary contact 1 standard would only apply to water quality samples collected below this flow threshold and excludes values collected above that level. The less restrictive standard applicable for flow conditions supporting extreme water sports should apply for all flow conditions including those above the flow threshold for safe flow conditions. Effectively, this standard applies for all contact recreation uses, but acknowledges the fact that natural hydrologic processes likely result in temporarily reduced water quality.

A risk-based approach to establishing alternative water quality standards can be used to set appropriate risk levels for differing thresholds. This approach considers the number of individual contact recreating on an annual basis. Improvements documenting the quantity of contact uses and the flow conditions when they occur are necessary. For example, if 5,000 individuals swim in a waterbody in a given year under normal flow conditions and only 50 individuals engage in extreme whitewater sports under high-flow conditions, separate standards can be established to allow acceptable *E. coli* concentrations in the waterbody. Current primary contact 1 standards described above predict an illness rate of 36 people per 1,000 individuals.

At the assumed number of swimmers listed above and the primary contact 1 standard, 180 individuals per year may become ill. However, only 1.8 individuals of the extreme sports group may become ill at the same water quality threshold due to the difference in amount of users. Increasing the water quality threshold for high-flow conditions to the secondary contact 1 use standard (630 cfu/100 mL) and applying it to individuals engaged in extreme sports results in 3.27 ill individuals out of the same 50 individuals during this one-year period. Translated to *E. coli* concentrations reported for safe and unsafe flow conditions and assumed number of recreators, the expected number of illnesses are 164 and 3.09, respectively. This is a nominal illness increase relative to the increase in allowable *E. coli* concentrations in all flow conditions.

Similarly, when evaluated using QMRA techniques, the estimated human health risks did not greatly differ between activities and flow conditions when using less stringent water quality standards. QMRA point value estimation provides a broad idea of risks across the assumed recreational scenarios. For primary contact recreation (swimming, wading by children) in safe flow conditions (assuming a geometric mean of 106.4 cfu/100mL) with treated wastewater as the contaminant source, the risk of a GI illness was estimated to be 4.8×10^{-4}. Whitewater type recreation activities occurring during unsafe flow conditions (using a geometric mean of 510.4 cfu/100mL) and primarily raw sewage influent as the contaminant source, the estimated risk for a GI illness would be 7.2 x 10^{-6}. The risk estimates should only be considered "back of the envelope" and an initial starting point for further risk assessment work that considers safe and unsafe flow conditions and their appropriate activities. Results do suggest that the risk of boating/kayaking/canoeing/rowing (and potentially capsizing) in water that exceeds current water quality standards may not pose as much of a risk for a GI illness as previously considered, especially considering the lower frequency of those uses.

Conclusions

The Navasota River provides a case study representative of many low-use waterbodies. Its water quality is currently impaired under the required primary contact 1 standard. Recent waterbody use assessment indicates that primary contact uses occur, but at low frequencies. No instances of use during high-flow conditions were observed or noted in surveys. Application of risk estimates by flow condition demonstrates that the expected number of individuals potentially becoming ill is considerably smaller for unsafe than safe flow conditions due in part to the smaller number of individuals engaged in recreation.

Grouping water quality data by flow threshold revealed significantly different mean *E. coli* concentrations, which suggests that altering water quality standards application as a result of changes in stream flow may not have a detrimental effect on human health protection. This approach requires more site-specific data collection prior to establishing flow rate-based thresholds and associated numeric criteria; however, it may reduce the number of impaired waterbodies by more accurately characterizing their use and allowing an appropriate standard to be selected. We realize that this is not a simple or perfect process, but it is one that has potential to reduce management and restoration costs in waterbodies where significant primary contact uses do not occur at all flow conditions. This allows natural hydrological processes to occur that would prevent waterbodies from fitting into traditional standards categories based on use without causing water quality impairments.

It is not the intent of this paper to promote water quality standards reductions but instead to propose an alternative application of current standards based on actual uses. Stringent standards are important for protecting public health and conserving natural waters; however, water quality standards should incorporate the best available science and acknowledge different levels and types of use that occur. Implementing variable condition standards will not compromise mandates to protect public health, but will support a targeted and reasonable approach that allows limited restoration resources available to be applied in critical areas.

Acknowledgements

Funding for this work was provided in part by grants from the Texas Commission on Environmental Quality and the Texas State Soil and Water Conservation Board. Thank you to Mr. John Derry for his assistance editing and reviewing this manuscript.

Author Bio and Contact Information

Dr. Lucas F. Gregory (corresponding author) currently serves as a senior research scientist for the Texas Water Resources Institute. In this role, he develops effective and efficient projects and provides leadership for multiple watershed assessment, planning, and implementation projects, focusing on water quality impairments in rural Texas water bodies. His research interests include: bacteria fate and transport in aquatic and soil environments, watershed assessment, planning and management, watershed assessment tool application and development, water quality monitoring, efficient water resource utilization, implications of water policy on local watershed decision making, and water and human health interactions. Dr. Gregory can be reached at 578 John Kimbrough Blvd., College Station, TX 77843 or LFGregory@ag.tamu.edu.

Anna Gitter is currently working as a Research Assistant for the Texas Water Resources Institute as well as pursuing her Ph.D. in the Water Management and Hydrological Sciences Program at Texas A&M University. Her research focuses on better understanding the impacts of water quality on human health, with a specific interest in quantitative microbial risk assessment. Her current Ph.D. research aims to quantify the human health risks associated with exposure to contaminated well water following flooding from Hurricane Harvey in Texas. Ms. Anna Gitter can be reached at 578 John

Kimbrough Blvd., College Station, TX 77843 or anna.gitter@ag.tamu.edu.

Stephen Muela is currently working as a research participant through the Oak Ridge Institute for Science and Education (ORISE) at the US EPA in the Office of Water. His research focuses on applied hydrologic modeling and geospatial analyses. He has his Masters of Water Management and Hydrological Science from Texas A&M University, and a MS in Geosciences from Trinity University, in San Antonio, Texas. Mr. Muela can be reached at 19 T Street NW, Washington D.C. 20001 or muela.stephen@gmail.com.

Dr. Kevin L. Wagner currently directs the activities of the Oklahoma Water Resources Center and is an associate professor in the department of plant and soil sciences at Oklahoma State University. His primary responsibilities include organizing interdisciplinary teams to further the knowledge of water resources in Oklahoma and connecting with agency personnel, stakeholders, and constituents. His research interests include watershed planning, watershed restoration, water use and efficiency, bacteria fate and transport, and best management practice effectiveness. Dr. Wagner can be reached at 139 Agricultural Hall, Stillwater, OK 74078 or kevin.wagner@okstate.edu.

References

Anderson, K.L., J.E. Whitlock, and V.J. Harwood. 2005. Persistence and differential survival of fecal indicator bacteria in subtropical waters and sediments. *Applied and Environmental Microbiology* 71(6): 3041-3048.

Brinkmeyer, R., R.M. Amon, J.R. Schwarz, T. Saxton, D. Roberts, S. Harrison, N. Ellis, J. Fox, K. Diguardi, M. Hochman, S. Duan, R. Stein, and C. Elliott. 2015. Distribution and persistence of *Escherichia coli* and Enterococci in stream bed and bank sediments from two urban streams in Houston, TX. *Science of the Total Environment* 502: 650-658.

Byapanahalli, M.N., B.M. Roll, and R.S. Fujioka. 2012. Evidence for occurrence, persistence, and growth potential of *Escherichia coli* and Enterococci in Hawaii's soil environments. *Microbes and Environments* 27(2): 164-170.

Chaudhry, R.M., K.A. Hamilton, C.N. Haas, and K.L. Nelson. 2017. Drivers of microbial risk for direct potable reuse and de facto reuse treatment schemes: The impacts of source water quality and blending. *International Journal of Environmental Research and Public Health* 14(6): 635.

Daniel, S.H. 2004. *Texas Whitewater*. Texas A&M University Press, College Station, Texas.

Dorevitch, S., S. Panthi, Y. Huang, H. Li, A.M. Michalek, P. Pratap, M. Wroblewski, L. Liu, P.A. Scheff, and A. Li. 2011. Water ingestion during water recreation. *Water Research* 45(5): 2020-2028.

Dufour, A. and R. Ballentine. 1986. Ambient Water Quality Criteria for Bacteria, 1986: Bacteriological Ambient Water Quality Criteria for Marine and Fresh Recreational Waters. National Technical Information Service, Department of Commerce, U.S.

Eftim, S.E., T. Hong, J. Soller, A. Boehm, I. Warren, A. Ichida, and S.P. Nappier. 2017. Occurrence of norovirus in raw sewage- A systematic literature review and meta-analysis. *Water Research* 111: 366-374.

Ferguson, C., A.M. de Roda Husman, N. Altavilla, D. Deere, and N. Ashbolt. 2003. Fate and transport of surface water pathogens in watersheds. *Critical Reviews in Environmental Science and Technology* 33(3): 299-361.

Fiore, D.C. and J.D. Houston. 2001. Injuries in whitewater kayaking. *British Journal of Sports Medicine* 35(4): 235-241.

Fujioka, R., C. Sian-Denton, M. Borja, J. Castro, and K. Morphew. 1998. Soil: The environmental source of *Escherichia coli* and Enterococci in Guam's streams. *Journal of Applied Microbiology* 85: 83S-89S.

Garzio-Hadzick, A., D.R. Shelton, R.L. Hill, Y.A. Pachepsky, A.K. Guber, and R. Rowland. 2010. Survival of manure-borne *E. coli* in streambed sediment: Effects of temperature and sediment properties. *Water Research* 44(9): 2753-2762.

Gerba, C.P. 2009. Indicator microorganisms. In: *Environmental Microbiology* (second edition), R.M. Maier, I.L. Pepper, and C.P. Gerba (Eds.). Academic Press, Boston, MA, pp. 485-498.

Gregory, L.F., A. Gitter, and K. Lazar. 2015. *Basin Approach to Address Bacterial Impairments in the Navasota River Watershed*. Texas Water Resources Institute Technical Report 476, College Station, Texas.

Harmel, R.D., R. Karthikeyan, T. Gentry, and R. Srinivasan. 2010. Effects of agricultural management, land use, and watershed scale on *E. coli* concentrations in runoff and streamflow. *Transactions of ASABE* 53(6):1833-1841.

Harmel R.D., J.M. Hathaway, K.L. Wagner, J.E. Wolfe, R. Karthikeyan, W. Francesconi, and D.T. McCarthy. 2016. Uncertainty in monitoring *E. coli* concentrations in streams and stormwater runoff. *Journal of Hydrology* 534: 524-533.

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

Jamieson, R.C., D.M. Joy, H. Lee, R. Kostaschuk, and R.J. Gordon. 2005. Resuspension of sediment-associated *Escherichia coli* in a natural stream. *Journal of Environmental Quality* 34(2): 581-589.

Lodder, W.J. and A.M. de Roda Husman. 2005. Presence of noroviruses and other enteric viruses in sewage and surface waters in the Netherlands. *Applied and Environmental Microbiology*. 71(3): 1453-1461.

McBride, G.B., R. Stott, W. Miller, D. Bambic, and S. Wuertz. 2013. Discharge-based QMRA for estimation of public health risks from exposure to stormwater-borne pathogens in recreational waters in the United States. *Water Research* 47(14): 5282-5297.

Messner, M.J., P. Berger, and S.P. Nappier. 2014. Fractional poisson- A simple dose response model for human norovirus. *Risk Analysis* 34(10): 1820-1829.

Muirhead, R.W., R.J. Davies-Colley, A.M. Donnison, and J.W. Nagels. 2004. Faecal bacteria yields in artificial flood events: Quantifying in-stream stores. *Water Research* 38(5): 1215-1224.

Muirhead, R.W. and E.D. Meenken. 2018. Variability of *Eschericia coli* concentrations in rivers during base-flow conditions in New Zealand. *Journal of Environmental Quality* 47(5): 967-973.

Odonkor, S.T. and J.K. Ampofo. 2013. *Escherichia coli* as an indicator of bacteriological quality of water: An overview. *Microbiology Research* 4: e2. Available at: https://doi.org/10.4081/mr.2013.e2. Accessed April 19, 2019.

Pachepsky, Y.A. and D.R. Shelton. 2011. *Escherichia coli* and fecal coliforms in freshwater and estuarine sediments. *Critical Reviews in Environmental Science and Technology* 41(12): 1067-1110.

Pachepsky, Y., A. Allende, L. Boithias, K.H. Cho, R. Jamieson, N. Hofstra, and M. Molina. 2018. Microbial water quality: Monitoring and modeling. *Journal of Environmental Quality* 47(5): 931-938.

Rose, J.B., S.R. Farrah, V.J. Harwood, A.D. Levine, J. Lukasik, P. Menendez, and T.M. Scott. 2004. *Reduction of Pathogens, Indicator Bacteria, and Alternative Indicators by Wastewater Treatment and Reclamation Processes.* Water for Reuse Final Report. IWA Publishing, London, United Kingdom.

Schoen, M.E. and N.J. Ashbolt. 2010. Assessing pathogen risks to swimmers at non-sewage impacted recreational beaches. *Environmental Science & Technology* 44(7): 2286-2291.

Schoen, R.G. and M.J. Stano. 2002. Year 2000 whitewater injury survey. *Wilderness and Environmental Medicine* 13(2): 119-124.

Soller, J.A., M.E. Schoene, T. Bartrand, J.E. Ravenscroft, and N.J. Ashbolt. 2010. Estimated human health risks from exposure to recreational waters impacted by human and non-human sources of faecal contamination. *Water Research* 44(16): 4674-4691.

Soller, J.A., M.E. Schoen, A. Varhese, A.M. Ichida, A.B. Boehm, S. Eftim, N.J. Ashbolt, and J.E. Ravenscroft. 2014. Human health risk implications of multiple sources of faecal indicator bacteria in a recreational waterbody. *Water Research* 66: 254-264.

Soller, J.A., T. Bartrand, J. Ravenscroft, M. Molina, G. Whelan, M. Schoen, and N.J. Ashbolt. 2015. Estimated human health risks from recreational exposures to stormwater runoff containing animal faecal material. *Environmental Modelling & Software* 72: 21-32.

Soller, J.A., M. Schoen, J.A. Steele, J.F. Griffith, and K.C. Schiff. 2017. Incidence of gastrointestinal illness following wet weather recreational exposures: Harmonization of quantitative microbial risk assessment with an epidemiologic investigation of surfers. *Water Research* 121: 280-289.

Solo-Gabriele, H.M., M.A. Wolfert, T.R. Desmarais, and C.J. Palmer. 2000. Sources of *Escherichia coli* in a coastal subtropical environment. *Applied Environmental Microbiology* 66(1): 230-237.

Stocker, M.D., M. Penrose, and Y.A. Pachepsky. 2018. Spatial patterns of *Escherichia coli* concentrations in sediment before and after high-flow events in a first-order creek. *Journal of Environmental Quality* 47: 958-966.

Sunger, N., K.A. Hamilton, P.M. Morgan, and C.N. Haas. 2018. Comparison of pathogen-derived 'total risk' with indicator-based correlations for recreational (swimming) exposure. *Environmental Science and Pollution Research*. Available at: https://doi.org/10.1007/s11356-018-1881-x. Accessed April 19, 2019.

TCEQ. 2010. Texas Administrative Code. Texas Surface Water Quality Standards. Austin, TX: TCEQ. Available at: http://texreg.sos.state.tx.us/public/readtac$ext.ViewTAC?tac_view=3&ti=30&pt=1. Accessed April 19, 2019.

TCEQ. 2012. Surface Water Quality Monitoring Procedures, Volume 1: Physical and Chemical Monitoring Methods. Austin, TX: TCEQ. Available at: https://www.tceq.texas.gov/publications/rg/rg-415. Accessed April 19, 2019.

UNICEF and WHO. 2012. Progress on Drinking Water and Sanitation. 2012 Update. Available at: https://www.unicef.org/media/files/JMPreport2012.pdf. Accessed April 23, 2019.

USEPA. 1986. Ambient Water Quality Criteria for Bacteria, 1986: Bacteriological Ambient Water Quality Criteria for Marine and Fresh Recreational Waters. National Technical Information Service, Department of Commerce, U.S. Available at: https://www.waterboards.ca.gov/water_issues/programs/tmdl/records/region_5/1986/ref2435.pdf. Accessed April 19, 2019.

USEPA. 2010. Quantitative Microbial Risk Assessment to Estimate Illness in Freshwater Impacted by Agricultural Animal Sources of Faecal Contamination. Washington, D.C.: Office of Water. 822-R-10-005. Available at: https://www.epa.gov/sites/production/files/2015-11/documents/quantitiave-microbial-risk-fecal.pdf. Accessed April 19, 2019.

USEPA. 2012. Recreational Water Quality Criteria. Washington, D.C.: Office of Water. 820-F-12-058. Available at: https://www.epa.gov/wqc/2012-recreational-water-quality-criteria. Accessed April 19, 2019.

Van Abel, N., M.E. Schoen, J.C. Kissel, and J.S. Meschke. 2017. Comparison of risk predicted by multiple norovirus dose-response models and implications for quantitative microbial risk assessment. *Risk Analysis* 37(2): 245-264.

Wagner, K.L., L.F. Gregory, and A. Berthold. 2016. Water quality management. In: *Sustainable Water Management and Technologies*, D.H. Chen (Ed.). CRC Press, Boca Raton, FL.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 24-34, April 2019

IWRM and the Nexus Approach: Versatile Concepts for Water Resources Education

Neil S. Grigg

Department of Civil and Environmental Engineering, Colorado State University

Abstract: Integrated Water Resources Management (IWRM) and the nexus approach are tools to identify solutions for water problems across interdependent sectors with interacting social and natural systems. Although both tools aim at solutions for complex water issues using an interdisciplinary approach, IWRM is a management process and the nexus approach is a systems tool to characterize problems. By clarifying their attributes and providing examples, instructors can use them to explain broad social problems and offer practical frameworks for problem-solving. Given their breadth, IWRM and the nexus approach can seem vague and attract criticism, but if they are replaced, the need for them will endure. The concepts are explained, and similarities between them are explored in the paper. Case study sources for them are identified, and the cases are classified by the processes of water resources management as applied across related sectors. How the concepts and their corresponding case studies can be used will vary by context. Suggestions are made for interdisciplinary instruction and discussions in disciplinary settings.

Keywords: *integrated management, case studies, water-energy-food nexus, complex problems*

Water managers and leaders require new tools to identify integrated solutions for problems across many complex and interdependent sectors. Both Integrated Water Resources Management (IWRM) and the nexus approach are tools developed to address issues where water actions interact with social and natural systems (Global Water Partnership (GWP) 2017a; UNECE 2017). Both concepts support problem-solving approaches where diverse groups can cooperate to address shared problems, but how they work can seem vague and abstract.

IWRM and the nexus approach meet recognized needs for tools to address integrative issues. Explaining them for different instructional settings can illuminate solutions in a global, economic, environmental, and societal context, which is a criterion to accredit engineering programs and can apply to other disciplines (ABET 2017). In addition, IWRM and the nexus approach offer practical frameworks for problem-solving. To implement these, a good place to start is in the educational arena, and the Universities Council on Water Resources (UCOWR) is positioned to lead in explaining them through its forums for interdisciplinary cooperation.

While IWRM and the nexus approach are useful concepts, they are difficult to explain and easy to criticize. However, the increasing scopes and scales of global water problems require such complex approaches (World Water Council 2017a). IWRM and the nexus approach will be subject to varying interpretations, and writers have tried to explain how they relate to each other (Rasul and Bikash 2016). Despite this interest in them, IWRM and the nexus approach continue to lack conceptual clarity (Water, Food, Energy Nexus Security Resource Platform 2017a). The fuzziness of these concepts is not unique, however, as the academic field of complex problem-solving is itself in disarray and in need of definitions (Quesada et al. 2005). Therefore, water resources educators should not hesitate to tread areas where solutions are not always clear-cut.

In IWRM, the lack of clarity causes controversy and some thought leaders have even recommended discarding the concept (Tortajada and Biswas

2017). Others suggest replacing it with names such as “Problem-driven iterative adaptation,” while retaining IWRM principles (Butterworth 2014). Examples of the nexus approach also show a wide divergence in understanding about its purpose and usefulness. A popular version of it is the water-energy-food nexus (WEFN), which can be used, for example, to quantify virtual water in international trade (Hanlon et al. 2013).

IWRM is usually defined broadly as a “process to promote the coordinated development and management of water, land and related resources to maximize economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems and the environment” (GWP 2017a). While it may be a “guiding water management paradigm” (Borchardt et al. 2016), it is not really a definite process because it lacks a systematic series of actions taking place in a definite manner. Rather, it is more of an instrument of change, promoting the use of management principles in problem-solving. The nexus approach lacks a formal definition and is explained in different ways (U.S. Department of Energy 2014; Benson et al. 2015). It generally means that when actions are taken in one sector, it is necessary to consider how they will affect other sectors (UNU-Flores 2017).

Though defining the two concepts precisely is difficult, the need for IWRM and the nexus approach to provide orderly solutions to messy water-related problems will endure. Rather than a problem, this can be an opportunity if effective instructional approaches for them are developed. This paper explores the similarities between IWRM and the nexus approach and offers a framework to explain them in instructional settings. In the paper, both concepts are reviewed, case studies are assessed and placed into categories, and suggestions are made for their use in instructional settings.

Co-evolution of IWRM and the Nexus Approach

Both IWRM and the nexus approach emerged in response to the needs for interdisciplinary tools to address complex issues. These same needs led to integrative paradigms in other sectors, such as the currently-popular “One Health Initiative” (2017). In fact, many new concepts have been developed to explain complex and interacting sectors involved in water issues. Most seek to displace what are perceived as linear and technocratic approaches to problem-solving.

To understand the IWRM concept, it is useful to explore its origins. It emerged from international dialogue dating from the 1977 United Nations Mar del Plata Water Conference (Biswas 2011). The concept has been developed and promoted by the World Water Council (2017b) and the Global Water Partnership (GWP 2017b), whose Technical Committee has responsibility to shepherd it. The origins of the nexus concept also date back several decades. As used in environmental management, it dates to the 1980s, but it has gained prominence recently. Its broad vision, as explained at the Bonn 2011 conference on the WEFN, is to improve water, energy, and food security by integrating management and governance, building synergies, promoting sustainability, and transitioning to a green economy (Hoff 2011; Martin-Nagle et al. 2011; UNU-Flores 2017).

The underlying concept of IWRM is water management itself, which is used in different contexts, such as environmental water, water in pipes, wastewater, stormwater, and floodwater. These contexts have led to a related integrative paradigm named “Total Water Solutions,” that signals how water managers are “interested in water no matter where it is found” (LaFrance 2013). While this may sound simplistic, it is actually a powerful idea about transforming how water utilities approach management in an integrated fashion. Another currently-popular slogan is “One Water,” which advocates viewing drinking water, wastewater, and stormwater as connected.

Defining IWRM is complicated by the fact that no consensus has been reached on precisely defining the related concept of water management itself. An example of its definition is “the control and movement of water resources to minimize damage to life and property and to maximize efficient beneficial use” (United Nations Secretary-Generals’ Advisory Board on Water & Sanitation 2017). However, once the word “resources” is added to “water management,” the definition can become more complex. Savenije and Hoekstra (2017) explained that “People from different backgrounds seldom have the same idea about what water resources management implies." They concluded that water resources management is a diffuse field that includes "the whole set of scientific, technical, institutional, managerial, legal, and operational activities required to plan, develop, operate, and manage water resources."

Although the concept of water management has expansive explanations, it is still narrower than IWRM, whose most-quoted definition is the one by the GWP (2017a) that was given earlier. The use of language to explain the concepts is important, and IWRM may simply be the same as water resources management, but with more emphasis on its integrative attributes.

Writers have criticized IWRM as too visionary and vague, oriented too much toward engineering or planning, and indifferent to societal needs (Ioris 2008; Moss 2010; Campana 2011). In the extreme, it is criticized as being an instrument of establishment institutions to promote a water crisis and impose elitist solutions (Trottier 2008). In their criticism of IWRM, Tortajada and Biswas (2017) wrote, "these non-performing concepts will become even more irrelevant in a future world which will be more complex, uncertain and unpredictable. Future water problems cannot be solved by using past paradigms and experiences that have not proven to be effective."

In response to such criticisms, IWRM could be viewed as not a process at all, but a vision of what water management should be (Moss 2010), or it could be viewed as simply good water resources management (Braga 2017). Addressing the criticisms, the Stockholm Water Institute (2019) explained that despite the criticism, it is an instrument of change to deal with the fragmented approach to water resource management. In that sense, it is like a bandage applied to the poorly-defined concept of water resources management.

No single definition is dominant for the nexus approach and, because it lacks the extensive analysis that IWRM has attracted, no systematic criticisms have emerged. A nexus is a connection between things, but this simple concept becomes more complex by explaining which attributes of connected sectors are included. Explanations of the nexus approach, as applied to different *environmentally-related sectors, sound like the familiar "systems approach" (*Vijay et al. 2014; UNU-Flores 2017) or simply as an approach that considers issues jointly, which is a goal of comprehensive planning itself (Rasul and Sharma 2016).

It is evident that IWRM and the nexus approach have similar goals, take a multi-sector approach, and focus on overlaps across sectors with the goal of making better plans by understanding interactions (Stockholm Water Institute 2019; Water, Energy & Food Security Resource Platform 2017a). These similarities lead educators to attempt to explain them, but without much distinction. For example, the University of Geneva (2017) offers a course module entitled "From Integrated Water Resource Management to the Water-Food-Energy and Ecosystem Nexus." It uses IWRM to focus on the coordinated management of water and associated resources and the nexus approach to show how water users interact with other sectors. It is not clear why the nexus approach is needed to supplement IWRM, since it already includes interaction among sectors. Perhaps an explanation is that IWRM starts with a water management perspective while the nexus approach is a way to view elements of a system (United Nations General Secretary's Advisory Board on Water and Sanitation 2017). However, the nuances between them are difficult to discern because both are multi-sector tools.

One nuance is that the leadership role may be different between IWRM and the nexus approach. In IWRM, one set of leaders comprises those who manage water itself. Another set comprises officials who make decisions about water, but who may be involved with issues of other sectors. Examples include local planners and officials, including regulators. With the nexus approach, assignment of the leadership role is not fixed because it is about a cooperative approach to identifying win-win strategies among diverse players and is not a process itself.

IWRM and the Nexus Approach as Paradigms for Complex Problems

While IWRM and the nexus approach are both attempts to characterize and resolve complex issues related to water, the question remains of how they can be used. Their application to social issues is especially challenging, where problems seem nuanced, difficult to define, and needing more careful approaches than technical solutions would indicate. An example is the shift toward non-structural solutions to flood problems, where typical engineering solutions had favored dams and channel works, but Gilbert White changed the conversation to emphasize human adjustment to floods (American Association of Geographers 2017).

IWRM can be sensitive to social issues as shown by the fact that it has a management instrument for "promoting social change" (GWP 2017c). It is related to other approaches proposed for complex social problems, which advocate incremental solutions rather than single projects. This approach to messy problems is explained by Hassan (2014), who proposed a "social lab" process to involve stakeholders struggling to seek a consensus. In reviewing his book, Bernholz (2014) wrote that such approaches are needed because standard planning processes of government and civil society are out of step with current knowledge of complexity, systems, networks, and how change happens.

In a similar vein, Mirumachi (2015) wrote pessimistically that managing water is a "wicked problem" and straightforward solutions will not work. She also thought that water managers might claim a spirit of cooperation, but it is not real because national interests and power asymmetries will drive the outcomes. Elinor Ostrom (1990) formulated an "Institutional Analysis and Development Framework" to relate concepts of collective action problems to social structures, positions, and rules, addressing complex problems by connecting policy analysis to analytical approaches used in the physical and social sciences.

The general concept of institutional analysis is used in different ways to explain social processes, which are inherent in IWRM. Ziegler (1994) offered a method that used key questions to define a situation by learning what goes on, what processes need adjustment, what know-how is available, what should happen, and what the impacts of change are. By adding details about authority and participation, laws and controls, incentives, roles, and management culture, a conceptual model of how the management and control systems work can be created. It will include identification of the key issues in each set processes and institutional changes required to lead to improvement.

These methods align with the discipline of systems thinking, which is a popular method of looking at the big picture. As explained by Senge (1990), systems thinking is one of the five disciplines of creating the learning organization. The others are personal mastery, mental models, shared vision, and team learning. The tools of systems thinking coordinate well with IWRM, and the nexus approach could also be viewed as a systems tool to create a valid mental model.

There are many tools for systems thinking, ranging from mind maps to complex simulation algorithms. One tool, the DPSIR framework (for drivers, pressures, states, impacts, responses), can be used to create a conceptual systems model of a nexus that includes the control points available to water managers. It can also show cause-effect relationships in social-ecological systems and has been used to describe many types of systems (Gari et al. 2015). The effects on water systems from basic drivers such as population growth and climate change can be shown, along with derived drivers such as changes in land use, species transitions, technology, external inputs such as irrigation, resource consumption, and other natural physical and biological drivers. (Bradley and Yee 2015).

The existence of competing paradigms leads to the conclusion that the science of complex systems is not settled. To illustrate, the nexus concept is a special case of "coupled natural and human systems," which is the name of a 16-year program of the U.S. National Science Foundation (2017). In this program, investigators have studied many nexus situations involving overlapping systems that link water to other human and natural systems. Many of the NSF studies can be used as examples of the systems approach.

IWRM and Nexus Case Studies

Despite extensive discussion of IWRM and the nexus approach, both still lack conceptual clarity. Studying how groups perceive them may help more than to focus on abstract definitions. To study this, cases of IWRM and the nexus approach are reviewed in this section. The examples of IWRM cases are from a previous study (Grigg 2015, 2016). They included those published by the GWP, mission-specific organizations, research institutes, individual researchers, and private companies. These were classified into archetypes by management situations, which will be listed later.

Eleven case IWRM categories were identified (Grigg 2016): institutional development; policy planning; river basin coordination planning; program planning; infrastructure planning; operations planning and assessment; regulation; financing; conflict management; analysis and assessment; and knowledge and information support. All of these categories include multi-sector cases, and it is evident that the nexus approach could be used within them to identify opportunities for resource savings and optimization.

Like the IWRM cases, the nexus cases address diverse situations with the central theme of connection of water and energy to some aspect of food systems. The examples are from a workshop on the WEFN that was co-organized by the writer and from the Water, Energy & Food Security Resource Platform (2017b). While the nexus cases are mainly about the WEFN, other combinations are also possible, such as water-climate and water-health.

The WEFN cases can seem like a laundry list, but they are really examples of systems methods used in inter-sectoral resource management problems. Examples from agriculture include energy from biomass, changes in grass cover in forest regions, and interventions to improve water quality, among others. Examples for village development in developing countries include a household biogas digester, improved cook stoves, and a biomass gas-based mini grid. Other examples include a national-level natural resources policy study, a book on the WEFN and the green economy, and the nexus applied to river basin management. Nexus cases are offered by a more diverse set of sources than IWRM cases. For example, two major professional associations included examples from agriculture, energy and environmental management (IUCN and IWA, 2017), and references to WEFN cases by other groups (German Association for International Cooperation and Local Governments for Sustainability (GIZ and ICLEI) 2014; Colorado State University 2017; GRACE Communications Foundation 2017; LIPHE 2017; World Business Council for Sustainable Development 2017).

WEFN projects organized by the U.S. National Science Foundation (2017) also illustrate the nexus approach and are a good source of instructional resources. Workshops organized in the projects were place-based, issue-based, and technology-based, and they showed the WEFN as applied in urban and rural contexts.

The selection of cases as outlined above shows similarities and differences between IWRM and the WEFN. IWRM is a management concept, and the nexus approach is a systems tool to identify inter-relationships to exploit when taking actions to improve resource sustainability. Stated another way, IWRM begins with something to be managed and the nexus approach begins by looking for something to manage. After interrelationships among resources are identified, projects or other actions may be formulated, whether they are water-centric or not. The nexus is not a process; however, if it is used as the basis for an action approach, then it is an integral part of a process.

Both concepts can be used in problem-solving situations, but the nexus concept applies more to the problem formulation phase, and additional actions must be planned to create an action process. IWRM as a management process extends across all steps, including project delivery and regulatory actions.

As an example where the two constructs seem similar, if IWRM is applied to a situation involving coordinated management of water, food, and energy in a watershed the two approaches seem almost identical. As an example where they are different, if a WEFN case is about recycling food waste to generate biogas energy for a community, it will not involve IWRM. Both constructs can involve multiple sectors, but with IWRM the situations are water-centric and leadership is presumed to be with the water sector. In the nexus approach, leadership choices are not specified and might fall outside of the water sector. IWRM can be applied at different levels and might focus only on the water sector, as in the integration of multiple water services.

Use in Instructional Settings

If IWRM and the nexus approach are used in instruction, benefits can accrue at two levels: explanations of broad approaches to solving societal problems can be useful in courses such as environmental science and policy where water is not the only topic, and explanations of specific tools can be useful in focused water management courses.

Instruction about broad approaches to societal problems aligns with the vision of Boyer (1996) to couple university scholarship with engagement in society. These should be pointed toward public leadership, which is a main subject of interdisciplinary schools of public administration and government. Some instruction about public leadership is needed in any field concerned with water management, even if it is not the main topic.

Explanations of broad approaches can begin with general discussions of societal problems and include water issues such as scarcity, pollution, flood damages, and lack of safe drinking water, among other problems. Once these broad problems are noted, case studies in the frameworks of IWRM and the nexus approach can demonstrate how knowledge and engagement apply to complex problems involving different sectors. The instructor can select cases that illustrate knowledge and tools specific to the relevant discipline. For example, the GWP (2017d) IWRM Toolbox includes cases to illustrate how discipline-oriented management instruments can be applied.

Many other cases are available. To illustrate a powerful lesson with both IWRM and nexus attributes, the Cochabamba (Bolivia) Water War case draws in several significant global issues (GWP 2017e). The central issue is management of an urban water supply system to improve performance and access. A government-sponsored attempt to privatize the system failed after large-scale social unrest, and the water system was turned over to a citizen cooperative to manage. The unrest led to a change in government and the presidency of Evo Morales. The case received wide attention through a book (Olivera 2004) and a movie entitled "Even the Rain." The nexus issue is in the systems combination of technical, economic, and social issues in the city. Instructors can use this case for different learning objectives relating to poverty, urban economics, health, and water management.

Specific tools for water management courses will have immediate impact at a practical level for students who will work in the water sector. Examples of these tools from the GWP (2017d) Toolbox can be derived from the management instruments, which range broadly across disciplines. A case that features a technical tool such as a decision support system might be chosen. The GWP (2017f) offers a case about planning in the Nile River Basin, with many lessons about shared governance, transboundary water management, hydrologic change, and various uses of water in a large and complex international basin. Given their popularity, many other cases about decision support systems can be located easily in research journals of the water sector.

As another example of IWRM instruction, the writer's course in Water Resources Planning and Management explains the concept in a general way. The lesson plan focuses on sectors such as water supply and hydroelectricity, and explains tools such as hydrologic simulation models that are used in safe yield analysis. This work is used in cooperation with an instructor in Pakistan, who is implementing a course with the title Integrated Water Resources Management (U.S.-Pakistan Centers for Advanced Studies in Water 2017). The course has some broad content, while the remainder focuses on concepts and tools for water resources management itself. Another cooperative effort was an IWRM training course in Peru for managers in the national water agency. It also included broad concepts but emphasized specific tools of management needed in Peru.

Instructors in many types of courses can consider inclusion of instruction about IWRM or the nexus approach. As explained by Savenije and Hoekstra (2017), water resources management should include diverse points of view, so no single course or program will have a monopoly on it. Only a few courses will use IWRM as a title, and most of those will probably be short courses for specialized training (United Nations Department of Economic and Social Affairs 2017). It would be surprising to find a full course with nexus in the title, although a course on socio-environmental modeling might include much of the same content. A discussion of the nexus concept can be embedded in an explanation of IWRM and used as a conceptual framework to explain intersections of problems and the special interests of sectors.

Considering the disciplines represented in UCOWR, it is evident that interest in IWRM should be significant to the interdisciplinary community of scholars participating in its annual conferences. Examples of possibilities for IWRM instruction include water engineering and science courses such as hydrology, modeling, systems analysis, and river mechanics; geography courses such as water resources planning; political science and sociology courses about water law and social aspects of water; economics courses such as water resources economics; and applied courses such as irrigation management. Additional topics such as climate change, ecology, coastal water, and environmental health also relate strongly to water management.

Both IWRM and the nexus approach can focus on public leadership to foster cooperative solutions, and instruction can show how win-win strategies for water management can be identified to increase total social returns and opportunities to correct inefficiencies and injustices.

Presentations about IWRM and the nexus approach should present material with academic as well as practical content, or in a "pracademic" way (Stockholm Water Institute 2019; Posner 2009). This could begin with the definitions presented earlier in the paper to clarify that water is managed in different contexts and must respond to many diverse sector needs. Then, the explanation can be about how an integrated approach will be better than a single-purpose approach.

Although the courses might not share the IWRM brand, many of them will be closely related to it. A reference to show the diversity and richness of programs and courses will be the UNESCO Chairs related to water resources (International Center for Integrated Water Resources Management (ICIWaRM) 2017). The list of UNESCO (2017) Chairs shows some 40 topics distributed around the world, and many of them involve IWRM topics that could be used in instruction. These include integrated river management; conflict resolution and transboundary water governance; hydropolitics; hydroinformatics; ecohydrology; sustainable water services and cities; water, culture, and indigenous peoples; and gender in water management, among others.

Conclusions

The concepts of IWRM and the nexus approach were developed because management tools were needed to address the complex issues inherent in the connections of water decisions to other sectors. Based on its levels of acceptance, IWRM became popular among international water leaders to provide a framework to address complex water-related issues with accepted principles and management instruments. The nexus concept has also become popular for use in framework and policy studies, as well as in planning for co-management of water, energy, and food resources. It can help stakeholders identify win-win projects, programs, and partnerships at different levels.

How IWRM and the nexus approach can be applied will vary between governance levels and across different types of countries. Given their many possibilities, how IWRM and the nexus approach can be used are best illustrated by cases. Many sources of cases are available, such as those in the GWP's IWRM Toolbox. The nexus cases exhibit attributes of a system-based approach to a range of resource management issues. Like the IWRM cases, they address highly-diverse situations with a central theme of connection of water to other sectors, such as energy, food systems, health, climate, and others and, like IWRM, many sources of cases are available.

The cases show similarities and differences. IWRM is, according to its definition, a management process, while the nexus approach is a systems tool to identify inter-relationships between resource categories. Given this aspect of the nexus approach, it will apply to many instances where IWRM is applied. In that sense, the nexus approach is like a special case of IWRM in situations where a given set of resource sectors is involved.

Whether it involves IWRM or the nexus approach, water will be a core element in the management situation or interaction among resource sectors. How to allocate the leadership role is a subtle nuance between the two paradigms. IWRM will be water-centric and leadership will normally come from the water sector. With IWRM, one set of leaders manages water itself, and another set comprises officials who make decisions about water, but who may be mainly involved with other sectors. Examples include local planners and officials, including regulators. In the nexus approach, the allocation of leadership roles is not evident because it is a shared and cooperative approach to identifying win-win strategies among diverse players. In either paradigm, the core leadership issue is the need to foster cooperation.

In instruction about IWRM and the nexus

approach, no discipline program area will have a monopoly, and many courses can include explanations of how the concepts work in a general way. Disciplinary presentations can be followed by examples and cases from diverse perspectives. Benefits can accrue from imparting broad knowledge about societal problems as well as from building water management capacity with specific tools.

While there has been a great deal of discussion about the shortcomings of IWRM, it is time to move on. Taking the criticism to heart can create an opportunity to explain it as an instrument of change and to utilize knowledge from the nexus approach to clarify it. The underlying concepts will remain complex and difficult to explain, but the need for concepts such as IWRM and the nexus approach will increase with the scale and severity of water issues.

Ultimately, the payoff from application of the concepts will be to improve the total returns to society from management of water and related resources. UCOWR members can take a leading role in explaining them in a range of disciplines.

Acknowledgements

The concepts in this paper evolved over several decades of teaching, research and practice in the field of water resources management. Some of the pioneers of the field also founded UCOWR in 1964 as the Universities Council on Hydrology. One founder, Warren A. Hall, joined Colorado State University after leaving his post as Director of the U.S. Office of Water Resources Research and helped develop the courses and projects that undergird the work described here.

Author Bio and Contact Information

Neil S. Grigg is a professor of civil and environmental engineering at Colorado State University. He has been involved in water resources teaching and research since the early 1970s, mainly at Colorado State University, but also in North Carolina with state government and as director of the Water Resources Research Institute. He has served as River Master of the Pecos River for the U.S. Supreme Court since 1988. His publications include "Integrated Water Resources Management: An Interdisciplinary Approach (Palgrave MacMillan, 2016). He may be contacted at Neil.Grigg@colostate.edu.

References

ABET. 2017. Criteria for Accrediting Engineering Programs, 2017-2018. Available at: http://www.abet.org/accreditation/accreditation-criteria/criteria-for-accrediting-engineering-programs-2017-2018/#objectives. Accessed January 28, 2019.

American Association of Geographers. 2017. Gilbert White. Available at: http://www.aag.org/cs/membership/tributes_memorials/sz/white_gilbert_f. Accessed January 28, 2019.

Benson, D., A.K. Gain, and J.J. Rouillard. 2015. Water governance in a comparative perspective: From IWRM to a 'nexus' approach? *Water Alternatives* 8(1): 756-773.

Bernholz, L. 2014. A messy solution to messy problems. (Book review). Available at: https://ssir.org/book_reviews/entry/a_messy_solution_to_messy_problems. Accessed January 28, 2019.

Biswas, A. 2011. Evolution of Global Developments in Urban Water Management: Part 1 of 2. Available at: http://www.globalwaterforum.org/2011/11/14/evolution-of-global-developments-in-water-management-part-1-of-2/. Accessed January 28, 2019.

Borchardt, D., J. Bogardi, and R.B. Ibisch. (Eds.). 2016. *Integrated Water Resources Management: Concept, Research and Implementation*. Springer, New York.

Boyer, E. 1996. The scholarship of engagement. *Journal of Public Service and Outreach* 1(1): 11-20.

Bradley P. and S. Yee. 2015. Using the DPSIR Framework to Develop a Conceptual Model: Technical Support Document. U.S. Environmental Protection Agency, Washington D.C., EPA/600/R-15/154. Available at: https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=311236. Accessed January 28, 2019.

Braga, B. 2017. Water Security in a Changing World. Available at: http://iwra.org/member/index.php?page=286&abstract_id=4143. Accessed January 28, 2019.

Butterworth, J. 2014. Is There Mileage Left in the IWRM Concept? Or is it Time to Move On? IRC. Available at: https://www.ircwash.org/blog/is-there-mileage-left-in-iwrm. Accessed January 28, 2019.

Campana, M. 2011. Special issue on integrated water resources management: The emperor's new clothes or indispensable process? *Impact American Water Resources Association* 13(3): 3.

Colorado State University. 2017. U.S. Perspective on the Water-Energy-Food Nexus. Colorado Water Institute Information Series No. 116. Available at: http://www.cwi.colostate.edu/workshops/nexus2014/CaseStudies.aspx. Accessed January 28, 2019.

Gari, S.R., A. Newton, and J.D. Icely. 2015. A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. *Ocean & Coastal Management* 103: 63-77.

GIZ and ICLEI (Deutsche Gesellschaft für Internationale Zusammenarbeit GmbH and Local Governments for Sustainability). 2014. Operationalizing the Urban NEXUS: Towards Resource-efficient and Integrated Cities and Metropolitan Regions, GIZ Eschborn. Available at: https://www2.giz.de/wbf/4tDx9kw63gma/UrbanNEXUS_Publication_ICLEI-GIZ_2014_kl.pdf. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017a. What is IWRM? Available at: http://www.gwp.org/en/GWP-CEE/about/why/what-is-iwrm/. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017b. GWP Technical Committee. Available at: http://www.gwp.org/en/About/who/GWP-Technical-Committee/. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017c. Promoting Social Change (C8). Available at: http://www.gwp.org/en/learn/iwrm-toolbox/Management-Instruments/Promoting_Social_Change/. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017d. Welcome to the IWRM ToolBox. Available at: http://www.gwp.org/en/learn/iwrm-toolbox/About_IWRM_ToolBox/. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017e. Bolivia: The Water War to Resist Privatisation of Water in Cochabamba (#157). Available at: http://www.gwp.org/en/learn/KNOWLEDGE_RESOURCES/Case_Studies/Americas--Caribbean/bolivia-the-water-war-to-resist-privatisation-of-water-in-cochabamba-157/. Accessed January 28, 2019.

Global Water Partnership (GWP). 2017f. Transboundary: SWOT Analysis and Challenge of Nile Basin Initiative (#393). Available at: http://www.gwp.org/en/learn/KNOWLEDGE_RESOURCES/Case_Studies/Africa/Transboundary-SWOT-analysis-and-challenge-of-Nile-Basin-Initiative-393/. Accessed January 28, 2019.

GRACE Communications Foundation. 2017. Food, Water and Energy: Know the Nexus. Available at: http://www.gracelinks.org/1802/issue-paper-know-the-nexus. Accessed January 28, 2019.

Grigg, N. 2015. Problem archetypes as common ground for water resources management education. *Water Resources Management* 29(10): 3535-3550.

Grigg, N. 2016. *Integrated Water Resources Management: An Interdisciplinary Approach*. Palgrave Macmillan, London.

Hanlon, P., R. Madel, K. Olson-Sawyer, K. Rabin, and J. Rose. 2013. Food, Water and Energy: Know the Nexus. GRACE Communications Foundation, New York. Available at: http://www.gracelinks.org/media/pdf/nexus_final_final.pdf. Accessed January 28, 2019.

Hassan, Z. 2014. *The Social Labs Revolution: A New Approach to Solving our Most Complex Challenges*. Berrett-Koehler Publishers, Inc., San Francisco, CA.

Hoff, H. 2011. Understanding the Nexus. Background Paper for the Bonn2011 Conference: The Water, Energy and Food Security Nexus, Solutions for the Green Economy. Stockholm Environment Institute, Stockholm. Available at: http://wef-conference.gwsp.org/fileadmin/documents_news/understanding_the_nexus.pdf. Accessed January 28, 2019.

International Center for Integrated Water Resources Management. 2017. About ICIWaRM. Available at: https://www.iwr.usace.army.mil/about/technical-centers/iciwarm-international-center-for-integrated-water/. Accessed January 28, 2019.

Ioris, A. 2008. The limits of integrated water resources management: A case study of Brazil's Paraíba do Sul River Basin. *Sustainability: Science, Practice, and Policy* 4(2): 4-11. Available at: https://doi.org/10.1080/15487733.2008.11908017. Accessed January 28, 2019.

IUCN and IWA. 2017. Nexus Dialogue on Water Infrastructure Solutions. Nexus Case Studies. Available at: http://www.waternexussolutions.org/2q2/solutions/nexus-case-studies.html. Accessed January 28, 2019.

LaFrance, D. 2013. Total water solutions: Caring about every drop, everywhere. *American Water Works Association* 105(8): 6.

LIPHE. 2017. The Nexus between Energy, Food, Land Use, and Water: Application of a Multi-Scale Integrated Approach. Available at: http://www.nexus-assessment.info/. Accessed January 28, 2019.

Martin-Nagle, R., E. Howard, A. Wiltse, and D. Duncan. 2011. Bonn2011 Conference: The Water, Energy and Food Security Nexus. Solutions for the Green

Economy. Conference Synopsis. 16-18 November 2011. Available at: https://www.water-energy-food.org/fileadmin/user_upload/files/documents/bonn2011_nexussynopsis.pdf. Accessed January 28, 2019.

Mirumachi, N. 2015. Wicked Problems, Messy Analysis, Clumsy Solutions? Available at: https://sites.tufts.edu/reinventingpeace/2015/04/23/wicked-problems-messy-analysis-clumsy-solutions/. Accessed January 28, 2019.

Moss, T. 2010. Managing Water Beyond IWRM: From Paradigm to Pragmatism. Presentation to 1st Water Horizon Conference, Berlin. 13-14 July 2010. Erkner, IRS.

Olivera, O. 2004. *¡Cochabamba! Water War In Bolivia.* South End Press, New York, NY.

One Health Initiative. 2017. One Health Initiative will Unite Human and Veterinary Medicine. Available at: http://www.onehealthinitiative.com/. Accessed January 28, 2019.

Ostrom, E. 1990. *Governing the Commons: The Evolution of Institutions for Collective Action*, J. Alt and D. North (Eds.). Cambridge University Press, New York, NY.

Posner, P.L. 2009. The pracademic: An agenda for re-engaging practitioners and academics. *Public Budgeting & Finance* 29(1): 12-26.

Quesada, J., W. Kintsch, and E. Gomez. 2005. Complex Problem Solving: A Field in Search of a Definition? Available at: http://lsa.colorado.edu/papers/manuscriptTIES.3.9.postReview.pdf. Accessed January 28, 2019.

Rasul, G. and B. Sharma. 2016. The nexus approach to water–energy–food security: An option for adaptation to climate change. *Climate Policy* 16(6): 682-702.

Savenije, H.H.G. and A.Y. Hoekstra. 2002. Water resources management. In: *Knowledge for Sustainable Development: An Insight into the Encyclopedia of Life Support Systems*, Vol. II, UNESCO Publishing, Paris, France/EOLSS Publishers, Oxford, pp. 155-180. Available at: https://www.eolss.net/Sample-Chapters/C07/E2-16-00-00.pdf. Accessed January 28, 2019.

Senge, P.M. 1990. *The Fifth Discipline: The Art & Practice of the Learning Organization*. Doubleday Currency, New York.

Stockholm Water Institute. 2019. Improved Water Governance. Available at: https://www.siwi.org/priority-area/water-governance/. Accessed April 10, 2019.

Tortajada, C. and A. Biswas. 2017. The rapidly changing global water management landscape. *International Journal of Water Resources Development* 33(6): 849-852.

Trottier, J. 2008. Water crises: Political construction or physical reality? *Contemporary Politics* 14(2): 197-214.

United Nations Department of Economic and Social Affairs (UNDESA). 2017. Integrated Water Resources Management. Available at: http://www.un.org/waterforlifedecade/iwrm.shtml. Accessed January 28, 2019.

United Nations Secretary-Generals' Advisory Board on Water & Sanitation. 2017. UN Sustainable Development Knowledge Platform: Champion IWRM and the Water-Food-Energy Nexus. Available at: https://sustainabledevelopment.un.org/topics/water/unsgab/iwrmnexus. Accessed January 28, 2019.

UNECE. 2017. Water-Food-Energy-Ecosystem Nexus. Available at: http://www.unece.org/env/water/nexus.html. Accessed January 28, 2019.

UNESCO. 2017. Water-related UNESCO Chairs. Available at: https://en.unesco.org/themes/water-security/chairs. Accessed January 28, 2019. University of Geneva. 2017. Water Resources Management and Policy. Available at: https://www.coursera.org/learn/water-management/lecture/7HVbP/3-3-the-concept-of-integrated-water-resources-management-iwrm. Accessed January 28, 2019.

United Nations University- Flores. 2017. The Nexus Approach to Environmental Resources Management. Available at: https://flores.unu.edu/en/research/nexus. Accessed January 28, 2019.

U.S. Department of Energy. 2014. The Water-Energy Nexus: Challenges and Opportunities. Available at: https://www.energy.gov/under-secretary-science-and-energy/downloads/water-energy-nexus-challenges-and-opportunities. Accessed January 28, 2019.

U.S. National Science Foundation. 2017. CNH_2: Dynamics of Integrated Socio-Environmental Systems. Available at: https://www.nsf.gov/funding/pgm_summ.jsp?pims_id=13681. Accessed January 28, 2019.

U.S.-Pakistan Centers for Advanced Studies in Water. 2017. Integrated Water Resources Management. Available at: http://water.muet.edu.pk/degree-programs/integrated-water-resources-management/. Accessed January 28, 2019.

Vijay, V., S.K. Yadav, B. Adhikari, H. Seshadri, and D.K. Fulwani. (Eds.). 2014. *Systems Thinking Approach for Social Problems*, Lecture Notes in Electrical Engineering 327. In: Proceedings of 37th National Systems Conference, Springer, New York. Available at: DOI 10.1007/978-81-322-2141-8. Accessed January 28, 2019.

Water, Energy & Food Security Resource Platform. 2017a. The Nexus Approach vs. IWRM- Gaining Conceptual Clarity. Available at: https://www.water-energy-food.org/news/2014-03-27-nexus-concept-the-nexus-approach-vs-iwrm-gaining-conceptual-clarity/. Accessed January 28, 2019.

Water, Energy & Food Security Resource Platform. 2017b. Implementation and Case Studies. Available at: https://www.water-energy-food.org/resources/implementation-and-case-studies/. Accessed January 28, 2019.

World Business Council for Sustainable Development. 2017. Water-Smart Agriculture. Available at: http://www.wbcsd.org/Clusters/Water/Water-smart-Agriculture. Accessed January 28, 2019.

World Water Council. 2017a. Water Crisis: Toward a Way to Improve the Situation. Available at: http://www.worldwatercouncil.org/en/water-crisis. Accessed January 28, 2019.

World Water Council. 2017b. Integrated Water Resources Management. Available at: http://www.worldwatercouncil.org/en/integrated-water-resources-management. Accessed January 28, 2019.

Ziegler, J.A. 1994. *Experimentalism and Institutional Change: An Approach to the Study and Improvement of Institutions*. University Press of America, Lanham, MD.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 35-45, April 2019

Toward a Better Understanding of Recurrence Intervals, Bankfull, and Their Importance

*Pamela J. Edwards[1], Edward A. Watson[2], and Frederica Wood[1]

[1]USDA Forest Service, Northern Research Station, [2]formerly, Downstream Strategies, LLC
**Corresponding Author*

Abstract: Bankfull is a concept that is intimately tied to the annual flood series through the well-accepted tenet that bankfull discharge occurs at approximately the 1.5-year recurrence interval on the annual series. Thus, due to this association the annual series provides a useful diagnostic tool for helping to identify the bankfull elevation in the field. The partial-duration series does not provide an equivalent tool because paired discharge and recurrence interval values from the flood frequency curve depend upon the minimum threshold selected for developing the partial-duration series. However, the interpretation that bankfull discharge occurs on average once every 1.5 years, or two out of every three years from that bankfull discharge/recurrence interval relationship on the annual series is incorrect. Frequencies of small floods (those with recurrence intervals ≤10 years) should be obtained using the partial-duration flood series because it contains a more accurate representation of the size and frequency of small events. We used discharge data from 11 streams in West Virginia watersheds that ranged from about 0.14 to 223 km^2 to compare the two series and to illustrate the variability in small flood frequencies through time. Flooding to the bankfull stage was absent some years but occurred as many as four or five times during other years.

Keywords: *annual flood series, partial-duration series, flood frequency, bankfull, small floods*

Flood recurrence is an important hydrologic concept from science, policy, management, and social perspectives. Recurrence intervals are used in a myriad of applications, including natural stream design, municipal zoning and planning, flood prediction, and insurance and actuarial purposes, to name just a few. Often interest in flood recurrence intervals is more focused on the more extreme, lower probability events (e.g., 100-year flood), as these typically are more catastrophic and receive substantial media coverage. However, small flood events are also important because they occur much more frequently. In particular, bankfull floods are very important because they are the most effective at changing channel shape and characteristics, and thus been given the title "dominant channel-forming flow" (Wolman and Miller 1960; Dunne and Leopold 1978; Copeland et al. 2000).

Initially, the concepts of bankfull discharge and recurrence intervals appear to be reasonably straight forward and simple. However, students and practitioners of hydrologic sciences often have an incomplete and sometimes incorrect understanding of one or both concepts. In this paper, we attempt to provide a fuller understanding of these concepts, including their identification or development, interpretation, and use in field applications. We explore the conundrum of and confusion that result from bankfull discharge being defined in terms of recurrence intervals derived from the annual flood series while the partial-duration series is recommended for the accurate determination of small flood recurrence intervals or flood frequencies.

Concepts

Bankfull and Bankfull Discharge

The term bankfull is used commonly to describe both a position on the stream or river bank that approximates the stage at which water overflows onto the floodplain as well as the specific discharge present when the water surface is at bankfull. For clarity in this paper, we use the term bankfull to reference the position or associated stage, and the term bankfull discharge to describe the flow rate (e.g., $m^3 s^{-1}$) at that stage.

To ensure correct estimates of bankfull and bankfull discharge, such as for natural stream design, both metrics should be determined from field observations. Bankfull should be determined along a reach (vs. a single location) using characteristics that are appropriate for that type of channel, and the characteristics should be verified using a reference reach. These include a variety of features, such as the mean elevation of the top of channel bars, the lower edge of perennial vegetation, the top of the streambank, and the highest scour line (Williams 1978; Wiley et al. 2002), with the specific bankfull-defining features in part depending on the type of channel (e.g., alluvial, presence or absence of a developed floodplain, etc.). There are a number of sources, such as Harrelson et al. (1994), Leopold et al. (1995), Wolman et al. (2003), and Verry (2005), that provide detailed instruction for identifying bankfull in various regions or conditions.

Bankfull discharge is unique to each stream or river, depending upon several factors, including size of the waterway and contributing area, underlying geology, channel geometry, and physiographic region. Consequently, bankfull discharge can range from very small values (e.g., less than 1 $m^3 s^{-1}$) to thousands of $m^3 s^{-1}$. Estimating bankfull discharge is a relatively straight forward task for streams and rivers that are gauged: determine bankfull, and then use the stream discharge records to identify the stage or flow associated with the bankfull position and confirm that it corresponds with a recurrence interval near 1.5 years on the annual series. If hydrologic records include only river stage, discharge for the site must be determined by other procedures such as Manning's equation.

In a given geographical region, the best predictor of bankfull discharge and hydraulic geometry is drainage area. Regional curves can be empirically determined to relate drainage area to bankfull discharge, as well as cross-sectional area, width, and mean depth. Regional curves are valuable for use when no gauging station is present on a stream or river. A regional curve is produced by identifying potential bankfull features at multiple gauging stations in that region and then using the annual series to verify that the features are associated with a flood recurrence of approximately 1.5 years. The curves can be refined with greater numbers and broader distribution of those gauges across the region. The more refined the regional curves, the better they are for validating bankfull features on other ungauged streams within the region. The U.S. Geological Survey (USGS) has taken a lead in developing and publishing regional curves throughout the United States. For a more in-depth discussion of their value and application see Dunne and Leopold (1978), particularly pages 15-17.

Recurrence Intervals

Recurrence intervals describe the frequency, on average, at which specific types of events occur. In hydrologic sciences, recurrence intervals can be developed for streamflow or precipitation. In this paper, we focus only on flow.

Recurrence intervals are calculated from the equation:

$$T = (n+1)/m \qquad \text{(Equation 1)}$$

where T = recurrence interval, n = number of observations, and m = rank of each observation, with observations ranked in descending order (Dunne and Leopold 1978). The observations are peakflows (e.g., $m^3 s^{-1}$), which can be from either the annual flood series or the partial-duration flood series (described below). Plotting the recurrence interval on the X-axis and the associated instantaneous peakflow value on the Y-axis (typically using graph papers with special distributions, such as Log Pearson Type III, Pearson Type IV, Gumbel Type I, Gumbel Type III semi-logarithmic or double logarithmic, generalized Pareto [Benson 1968; Dunne and Leopold 1978; Keast and Ellison 2013]) and then fitting a smooth line to the plotted data produces the flood frequency curve.

Annual vs. Partial-Duration Series

As alluded to earlier, there are two series of discharge data from which recurrence intervals can be derived: the annual series and the partial-duration series. Both use similar procedures for calculating recurrence intervals but the flow data in each series differ. The annual series builds the resulting flood frequency curve from the single maximum instantaneous peakflow that occurs each year for the stream of interest, while the partial-duration series employs all the single-storm instantaneous peakflows that equal or exceed some minimum threshold (i.e., a low-end high flow) for the stream. Therefore, the partial-duration series contains the annual series as well as additional data, with most of the additional data in the partial-duration series being from smaller flood events and events with less than bankfull discharge. For equation 1 to hold, the peakflows included in the partial-duration series must be temporally independent (Beguería 2005). If peaks occur so closely in time that they are not independent, the greater peakflow is the one included in the partial-duration series (Dunne and Leopold 1978).

Flood frequency curves for the annual series and the partial-duration series converge at or before the 10-year recurrence interval (Langbein 1949; Dunne and Leopold 1978; Keast and Ellison 2013). In other words for the typical duration of records, the data pairs and graphical response are very similar for the two series for events that have recurrence intervals >10 years, but they differ between the two series for recurrence intervals <10 years – the latter being the most common flood events.

This divergence of the two series raises the question: "which data series should be used to develop flood frequency curves for small floods (i.e., those with recurrence intervals <10 years)?" From purely a mathematical perspective, the answer is the partial-duration series, but for field practitioners the answer depends on the use. The partial-duration series provides a more accurate depiction of the relationship between small flood flows and their recurrence intervals or frequencies, which is described in further detail below. But from the perspective of helping to confirm the bankfull position identified in the field, the annual series serves as a diagnostic tool in a way that the partial-duration series cannot. Here is why. Based on data from many studies (e.g., Dury et al. 1963; Leopold et al. 1964; Hickin 1968; Leopold 1994), the statement "bankfull discharge from most rivers has a recurrence interval on the annual flood series of 1.5 years" (Dunne and Leopold 1978, page 315) is a well-accepted hydrologic tenet. Individual streams often show some variation in this value, but 1.5 is typically a good approximation regionally (e.g., see Castro and Jackson 2007). Therefore, once a flood frequency curve is developed from the annual series, the discharge associated with the 1.5-year recurrence interval can be used for most streams and rivers to help confirm or fine tune the position of bankfull in the field. From a practical standpoint there are few gauged streams, but even fewer gauged with equipment that provide continuous streamflow measurement to allow identification of individual storm peakflows (i.e., instantaneous peakflows) throughout the year, over multiple years as required for development of the partial-duration series. The maximum annual peakflow datasets are more readily available, which may be why the discovery of the relationship between recurrence interval and bankfull discharge was developed and reported from annual series curves.

The partial-duration series does not provide this same diagnostic capability. This is because the recurrence interval determined from the partial-duration series depends upon the selection of the minimum threshold used to define which instantaneous peaks are included in the partial-duration series dataset. Raising or lowering that minimum will change the associated recurrence interval (T in eq. 1) because the number of events and rank values (respectively, n and m in eq. 1) will change. By including only the highest instantaneous value in each year, the annual series avoids the subjectivity in defining the minimum threshold and the associated variability in recurrence intervals for bankfull discharge (and other small flood events) that results.

The question about which series is appropriate for specific purposes is made even more confusing by a common misinterpretation of flood frequency recurrence intervals derived from the bankfull discharge recurrence definition. The subsequent sentence from Dunne and Leopold (1978, page 315) states – "This means that 1 year out of 1.5 or 2 years out of 3, the highest discharge for the year will be equal to or will exceed the bankfull capacity of the channel." Nearly identical, though sometimes simpler pronouncements are found elsewhere in landmark hydrologic literature (e.g., Leopold et al. 1964). Unfortunately this interpretation is incorrect, even though it is still commonly repeated by practitioners. The error in interpretation stems from the composition of the annual flood series. To estimate recurrence intervals or flood frequencies (or probability, which is the inverse of recurrence interval) accurately, the dataset must include a sufficient number of data points and sufficient duration of measurements to adequately represent the true frequency of different sized flood events. The annual series fails to represent the frequency of small floods (<10-year recurrence intervals) due to the limited amount of data (single highest discharge per year) included in the annual series. Consequently, even though bankfull discharge is *associated with* approximately the 1.5-year recurrence interval on the annual series, the actual recurrence interval or flood frequency of bankfull discharge is generally underestimated by the annual series (Armstrong et al. 2012). In other words, events that equal or exceed bankfull discharge typically occur more frequently than once out of 1.5 years or two out of three years on average, and often much more frequently.

The requirement for adequate representation of flood frequencies is the reason that the partial-duration series is better suited for flood frequency analysis, especially of small events. Establishment of the minimum threshold for the development of the partial-duration series is somewhat arbitrary, but the minimum instantaneous peakflow value from the annual series is a common recommendation for use as the threshold (Dunne and Leopold 1978). With a sufficiently long record (at least 10 years), use of that threshold typically provides a robust estimate of the minimum annual peak that might be expected for a stream or river within expected climate and runoff conditions.

Flood Frequency Analysis Using Data

To examine the frequency of small floods and illustrate that the interpretation of bankfull discharge frequency is incorrect using the annual series results, we compared bankfull discharge frequency results from annual and partial-duration series data for 11 streams within West Virginia. The corresponding watersheds ranged in size from about 0.14 to 223 km^2 (Table 1). The four smallest of these are located in the Fernow Experimental Forest (FEF) (https://www.nrs.fs.fed.us/ef/locations/wv/fernow/data/), which is administered by the U.S. Forest Service's Northern Research Station. The remaining streams are gauged by the USGS (webpage: USGS Surface-Water Historical Instantaneous Data for West Virginia: Build Time Series; https://waterdata.usgs.gov/wv/nwis/uv/?referred_module=sw). FEF data were collected continuously while USGS data were collected on 15-, 30-, or 60-minute time steps. Peakflows were determined for each individual storm. FEF and USGS data span the periods shown in Table 1. The USGS data include only years for which non-provisional data were available for each stream (Table 1).

Individual storm hydrographs were identified from the FEF and USGS data files by projecting a line with a slope of 0.0005 m^3 s^{-1} km^{-2} (0.05 ft^3 s^{-1} mi^{-2}) per decimal hour from the point where each storm hydrograph began to rise through the point where that line intersected the receding limb of the hydrograph (Hewlett and Hibbert 1967; Harr et al. 1975; Dunne and Leopold 1978; Dingman 2002; Blume et al. 2007). Once all individual storm hydrographs were identified for each watershed across the available time series, the instantaneous peakflow (m^3 s^{-1}) for each storm (or snowmelt) event was identified. From these, the annual maximum instantaneous peakflow for each waterway was identified for each year of record to develop its annual series. The overall largest instantaneous peakflow for the period of record for each stream is given in the maximum peakflow column in Table 1. The minimum instantaneous peakflow value in the annual series (minimum peakflow column, Table 1) was used as the threshold for the associated partial-duration series (Dunne and Leopold 1978).

The recurrence interval was calculated using equation 1 for each peakflow value in each annual series, and flood frequency graphs were developed from those results. For each of the FEF streams, there was one flood, which was the flood of record, that was well outside the population of

Table 1. Streams and rivers used in the annual series and partial-duration (PD) series analysis. RI = recurrence interval. The minimum annual peakflow for each waterway was used as its threshold for the partial-duration series.

Waterway	Drainage area (km^2)	Years gauged	Number of years	Maximum peakflow from annual series (m^3 s^{-1})	Minimum peakflow from annual series (m^3 s^{-1})	Flow at 1.5-yr RI from annual series (m^3 s^{-1})	RI from PD series associated with annual 1.5-yr RI (yr)
Fernow WS13	0.14	1989-2016	28	0.282	0.035	0.084	4.63
Fernow WS10	0.15	1985-2016	32	0.279	0.030	0.068	3.98
Fernow WS4	0.39	1952-2016	65	0.72	0.077	0.157	3.75
Fernow WS14	1.32	1994-2016	23	1.96	0.345	0.56	2.84
Sand Run (USGS 03052500)	37.0	1998-2017	20	84.7	11.3	19.0	2.37
Panther Creek (USGS 03213500)	80.3	2003-2017	15	162.5	13.4	37.6	3.19
East Fork Twelvepole Creek (USGS 03206600)	98.2	1997-2017	21	214.4	11.7	38.5	4.12
Peters Creek (USGS 03191500)	104.1	2004-2017	14	214.4	20.0	35.8	2.67
Piney Creek (USGS 03185000)	136.5	2003-2017	15	85.8	17.2	32.8	2.95
Shavers Fork River (USGS 03067510)	155.9	2001-2017	17	300.2	73.9	122.8	2.76
Blackwater River (USGS 03066000)	222.5	1997-2017	21	117.2	34.3	57.5	4.80

the remaining instantaneous peakflow values. Each of those extreme values was included in the rankings and recurrence interval calculations, but as recommended by Dalrymple (1960) those extreme values were not used for fitting the flood frequency curves.

The values in the second to last column in Table 1 are the discharges (m^3 s^{-1}) associated with the 1.5-year recurrence interval on the annual series, or bankfull discharge, for purposes of illustration. Each of the bankfull discharge values from the annual series then was applied to the flood frequency curves developed from the partial-duration series to determine the corresponding recurrence intervals for each stream for the partial series. Those recurrence intervals from the partial-duration series are all larger than those from the annual series (last column in Table 1), which is expected since the partial-duration series contains more flood events than the annual series.

The numbers of events included in the partial-duration series (i.e., those that were above the minimum threshold) for the watersheds are shown in Table 2. The peakflows identified as being above the minimum threshold within each watershed were found to be independent using the Durbin-Watson test for autocorrelation (SAS Institute Inc. 2013). Consequently, all peakflows above the

Table 2. Metrics associated with the partial-duration (PD) series. RI = recurrence interval.

Waterway	Number of years	Total number of events with peakflow ≥ PD threshold	Number of events with peakflow ≥ annual 1.5-yr RI	Mean number of events/year ≥ annual 1.5-yr RI
Fernow WS13	28	167	35	1.25
Fernow WS10	32	170	40	1.25
Fernow WS4	65	274	68	1.05
Fernow WS14	23	95	30	1.30
Sand Run (USGS 03052500)	20	58	25	1.25
Panther Creek (USGS 03213500)	15	51	13	0.87
East Fork Twelvepole Creek (USGS 03206600)	21	104	25	1.19
Peters Creek (USGS 03191500)	14	54	23	1.64
Piney Creek (USGS 03185000)	15	68	25	1.67
Shavers Fork River (USGS 03067510)	17	52	20	1.18
Blackwater River (USGS 03066000)	21	88	18	0.86

threshold were retained in the final partial-duration series dataset.

One-fifth to just under half of those events, depending on the stream/river, had instantaneous peakflows that equaled or exceeded the bankfull discharge associated with the annual 1.5-year recurrence interval on the annual series (Table 2, Events with flow ≥ annual RI 1.5 column). The mean number of events per year (Table 2) confirms that the frequency of events for which at least bankfull discharge occurred exceeds the average frequency of once every 1.5 years (or 0.666). For most of these waterways, floods with peakflows that equaled or exceeded bankfull discharge occurred, on average, at least twice that frequently. However, those values represent only the averages and every year is unique. Years with no flood events or only one flood did occur, as did years with multiple events (Figure 1). Indeed, 9 of the 11 waterways had at least one year with four or five flood events, and that flood frequency was observed even for shorter-duration streamflow records. That all 11 channels had at least a single year with no bankfull discharge (Figure 1), indicates that using the minimum annual value for the threshold provided robust datasets of low-end high flow data for examining small flood frequencies.

Discussion

For illustrative purposes, the concepts and analyses presented in this paper were framed in terms of the accepted tenet that the bankfull discharge recurrence interval is at 1.5 years on the annual series. However, we fully recognize that

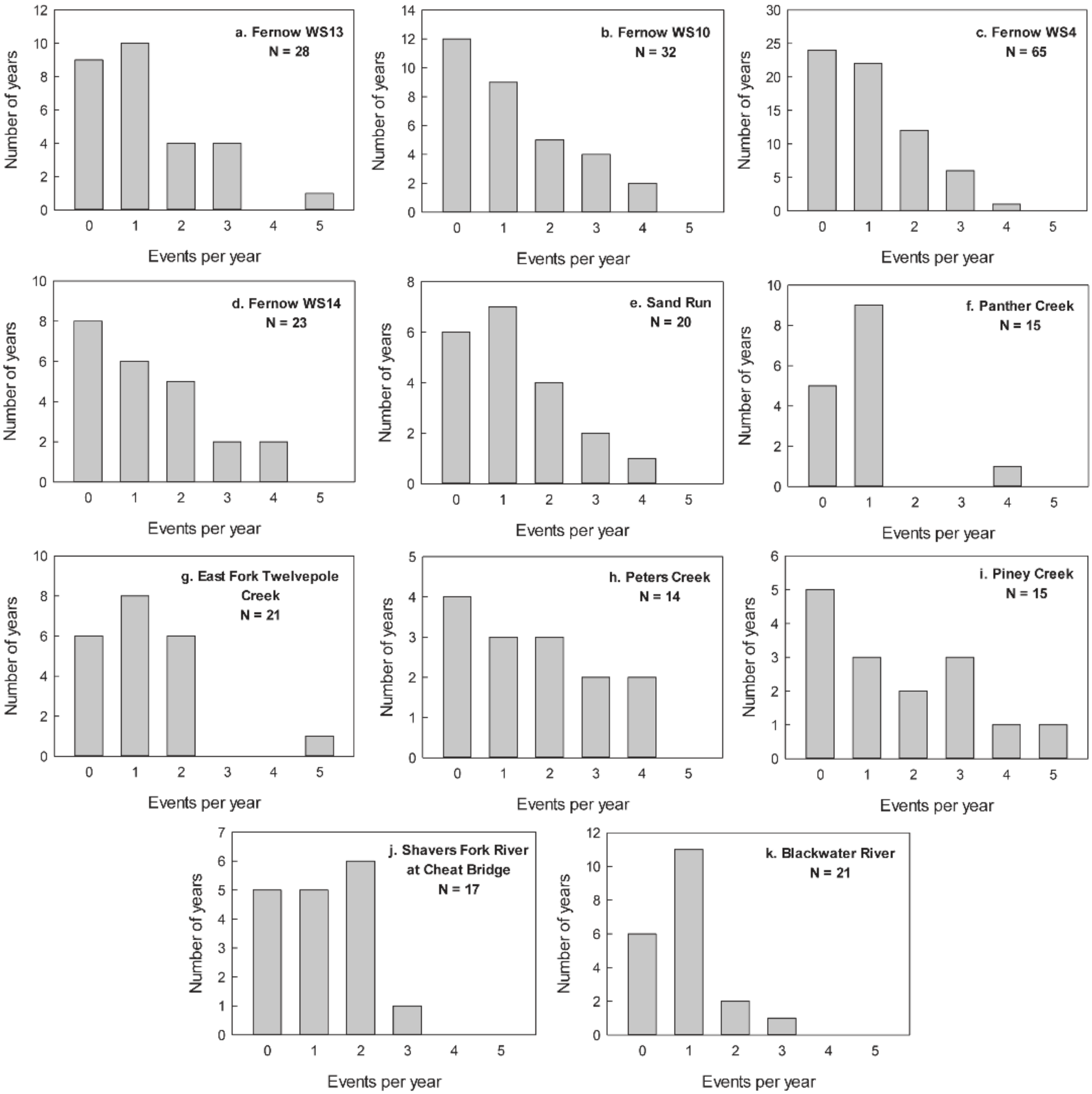


Figure 1. Frequency that each number of events per year with peakflows equal or exceeding bankfull discharge (based on 1.5-year recurrence interval) occurred for each of the watersheds. N refers to the number of years of record included in the analysis. See Table 1 for the specific years included.

there is variation among streams in the bankfull recurrence interval. Most values reported in the literature appear to fall somewhere within the 1- to 4-year recurrence interval on the annual series (e.g., Williams 1978; Andrews 1980; Petit and Pauquet 1997; Castro and Jackson 2007; Ahilan et al. 2013), but some streams have bankfull discharge recurrence intervals reported to be as high as a few decades (Williams 1978; Ahilan et al. 2013). In practice it is necessary to collect sufficient data and make thorough observations for streams in the region of interest to more accurately estimate and confirm bankfull in the field. Throughout much of West Virginia we have found that bankfull often is associated with a 1.3-year recurrence interval on the annual series, rather than 1.5 years; consequently, where appropriate we use the 1.3-year recurrence interval and associated discharge in fluvial applications. In all situations, bankfull should be determined locally from field conditions, and the associated discharge should be determined before proceeding with any type of action or assessment.

As noted previously, the annual series provides a useful diagnostic tool for estimating and confirming bankfull, while the frequency of bankfull discharges or other small floods should be determined from the partial-duration series. It is incorrect to describe bankfull discharge as the event that occurs only two out of every three years (or once every 1.5 years), even though this remains a commonly held and repeated interpretation and definition (e.g., Rosgen 1994, 1996; Harman and Jennings 1999; Vermont Agency of Natural Resources 2004; Mulvihill et al. 2009), largely due to this original misinterpretation in several important, early hydrology treatises (Leopold et al. 1964; Dunne and Leopold 1978) that otherwise provided indispensable information. However, for most waterways, a flood that occurs only two out of every three years is much bigger than the true bankfull flood.

Fundamentally, bankfull discharge is independent of the series from which it is associated or determined; bankfull discharge is whatever it is for the stream or river of interest – only the accurate estimation of flood frequency depends upon flood series. Because hydrologists and fluvial geomorphologists involved in natural stream design develop channel designs based on bankfull discharge and not flood frequency, there is little chance that errors in design dimensions will result simply from using the wrong series. That said, the authors have had experience with a regulator whose metric of an approved design was based on requiring a specified flood frequency (three floods per year). While flood frequency and bankfull discharge are related on the partial-duration series, we have shown that there is substantial variability in the frequency from year to year (Figure 1). Therefore, there is risk in predicating channel design on a required number of floods per year, rather than an average number per year (based on the flood frequency curve from the partial-duration series). The former channel would likely have a much smaller width and depth, and be able to convey less water than a stable channel in order to ensure flooding a predefined number of times per year, including during years when no bankfull events would have occurred.

Eventually such undersized channels will re-adjust and develop larger and more stable width and depth dimensions, but during the period of re-adjustment the location of the channel may move laterally within the floodplain. This is because in a channel, bankfull discharge has the power to move a certain amount of sediment and a certain maximum particle size, and stream reaches do not exist in isolation and are influenced by both upstream and downstream conditions. Bedload delivery from upstream, where channel dimensions are not undersized, will fill and clog the smaller, re-designed reach since it is too small to transport the full volume of water and bedload. The energy of the water will cut around the reach in areas of the bank that are less resistant to eroding than the bedload-choked channel. Eventually a channel will develop that has width and depth dimensions and other energy-controlling attributes that are appropriate for the true bankfull discharge. Unfortunately from the human perspective the position of the new channel may be less desirable than the original position.

Undesirable outcomes also result when channels are intentionally manipulated to reduce the frequency of flooding. These actions disregard, often due to ignorance, the processes to which all channels are subject in their continued evolution to maintain or return to dynamically stable conditions. Reducing the frequency of flooding usually takes the form of treatments that increase in-channel water storage; thus, overflow onto the floodplain occurs less frequently than it would naturally. Actions aimed at reducing flooding include dredging, flood wall construction, and other similar types of flood containment.

Most treatments aimed at increasing storage are focused primarily on deepening the channel because surface landowners are sensitive to losing acreage. A channel that is deepened below its natural bankfull depth is considered disconnected from its floodplain – which is actually the desired effect of dredging. However, disconnection from the floodplain results in drier floodplain soils, which can significantly affect floodplain-dependent land uses such as agricultural operations. A lower channel bed also can deplete groundwater reserves; more of the aquifer is intercepted by the channel, allowing emergent flow to leave the watershed quickly as concentrated streamflow rather than remaining in the aquifer. Lowering the water table further disconnects groundwater from floodplain soils, thereby exacerbating droughty conditions.

Channel widening is sometimes included as part of flood control operations. Unintended effects of widening include intensifying low-flow conditions. In an over-widened channel, low flows are spread over a wider distance, making them shallower than they would be in a more-stable channel configuration. This condition often results in disconnected refugia in which aquatic organisms are stranded in small pools where food, oxygen, suitable temperatures, and cover may be limited, exceeding tolerances for organism survival.

Regardless of the technique used to increase water storage (dredging, flood walls, etc.), during high flows the water's energy continues to build within the channel, exceeding the maximum energy of true bankfull because the flow cannot spill onto the floodplain. As the energy of the water builds with increasing volume, the shear stress likewise increases, leading to channel scour, erosion of the floodplain once flooding begins (which can include lateral channel migration and re-alignment elsewhere on the floodplain), and the transport and deposition within and outside the channel of more sediment, as well as more and larger-sized bedload.

Conclusion

The annual flood series, while extremely useful as a diagnostic tool for identifying and/or confirming bankfull discharge, is misleading when used to quantify the frequency of high probability events (i.e., small floods). Even some practitioners of hydrology do not fully understand the differences, applications, and interpretations of the annual and partial-duration series. It is extremely important for these series to be taught comprehensively so their uses are fully understood. It is important to understand that floods are natural events that do and should occur frequently and floodplains are an integral part of every river system. This understanding is critical to protecting water resources and aquatic health, as well as for protecting human lives and making informed decisions for watershed planning and management.

Author Bio and Contact Information

Pamela J. Edwards (corresponding author) is a retired research hydrologist with the USDA Forest Service, Northern Research Station in Parsons, WV. Her research focused on the effects of forest management on water quality, streamflow and stream stability, and the development and effectiveness of best management practices for protecting water, riparian, and aquatic resources. She can be reached at h2ologicwv@gmail.com.

Edward A. Watson is a hydrologist in Davis, WV. He has practiced stream design and construction throughout the Mid-Atlantic region for 14 years, and has worked in watershed restoration for more than 20 years. He can be reached at edwardawatson@gmail.com.

Frederica Wood is a Natural Resources Specialist with the USDA Forest Service, Northern Research Station in Parsons, WV. She has worked extensively with watershed scientists and hydrologists over her 31 years with the agency in topics related to streamflow and water quality. She can be reached at frederica.wood@usda.gov.

References

Ahilan, S., J.J. O'Sullivan, M. Bruen, N. Brauders, and D. Healy. 2013. Bankfull discharge and recurrence intervals in Irish rivers. Proceedings of the Institution of Civil Engineers. *Water Management* 166(7): 381-393.

Andrews, E.D. 1980. Effective and bankfull discharges of streams in the Yampa River Basin, Colorado and

Wyoming. *Journal of Hydrology* 46(3): 311-330.

Armstrong, W.H., M.J. Collins, and N.P. Snyder. 2012. Increased frequency of low-magnitude floods in New England. *Journal of the American Water Resources Association* 48(2): 306-320.

Beguería, S. 2005. Uncertainties in partial duration series modelling of extremes related to the choice of the threshold value. *Journal of Hydrology* 303(1-4): 215-230.

Benson, M.A. 1968. Uniform flood-frequency estimating methods for federal agencies. *Water Resources Research* 4(5): 891-908.

Blume, T., E. Zehe, and A. Bronstert. 2007. Rainfall-runoff response, event-based runoff coefficients and hydrograph separation. *Hydrological Sciences Journal* 52(5): 843-862.

Castro, J.M. and P.L. Jackson. 2007. Bankfull discharge recurrence intervals and regional hydraulic geometry relationships: Patterns in the Pacific Northwest, USA. *Journal of the American Water Resources Association* 37(5): 1249-1262.

Copeland, R.R., D.S. Biedenharn, and J.C. Fischenich. 2000. Channel-forming Discharge. US Army Corps of Engineers. ERDC/CHL CHETN-VII-5. Available at: http://www.spa.usace.army.mil/Portals/16/docs/civilworks/regulatory/Stream%20Information%20and%20Management/ERDC%20Channel%20Forming%20Discharge.pdf. Accessed March 21, 2019.

Dalrymple, T. 1960. *Flood Frequency Analysis. Manual of Hydrology: Part 3. Flood-flow Techniques*. US Geological Survey Water Supply Paper 1543-A. Available at: https://doi.org/10.3133/wsp1543A. Accessed February 11, 2019.

Dingman, S.L. 2002. *Physical Hydrology, Second Edition*. Waveland Press, Inc., Long Grove, Illinois.

Dunne, T. and L.B. Leopold. 1978. *Water in Environmental Planning*. W.H. Freeman and Company, San Francisco, California.

Dury, G.H., J.R. Hails, and H.B. Robbie. 1963. Bankfull discharge and the magnitude frequency series. *Australian Journal of Science* 26: 123-124.

Harman, W.A. and G.D. Jennings. 1999. River Course—Application of the Rosgen Stream Classification System to North Carolina. North Carolina State Cooperative Extension Service. Available at: https://content.ces.ncsu.edu/application-of-the-rosgen-stream-classification-to-north-carolina. Accessed February 11, 2019.

Harr, R.D., W.C. Harper, J.T. Krygier, and F.S. Hsieh. 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range. *Water Resources Research* 11(3): 436-444.

Harrelson, C.C, C.L. Rawlins, and J.P. Potyondy. 1994. *Stream Channel Reference Sites: An Illustrated Guide to Field Technique*. USDA Forest Service, Rocky Mountain Forest and Range Experiment Station. General Technical Report RM-245.

Hewlett, J.D. and A.R. Hibbert. 1967. Factors affecting the response of small watersheds to precipitation in humid areas. In: *Forest Hydrology*, W.E. Sopper and H.W. Lull (Eds.). Pergamon Press, New York. pp. 275-290.

Hickin, E.J. 1968. Channel morphology, bankfull stage and bankfull discharge of streams near Sydney. *Australian Journal of Science* 30(7): 274-275.

Keast, D. and J. Ellison. 2013. Magnitude frequency analysis of small floods using the annual and partial series. *Water* 5(4): 1816-1829.

Langbein, W.B. 1949. Annual floods and the partial-duration flood series. *Transactions of the American Geophysical Union* 30(6): 879-881.

Leopold, L.B. 1994. *A View of the River*. Harvard University Press, Cambridge, Massachusetts.

Leopold, L.B., W.E. Emmett, H.L. Silvey, and D.L. Rosgen. 1995. *A Guide for Field Identification of Bankfull Stage in the Western United States* (video). USDA Forest Service, Rocky Mountain Forest and Range Experiment Station, Stream Systems Technology Center. Fort Collins, Colorado.

Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. *Fluvial Processes in Geomorphology*. W.H. Freeman and Company, San Francisco, California.

Mulvihill, C.I., B.P. Baldigo, S.J. Miller, D. DeKoskie, and J. DuBois. 2009. *Bankfull Discharge and Channel Characteristics of Streams in New York State*. US Geological Survey. Scientific Investigations Report 2009-5144.

Petit, F. and A. Pauquet. 1997. Bankfull discharge recurrence interval in gravel-bed rivers. *Earth Surface Processes and Landforms* 22(7): 685-693.

Rosgen, D.L. 1994. A classification of natural rivers. *Catena* 22(3): 169-199.

Rosgen, D.L. 1996. *Applied River Morphology*. Wildland Hydrology Books, Pagosa Springs, Colorado.

SAS Institute Inc. 2013. *SAS/ETS® 12.3 User's Guide*. SAS Institute Inc., Cary, North Carolina.

Vermont Agency of Natural Resources. 2004. Vermont Stream Geomorphic Assessment: Appendix K, Identification of Bankfull Stage. Available at:

https://dec.vermont.gov/sites/dec/files/wsm/rivers/docs/assessment-protocol-appendices/K-Appendix-K-04-Identification-of-Bankfull-Stage.pdf. Accessed February 11, 2019.

Verry, E.S. 2005. *A Guide to Identification of Bankfull Stage in the Northeastern United States* (video). USDA Forest Service, Stream Systems Technology Center. Fort Collins, Colorado.

Wiley, J.B., J.T. Atkins, Jr., and D.A. Newell. 2002. *Estimating the Magnitude of Annual Peak Discharges with Recurrence Intervals Between 1.1 and 3.0 Years for Rural, Unregulated Streams in West Virginia*. US Geological Survey. Water-Resources Investigations Report 02-4164.

Williams, G.P. 1978. Bank-full discharge of rivers. *Water Resources Research* 14(6): 1141-1154.

Wolman, M.G., W.E. Emmett, E.S. Verry, D.A. Marion, L.W. Swift, and G.B. Kappesser. 2003. *Identifying Bankfull Stage in Forested Streams in the Eastern United States* (video). USDA Forest Service, Stream Systems Technology Center. Fort Collins, Colorado.

Wolman, M.G. and J.P. Miller. 1960. Magnitude and frequency of forces in geomorphic processes. *The Journal of Geology* 68(1): 54-74.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 46-56, April 2019

Water Scavenging from Roadside Springs in Appalachia

*Leigh-Anne Krometis[1], Hannah Patton[2], Austin Wozniak[3], Emily Sarver[4]

[1]*Associate Professor, Biological Systems Engineering, Virginia Tech, Blacksburg, VA*
[2]*Graduate Research Assistant, Biological Systems Engineering, Virginia Tech, Blacksburg, VA*
[3]*Undergraduate Research Assistant, Biological Systems Engineering, Virginia Tech, Blacksburg, VA*
[4]*Associate Professor, Mining and Minerals Engineering, Virginia Tech, Blacksburg, VA*
**Corresponding author*

Abstract: Significant challenges in the provision of safe drinking water and appropriate, effective sanitation remain in the United States, particularly among communities with few financial resources and/or situated in challenging terrain. Though previous formal research is limited, anecdotal reports suggest that some households in Appalachia may rely on untreated, unregulated roadside "springs" as a primary source of potable water. This effort monitored the water quality at twenty-one of these springs in Central Appalachia and identified potential motivations for this behavior through volunteer surveys in order to better define community challenges and to establish communication for future outreach. The majority (>80%) of spring samples collected were positive for *E. coli,* indicating a potential risk of exposure to waterborne pathogens; measured concentrations of metals and nutrients were generally in accordance with USEPA recommendations for drinking water. Survey respondents generally had a piped source of in-home water available yet primarily collected the water due to "taste" and "quality/health" and used it directly for drinking. Multiple respondents included extra written information indicating that they either did not trust their in-home water source or considered it unreliable. Collectively these results suggest that these roadside springs do serve as a regular source of household water for some communities though they generally do not meet federal drinking water standards. Future efforts are encouraged to work with local municipal water authorities to rebuild community trust and/or to determine whether on-site treatment at these springs is practicable.

Keywords: *drinking water quality, springs, rural health, environmental health*

The United States currently reports near 100% access to drinking water but there is increasing recognition that significant issues of water quality and equity remain unsolved (World Bank 2015). Recent high profile failures in municipal safe drinking water systems (e.g., Flint, MI and Charleston, WV) (Katner et al. 2016; Thomasson et al. 2017) have drawn attention to the vulnerability of populations reliant on aging infrastructure and/or systems with limited financial resources. Systematic analyses of drinking water quality violations reported to the USEPA under the Safe Drinking Water Act (SDWA) have also revealed potential issues of environmental justice. Most recently, a 2017 national analysis of municipal systems serving more than 10,000 homes indicated that the prevalence of health-based drinking water violations was significantly correlated to both race/ethnicity and poverty, i.e., poorer communities with higher numbers of black or Hispanic residents were more likely to have drinking water that did not consistently meet national health standards (Switzer and Teodoro 2017).

Past examinations of potential drinking water contamination exposure disparities have largely focused on urban drinking water systems. These systems serve the majority of the United States population and due to SDWA monitoring and reporting requirements, data on elevated levels of contaminants of human health concern are publicly available. However, an estimated 15 million U.S. households are reliant on private drinking water systems such as groundwater wells (CDC 2018). As these systems fall outside the auspices of the SDWA, monitoring water quality and maintenance of system function are solely the responsibility of the individual homeowner. Multiple studies suggest that contamination at the system point of use by fecal indicator bacteria such as coliform and *E. coli* is quite common for these homes (Allevi et al. 2013) and lower income households reliant on private systems are more likely to have drinking water that is fecal indicator bacteria positive (Smith et al. 2014). The presence of fecal indicator bacteria in drinking water from private wells has been linked to elevated prevalence of acute and/or chronic gastroenteritis (Denno et al. 2009; Wallender et al. 2014; DeFelice et al. 2016). In addition to an elevated risk of exposure to infectious waterborne microorganisms, water from these systems can also contain elevated concentrations of toxins such as heavy metals. Pieper et al. (2015) reported that up to 20% of household water samples from private wells and springs submitted to a state extension program in Virginia contained lead above the 15 ppb limit recommended by the USEPA, with 1% of samples containing levels over 97 ppb.

In the Central Appalachian Coalfields in the eastern United States the challenges inherent in providing homes with reliable safe drinking water are exacerbated by poverty and unique topographical challenges. Of particular interest to this work are those homes in the region without reliable in-home access to safe drinking water and/or appropriate sanitation. Despite decades of investment, there remain regions of West Virginia and Kentucky where up to one in ten homes lack complete indoor plumbing (Krometis et al. 2017). Incomplete or inadequate household plumbing can result in makeshift solutions that potentially expose residents to elevated levels of water quality contamination, but, because they circumvent regulations, are difficult to locate or quantify. For example, low population densities generally preclude the development of centralized wastewater treatment, but because of the thin soils and karstic geology of the region, septic systems are often inappropriate or prone to failure. Consequently, some residents simply "straight pipe" their household wastewater, i.e., all grey and blackwater is simply piped to an open-air ditch and directed into nearby surface water (Banks et al. 2005; Cook et al. 2015; Lilly et al. 2015). The discharge of untreated wastewater is technically illegal, but this not uncommon strategy is not formally inventoried by water quality or public health managers. Recent estimates suggest up to two-thirds of homes in McDowell County, WV (Lilly et al. 2015) and 3,000 homes in Letcher County, KY (Glasmeier and Farrigan 2003) straight-pipe their sewage to local streams. Not surprisingly, streams receiving straight-piped sewage contain elevated concentrations of fecal indicator bacteria, at times detectable for miles beyond the initial discharge (Cantor et al. 2017)

This ambient contamination of environmental waters is of particular public health concern given that many homes reliant on private systems do not employ treatment (Smith et al. 2014), and that households without in-home access to acceptable drinking water may rely on these waters to meet their needs. Although some homes that either do not have indoor plumbing or perceive their drinking water to be contaminated may meet their drinking and cooking needs with bottled water, this can be quite expensive and represent a significant portion of total household income (McSpirit and Reid 2011). Other homes may therefore rely on roadside or "spout" springs, i.e., piped surface or groundwaters freely available at a public location. Very little is known about typical use of these sources, the quality of this water, and the motivations for collecting water at these locations. Swistock et al. (2015) collected water samples from 35 roadside springs in Pennsylvania and reported that 91% of samples were positive for total coliform and 32% were positive for *E. coli*. A parallel survey of attendees at Pennsylvania Extension workshops indicated that over 30% of the >1,000 attendees had used a roadside spring for drinking water, though only a small number of these attendees were regular users (i.e., <3% used the water at least once a week) (Swistock et al. 2015). In an interdisciplinary effort aimed at inventorying Appalachian water access and disaster preparedness, Arcipowski et al. conducted extensive surveys of 30 homes in eastern Kentucky and sampled 16 local surface water access points used for drinking water and/or recreation (Arcipowski et al. 2017). All sites but one were positive for fecal coliform, and 11 sites exceeded the Kentucky surface water standard of 200 MPN fecal coliforms/100 mL. Households without in-home piped water indicated that they were at times dependent on some of these sources for potable water. Of those homes surveyed, 17% did not have an indoor toilet. Though the remaining 83% reported use of a septic system, the researchers observed straight-piped wastewater entering these surface waters, which represents a potential source of contamination of water collection points.

This present effort aimed to conduct a preliminary investigation of water quality at public water collection points ("spout springs") located in the Central Appalachian region and to determine the motivations of regular spring users. This work is designed to lay a foundation for future outreach efforts and to better define the remaining challenges that render provision of safe drinking water in rural communities in the United States difficult. Explicitly defining these rural environmental health challenges will allow for comparison with more urban issues in the provision of safe drinking water to determine potential common solutions.

Methodology

Spring Selection

Between 2016 and 2018, a total of 83 samples were collected at 21 separate spout springs in five states (Virginia, West Virginia, North Carolina, Kentucky, and Tennessee). Given the considerable travel distances required to reach some of these spring sites, the total number of samples collected at each spring varied from 1 to 13 samples over this time frame. Spring sites were located using the public website www.findaspring.com, discussions with local public health offices, and community word-of-mouth. All springs were publicly accessible, i.e., they were directly adjacent to a public road or on public land. At some springs there was occasional makeshift signage (e.g., a sign tied to a tree) indicating that water quality was not monitored, or suggesting boiling prior to use.

Sample Collection and Analysis

Water was collected on-site at each spring and tested for conductivity, pH, and temperature via a YSI Quattro Pro (YSI Inc., Yellow Springs, OH). On all sampling trips, an additional sample was collected in a pre-sterilized polypropylene bottle and transported to Virginia Tech on ice for bacteriological analysis. Samples were analyzed promptly upon return to the lab via the Colilert defined substrate method for total coliforms and *E. coli* (www.idexx.com, Westbrook, MN). Additional funding during the second year of the project facilitated collection and analysis of samples for inorganic metallic ions. Samples were collected at 19 of the 21 springs (samples from one spring were lost in analysis; a neighbor adjacent to another spring requested no more sampling, which we honored although the spring was on public land). These samples were collected in a separate acid-washed sterile bottle and analyzed via ICP-IMS according to Standard Methods 3030D and 3125B (APHA/AWWA/WEF 1998). Nitrate and fluoride concentrations were determined via Standard Methods 4500-NH and 300, respectively (APHA/AWWA/WEF 1998).

Household Survey

Pre-addressed and pre-stamped short surveys were left at 12 spring locations identified as having interesting water quality results, convenient access, and/or active user communities (Figure 1). Surveys consisted of four short multiple-choice questions (Table 1) crafted to determine typical rates of use (question 1); types of use (question 2); potential alternative sources of water (question 3); and motivations (question 4). Questions were designed to be short, direct, and at a middle-school or below reading level given low rates of regional literacy (Shaw et al. 2004). Survey design, collection, and analysis were approved by the Virginia Tech Institutional Review Board (IRB#16-910). Upon receipt, surveys were coded within a Microsoft Access database. As respondents could select more than one option for multiple choice questions, each category was coded as a "1" (checked) or "0" (unchecked). Comments for "other" categories and/or marginalia were recorded verbatim.

Results and Discussion

Water Quality at Springs

All samples except for one (99%) were positive for total coliform bacteria, sometimes at very high

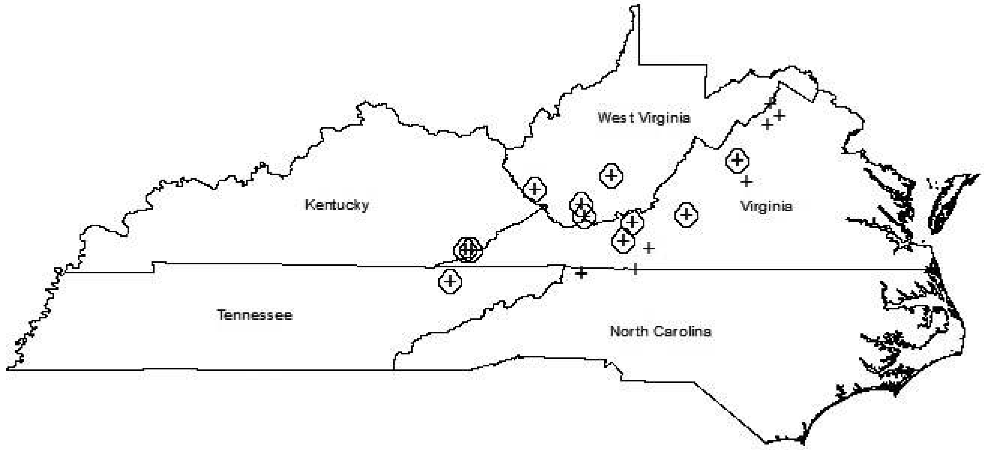


Figure 1. Spring locations for water quality sampling. Surveys were left at circled sites.

Table 1. Spring use survey questions.

How often do you collect water at this spring?
□ Once a day
□ Once a week
□ Once a month
□ Other:______________________________
What do you use the spring water for? (Check all that apply)
□ Drinking
□ Brewing beer
□ Cooking
□ Washing
□ Other:______________________________
What kind of water do you have at home?
□ City/municipal water
□ Well water
□ Cistern water
□ Other:______________________________
Why do you collect spring water?
□ Taste
□ Easy (convenient)
□ Quality/health
□ Price
□ Other:______________________________

levels (Table 2). Current USEPA standards for municipal drinking waters mandate that coliforms be entirely absent (USEPA 2018). It is not surprising however, that coliforms were present in spring samples as this bacterial family includes many species naturally present in soil (Leclerc et al. 2001), and these waters are wholly untreated and not subject to disinfection. Perhaps of greater concern is the finding that 86% of all samples were positive for *E. coli*, and 17 different springs (81% of springs) were positive for *E. coli* at least once during sampling (Table 2). The presence of *E. coli*, a specific species of coliform, is considered indicative of direct fecal contamination and potential human health risk (Paruch and Mæhlum 2012). Detection of *E. coli* in municipal waters would not only be in violation of the associated USEPA SDWA standard, but would trigger a local boil advisory to safeguard the public health.

Spring water samples were largely in accordance with SDWA standards for municipal waters for the remaining water quality targets, with the exception of two springs that exceeded the guidance level for sodium at least once, two springs that exceeded the secondary maximum contaminant level (SMCL; for taste and aesthetics) for manganese at least once, and six springs that exceeded the SMCL for aluminum at least once (Table 3). The current sodium guideline (20,000 ppb, i.e., 20 mg/L) is specifically designed to accommodate individuals following a low-salt diet based on a physician's recommendation; it is therefore worth noting that

Table 2. Bacteriological spring water quality results (* = spring with usage survey results).

Spring #	State	Samples Collected (#)	----------Total coliforms---------- % Positive	Concentration Range (MPN/100 mL)	----------*Escherichia coli*---------- % Positive	Concentration Range (MPN/100 mL)
1*	VA	4	100	23 - 39	25	0 - 7
2*	VA	6	100	21 - 908	67	0 - 71
3	VA	2	100	24 - 159	50	0 - 1
4	VA	1	100	159	100	3
5	VA	3	100	299 - 417	33	0 - 33
6	VA	3	100	81 - 292	100	5 - 22
7*	VA	4	100	27 - 505	100	1 - 18
8	NC	2	100	57 - 2,419	0	0
9	NC	2	100	20 - 74	0	0
10	NC	2	50	0 - 134	0	0
11*	VA	3	100	17 - 60	67	0 - 1
12*	VA	13	100	295 - 2,149	100	1 - 583
13*	WV	9	100	15 - 438	67	0 - 4
14*	WV	5	100	1 - 24	20	0 - 2
15	VA	6	100	1 - 195	17	0 - 1
16*	KY	1	100	6	0	0
17*	WV	5	100	3 - 6	20	0 - 1
18	VA	1	100	1,413	100	26
19*	TN	1	100	28	100	3
20*	KY	1	100	2,203	100	14
21*	WV	6	100	87 - 1,230	83	0 - 113

several common chronic illnesses that are often partially treated with a low salt diet, including heart disease, are notably higher in this region of Appalachia (Krometis et al. 2017). The origin of the high sodium level has not been confirmed, though the natural geology of this region is characterized by ancient sea water trapped in sediments at the time of deposition which can then be released via groundwater ion exchange (Heath 1983). In addition, a survey respondent stated that s/he believed that spring 13 (which had the highest recorded sodium levels) was actually the outfall of a flooded underground mine. The respondent still collected this water for drinking regularly and did not note a poor or salty taste.

Motivations for Water Collection at Springs

In total, 35 surveys were returned. The number of surveys returned varied from one to seven per spring. The majority of respondents indicated that they collected the water directly for drinking (86%), with 63% indicating that they visited the spring at least once per week. This is noteworthy, as many of these sites are not located near communities

Table 3. Maximum observed concentrations of inorganic ions at each spring (ppb). Comparable standards are USEPA maximum contaminant levels for municipal drinking water unless otherwise noted. Values in exceedance of these standards are in bold. (“BD” = below detection limit; “ND” = sampling not done for this spring).

Spring #	NO_3-N	Fe	Mn	Pb	As	F	Na	Cu	Se	Cd	U	Al
1	0	25.7	0.7	0	0	110	934	0	BD	0	0.2	13
2	190	23.8	1.4	0	0	30	6,100	0.1	0.4	0	0	10
3	20	19.7	0.7	BD	BD	30	1,397	BD	BD	0	0	12.9
4	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
5	BD	18.5	2.7	BD	BD	30	871	0.3	0	0	0.1	17.5
6	2,230	9.4	2	0.3	0	40	4,970	1.1	0.5	0	0.3	4.3
7	3,210	29.9	0.7	0	0	30	5,082	1.2	0.2	0	0.3	13.1
8	980	61.4	3.1	BD	BD	50	5,468	0	BD	0	0	12.6
9	40	16.3	0.4	BD	BD	30	1,491	BD	BD	0	0	9.5
10	BD	12.8	37.7	BD	0.4	70	6,598	BD	BD	0	0	5.2
11	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
12	290	75.5	3.9	0	0.1	30	1,078	0.1	0.2	0	0.1	49.2
13	160	265	43.2	0.1	0.4	150	**198,400**	0.8	2.9	0	0.6	**72.6**
14	790	70.6	**1,903**	2.6	0	150	2,491	3.4	0.7	0.3	0.1	**1,808**
15	30	82	2	0	0.1	70	2,595	0.4	0.2	0	0.1	**50.3**
16	150	104.8	39.4	0.1	0.2	270	**93,840**	0.3	0.4	0	0	8.9
17	220	85.1	0.5	0	0	60	8,064	0.6	3.4	0	0.1	49.3
18	600	103.2	2.9	0.4	0	20	2,631	0.1	0.2	0	0	**100.2**
19	350	5.9	0.9	0	0.2	10	644	0.1	0.5	0	0.2	3.7
20	460	55.7	**166.4**	0.1	0	80	6,443	0.4	0.6	0.2	0	**313.9**
21	90	192.2	2.9	0.2	0	70	16,200	0.5	0.9	0	0.1	**168.1**
Standard	10,000	300ǂ	50ǂ	15	10	4,000	20,000*	1,300	50	5	30	50 - 200ǂ

ǂSecondary maximum contaminant level (SMCL)

*USEPA guidance level for low-salt diets and so would require some time and planning to reach. This also represents a slightly different population than that identified by Swistock et al. (2015), which primarily inventoried occasional spring users. Of those responding, 48% indicated they had municipal water at home, 40% were dependent on a well, and two listed "other". One respondent was dependent on a cistern s/he filled regularly with spring water. This was an intriguing finding, as it was initially hypothesized that regular spring users might not have in-home water as described by Arcinpowski et al. (2017)'s work in rural Kentucky. The respondents to this survey largely had in-home water sources but preferred spring water. The majority indicated that taste was a primary reason to collect spring water (66%), with 57% also selecting "quality/health" as a motivating factor.

Somewhat surprisingly, many of the respondents included substantial marginalia or even short letters accompanying their returned surveys. These comments provide additional subtlety to the short survey responses and suggest important areas for future research and outreach or community education efforts. For example, it appears many of the respondents simply do not trust their home water source, given responses such as:

> *The well water we have is not good to drink or cook with.*

> *Too many times we don't get notified if there is a boil advisory…They have also been cited with chemical violations (not enough or too much) and we don't hear about them til after the fact.*

> *City water is toxic.*

> *I have had the honor of being raised on well and spring water…I love good old mountain spring water and truly believe it's better than any nasty, chlorine tasting city water.*

However, some respondents indicate that they are reliant on this water as their only option:

> *People cannot afford their water bills.*

> *The president of the water system didn't bring the water meter in the yard. We can't afford to dig a ditch from the yard to water meter that [sic] about 300 feet from the house.*

> *When our old water system for [X] fails, we often used this water source.*

> *When there is a dry season or when our pump went out in our well, we collected gallons and gallons of this spring water to get us through.*

Potential education and outreach efforts to spring users would differ substantially based upon these users' stated motivations for collecting and drinking spring water, as well as the actual quality of their in-home water source. The perception that spring water is more "natural" or pleasant-tasting was also cited by Swistock et al. (2015) in their survey of roadside spring users. Water taste can vary greatly amongst individuals, and is a poor indicator of most contaminants. However, perceptions such as poor taste or changes in color can be critical in an individual's decision to have their drinking water tested or seek a different source (Imgrund et al. 2011; Kreutzwiser et al. 2011; McSpirit and Reid 2011; Wedgworth et al. 2014). It is critical for local physicians, extension agents, and health departments to emphasize that taste or appearance alone is not a sufficient indicator that water is safe to drink. Future work should investigate whether this messaging is most effective if conveyed via simple roadside signage, extension publications, or more targeted community messaging.

Given local reports of municipal water infrastructure challenges and frequent violations of the SDWA by some treatment plants, for some communities these springs may present less risk than in-home drinking water (Kounang 2018; Pytalski 2018). For example, a cursory review of SDWA violations in McDowell County, WV, where one of these springs is located, lists 3,613 violations by the county's 25 municipal water plants since 2008 (https://www3.epa.gov/enviro/facts/sdwis/search.html). Simultaneously, residents in Appalachia often have higher water utility rates than national averages (Hughes et al. 2005). It is likely extensive investment in local infrastructure coupled with a substantial public outreach campaign would be required in these areas to rebuild the public trust in point-of-use drinking water.

Limitations

Though the results presented here are at times compelling, it is important to make several key limitations explicit. First, it is likely that spring water quality varies considerably based on climatic conditions and seasonality, especially given the karstic geology of the region; many of these "springs" may be re-emergent surface water or heavily influenced by surface water contamination (White 2018). Second, respondents were self-selected: those who responded were likely interested in the springs, comfortable with providing their information and opinions, and had the time and capacity to respond. Though their experiences and responses echo those reported in previous research efforts reported in Pennsylvania (Swistock et al. 2015) and current reports in popular media (Kounang 2018), these findings should not at this point be considered representative of their communities as a whole.

Future Needs

It is certainly striking to learn that some rural Americans find unregulated and untreated environmental waters preferable to the water from their tap, given the current assumption that the United States has near universal access to clean water. Appalachia is not the only rural region of the United States with struggles in providing residents safe drinking water and adequate sanitation (Gasteyer and Vaswani 2004; Izenberg et al. 2014; Wedgworth et al. 2014). Recent national analyses suggest that rural drinking water systems are more likely to report health-based SDWA violations (Allaire et al. 2018) as well as failures to adequately monitor and report water quality (Rubin 2013). A critical need when assessing the relative impacts of these failures is an investigation of whether substandard drinking water quality results in measurable adverse health outcomes. The previous Pennsylvania roadside spring study cited anecdotal health provider reports of elevated incidence of waterborne diseases such as giardiasis in individuals who use roadside springs (Swistock et al. 2015), but there have been no epidemiological studies reporting on the impacts of exposure to chronically noncompliant municipal drinking water in this region. Regardless, local physicians and health departments should be aware of this potential risk, and the means by which these communities attempt to avoid these risks by seeking out waters they perceive as healthier. Systematic door-to-door surveys should be used to determine water and sanitation challenges in rural regions in order to create sustainable communities with adequate infrastructure, and point-of-use water quality checks should be used as a means to simultaneously educate local citizens and identify contaminants of concern.

Conclusions

This effort demonstrated both that roadside springs are used as a source of potable household water by some households in Appalachia and that water from these springs is frequently contaminated by fecal indicator bacteria, suggesting a potential health risk. These results are currently being used to design and implement a household study to determine whether the in-home water of regular spring users is of comparatively better or worse quality than that observed for their spring, and to more intentionally examine how perceptions of water quality drive behavior. In addition, Cooperative Extension materials are being planned to provide these data and information on local springs to the public. Given survey responses, it appears many of these springs are culturally significant and may also meet a real need when other sources are unavailable. Consequently, it may prove most effective in some communities to work to develop simple treatment and/or water quality protection plans at spring collection sites rather than solely discouraging their use.

Acknowledgements

This work was supported by the Virginia Tech Institute for Science, Culture, and the Environment; Virginia Tech Fralin Life Science Institute; Virginia Tech Global Change Center; and the Virginia Tech Exposome Center. The authors gratefully acknowledge Ethan Smith, Casey Schrading, and Lauren Wind for their assistance in sample collection.

Author Bio and Contact Information

Leigh-Anne Henry Krometis (Ph.D.) (corresponding author) is an Associate Professor in Biological Systems Engineering at Virginia Tech. She earned her Ph.D. in Environmental Engineering from the Gillings School of Global Health at the University of North Carolina in 2009. Her areas of expertise include microbial source tracking, waterborne pathogen transport, and provision of safe drinking water and appropriate sanitation within the United States. She has completed field, community, and laboratory level work investigating the movement of waterborne contaminants in rural environments, with a particular focus in Appalachia. Leigh-Anne can be contacted at krometis@vt.edu.

Hannah Patton is a current graduate student in Biological Systems Engineering at Virginia Tech. She has completed coursework in environmental health, environmental microbiology, and environmental ethics, with a planned thesis focus on the provision of safe drinking water in the Appalachian Coalfields. She completed a BS in Environmental Engineering at Saint Francis University in 2017, during which she completed undergraduate research and outreach projects related to acid mine remediation and minelands restoration. Hannah can be contacted at hpatton@vt.edu.

Austin Wozniak is an undergraduate research student at Virginia Tech. He completed a BS in Biological Systems Engineering in 2018, with a BA in Finance anticipated for 2019. Austin spent two years as an undergraduate student completing field work and laboratory analysis related to this project. He can be contacted at woz1@vt.edu.

Emily Allyn Sarver (Ph.D.) is an Associate Professor in Mining and Minerals Engineering at Virginia Tech. She completed her Ph.D. in Environmental Engineering at Virginia Tech in 2010. Her areas of research interest include prevention of black lung, responsible energy development, corrosion control, and characterization of environmental impacts. In 2015, she was the co-recipient of the Appalachian Research Initiative for Environmental Sciences' Researcher of the Year award with co-author L. Krometis for their interdisciplinary work on the ecological and human health impacts of straight pipe sewage discharges in Virginia and Kentucky. Emily can be contacted at esarver@vt.edu.

References

Allaire, M., H. Wu, and U. Lall. 2018. National trends in drinking water quality violations. *Proceedings of the National Academy of Sciences* 115(9): 2078-2083.

Allevi, R.P., L.-A. Krometis, C. Hagedorn, B. Benham, A.H. Lawrence, E.J. Ling, and P.E. Ziegler. 2013. Quantitative analysis of microbial contamination in private drinking water supply systems. *Journal of Water and Health* 11(2): 244-255.

APHA/AWWA/WEF. 1998. *Standard Methods for Examination of Water and Wastewater* (20th ed.). American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, D.C.

Arcipowski, E., J. Schwartz, L. Davenport, M. Hayes, and T. Nolan. 2017. Clean water, clean life: Promoting healthier, accessible water in rural Appalachia. *Journal of Contemporary Water Research & Education* 161(1): 1-18.

Banks, A., A. Jones, and A. Blakeney. 2005. Headwaters: A student/faculty participatory research project in an eastern Kentucky community. *Journal of Appalachian Studies* 11(1/2): 104-132.

Cantor, J., L.-A. Krometis, E. Sarver, N. Cook, and B. Badgley. 2017. Tracking the downstream impacts of inadequate sanitation in central Appalachia. *Journal of Water and Health* 15(4): 580-590.

CDC. 2018. Private Ground Water Wells. Available at: https://www.cdc.gov/healthywater/drinking/private/wells/index.html. Accessed April 1, 2019.

Cook, N., E. Sarver, and L.-A. Krometis. 2015. Putting corporate social responsibility to work in mining communities: Exploring community needs for central Appalachian wastewater treatment. *Resources* 4(2): 185-202.

DeFelice, N., J. Johnston, and J. MacDonald-Gibson. 2016. Reducing emergency department visits for acute gastrointestinal illnesses in North Carolina (USA) by extending community water service. *Environmental Health Perspectives* 124(10): 1583-1591.

Denno, D.M., W.E. Keene, C.M. Hutter, J.K. Koepsell, M. Patnode, D. Flodin-Hursh, L.K. Stewart, J.S. Duchin, L. Rasmussen, and P.I. Tarr. 2009. Tri-County comprehensive assessment of risk factors for sporadic reportable bacterial enteric infection in children. *Journal of Infectious Disease* 199(4): 467-476.

Gasteyer, S. and R. Vaswani. 2004. Still Living Without the Basics in the 21st Century: Analyzing the Availability of Water and Sanitation Services in the United States. Rural Community Assistance Project. Available at: http://opportunitylinkmt.org/wp-content/uploads/2015/07/Still-Living-Without-the-Basics-Water.pdf. Accessed April 1, 2019.

Glasmeier, A.K. and T.L. Farrigan. 2003. Poverty, sustainability, and the culture of despair: Can sustainable development strategies support poverty alleviation in America's most environmentally challenged communities? *The Annals of the American Academy of Political and Social Science* 590: 131-149.

Heath, R. 1983. *Basic Ground-Water Hydrology*. U.S. Geological Survey Water-Supply Paper 2220. Reston,VA, p. 86. Available at: https://doi.org/10.3133/wsp2220. Accessed April 1, 2019.

Hughes, J., R. Whisnant, L. Weller, S. Eskaf, M. Richardson, S. Morrissey, and B. Altz-Stamm. 2005. Drinking Water and Wastewater Infrastructure in Appalachia: An Analysis of Capital Funding and Funding Gaps. Available at: http://www.arc.gov/assets/research_reports/drinkingwaterandwastewaterinfrastructure.pdf. Accessed April 1, 2019.

Imgrund, K., R. Kreutzwiser, and R. de Loë. 2011. Influences on the water testing behaviors of private well owners. *Journal of Water and Health* 9(2): 241-252.

Izenberg, M., O. Johns-Yost, P.D. Johnson, and J. Brown. 2014. Nocturnal convenience 1: The problem of securing universal sanitation access in Alabama's Black Belt. *Environmental Justice* 6(6): 200-205.

Katner, A., K.J. Pieper, Y. Lambrinidou, K. Brown, C.-Y. Hu, H.W. Mielke, and M.A. Edwards. 2016. Weaknesses in federal drinking water regulations and public health policies that impede lead poisoning prevention and environmental justice. *Environmental Justice* 9(4): 109-117.

Kounang, N. 2018. The Kentucky County Where the Water Smells Like Diesel. Available at: https://www.cnn.com/2018/03/30/health/kentucky-water-crisis/index.html. Accessed April 1, 2019.

Kreutzwiser, R., R. de Loë, K. Imgrund, M.J. Conboy, H. Simpson, and R. Plummer. 2011. Understanding stewardship behaviour: Factors facilitating and constraining private water well stewardship. *Journal of Environmental Management* 92(4): 1104-1114.

Krometis, L.-A., J. Gohlke, K. Kolivras, E. Satterwhite, S.W. Marmagas, and L.C. Marr. 2017. Environmental health disparities in the Central Appalachian region of the United States. *Reviews on Environmental Health* 32(3): 253-266.

Leclerc, H., D.A.A. Mossel, S.C. Edberg, and C.B. Struijk. 2001. Advances in the bacteriology of the coliform group: Their suitability as markers of microbial water safety. *Annual Review of Microbiology* 55: 201-234.

Lilly, J., G. Board, and R. Todd. 2015. Inside Appalachia: Water in the Coalfields. WV Public Broadcasting. Available at: http://www.wvpublic.org/post/inside-appalachia-water-coalfields#stream/0. Accessed April 1, 2019.

McSpirit, S. and C. Reid. 2011. Residents' perceptions of tap water and decisions to purchase bottled water: A survey analysis from the Appalachian, Big Sandy coal mining region of West Virginia. *Society and Natural Resources* 24(5): 511-520.

Paruch, A.M. and T. Mæhlum. 2012. Specific features of *Escherichia coli* that distinguish it from coliform and thermotolerant coliform bacteria and define it as the most accurate indicator of faecal contamination in the environment. *Ecological Indicators* 23: 140-142.

Pieper, K.J., L.-A. Krometis, D.L. Gallagher, B.L. Benham, and M. Edwards. 2015. Incidence of waterborne lead in private drinking water systems in Virginia. *Journal of Water and Health* 13(3): 897-908.

Pytalski, J. 2018. Water in Appalachia Needs a Trillion Dollar Solution. WV Public Broadcasting. Available at: http://www.wvpublic.org/post/water-appalachia-needs-trillion-dollar-solution#stream/0. Accessed April 1, 2019.

Rubin, S.J. 2013. Expanded summary: Evaluating violations of drinking water regulations. *Journal of the American Water Works Association* 105(3): 51-52.

Shaw, T., A. DeYoun, and E. Redemacher. 2004. Educational attainment in Appalachia: Growing with the nation, but challenges remain. *Journal of Appalachian Studies* 10(3): 307-329.

Smith, T., L.-A.H. Krometis, C. Hagedorn, A.H. Lawrence, B. Benham, E. Ling, P. Ziegler, and S.W. Marmagas. 2014. Associations between fecal indicator bacteria prevalence and demographic data in private water supplies in Virginia. *Journal of Water and Health* 12(4): 824-834.

Swistock, B., J. Clark, S. Boser, D. Oleson, A. Galford, G. Micsky, and M. Madden. 2015. Issues Associated with the use of untreated roadside springs as a source of drinking water. *Journal of Contemporary Water Research & Education* 156: 78-85.

Switzer, D. and M. Teodoro. 2017. The color of drinking water: Class, race, ethnicity, and safe drinking water act compliance. *Journal American Water Works Association* 109(9): 40-45.

Thomasson, E., E. Scharman, E. Fechter-Leggett, D. Bixler, S. Ibrahim, M.A. Duncan, et al. 2017. Acute health effects after the Elk River chemical spill, West Virginia, January 2014. *Public Health Reports* 132(2): 196-202.

USEPA. 2018. Drinking Water Contaminants – Standards and Regulations. Available at: https://

www.epa.gov/dwstandardsregulations. Accessed April 1, 2019.

Wallender, E., E. Ailes, J. Yoder, V. Roberts, and J. Brunkard. 2014. Contributing factors to disease outbreaks associated with untreated groundwater. *Ground Water* 52(6): 886-897.

Wedgworth, J.C., J. Brown, P. Johnson, J.B. Olson, M. Elliott, R. Forehand, and C.E. Stauber. 2014. Associations between perceptions of drinking water service delivery and measured drinking water quality in rural Alabama. *International Journal of Environmental Research and Public Health* 11(7): 7376-7392.

White, W.B. (Ed.). 2018. Karst geomorphology. In: *Caves and Karst of the Greenbrier Valley in West Virginia*. Springer, pp. 45-62.

World Bank. 2015. People Using Safely Managed Drinking Water Services (% of Population). Available at: https://data.worldbank.org/indicator/SH.H2O.SAFE.ZS. Accessed April 1, 2019.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 57-78, April 2019

Groundwater Seeps: Portholes to Evaluate Groundwater's Influence on Stream Water Quality

*Michael O'Driscoll[1], David DeWalle[2], Charles Humphrey Jr.[3], and Guy Iverson[4]

[1]*Department of Coastal Studies, East Carolina University, Greenville, NC*
[2]*Ecosystem Science and Management, Pennsylvania State University, University Park, PA*
[3]*Environmental Health Sciences Program, East Carolina University, Greenville, NC*
[4]*Coastal Resources Management Program, East Carolina University, Greenville, NC*
**Corresponding Author*

Abstract: Recent legal cases have suggested that contaminated seeps and/or springs that have measurable impacts on adjacent surface water quality may fall under the jurisdiction of the Clean Water Act (CWA). An improved understanding of the effects of groundwater seeps on surface water quality is needed to support the evolving legal and regulatory environment. Surface seeps or seepage zones are locations where upwelling groundwater saturates the surface. Seeps can provide groundwater that may be transported to nearby surface waters along surface and shallow subsurface flowpaths. From a water quality perspective, seeps can provide portholes to observe groundwater quality. Here we consider examples of seeps as contaminant sources or sinks across a range of watershed disturbance and synthesize the seep water quality literature to help answer the questions: Why do seeps act as contaminant sinks in some cases and contaminant sources in others? What areas of seep water quality research can help apprise the legal and policy discussion on the role of the CWA to address groundwater contamination that is conveyed to streams? Overall, the case studies and literature review indicated that seep water quality data can provide valuable insights into the effects of stream-groundwater interactions on stream water quality. Future work on seep-surface water interactions is needed to characterize seep water quality behavior across a range of hydrogeological, meteorological, and land-use conditions to better understand the locations where seeps are more likely to convey contaminants to streams and affect stream water quality.

Keywords: *stream-groundwater interactions, riparian, Clean Water Act, hydrologic connection*

The 2018 Universities Council on Water Resources (UCOWR) Annual Meeting (Pittsburgh, PA) session on "Springs and Seeps: Hydrology, Ecosystem Functions, and Management" covered a wide range of spring and seep research and management issues. The general theme was that seeps and springs are valuable windows to better understand groundwater systems and their influence on streams and groundwater-dependent ecosystems. The session called for more seep and spring research to improve understanding of the links between groundwater inputs and stream water quality/ecology. This work is needed to support the evolving legal/regulatory environment.

In the current study, a review of seep water quality literature was supplemented with select case studies of seep behavior across a range of watershed disturbance. This approach was used to answer the question: Why do seeps act as contaminant sinks in some cases and contaminant sources in others? A review of recent legal opinions and seep literature provided a basis for the framing of scientific questions to support the legal and regulatory aspects of contaminated seeps. This work highlights areas of seep water quality and hydrological research that can apprise the legal and policy discussion on the role of the Clean Water Act (CWA) to address groundwater contamination that is conveyed to streams.

Surface seeps are locations where upwelling groundwater saturates the surface. The groundwater may be transported to nearby surface waters along surface and shallow subsurface flowpaths. Seeps are generally considered to be springs with lower discharge magnitudes (Springer and Stevens 2009). Seeps also may be submerged. Although there is extensive literature on spring occurrence (Alfaro and Wallace 1994; Stevens and Meretsky 2008; Springer and Stevens 2009), less research has focused on seeps (Williams 2016). Seeps may differ from springs in that they often emerge over a diffuse area and generally have low flows that do not form channels. Groundwater seeps often flow diffusely through soils and vegetation (Williams 2016), therefore seep discharge may be more difficult to measure relative to larger springs. However, from a water quality perspective, diffuse seeps may receive more filtration and greater potential for biological interaction and treatment.

There is generally a flow-based continuum between seeps and springs (Springer and Stevens 2009); seeps may have a range of conditions from diffuse flow to rivulet-pipe flow (Shabaga and Hill 2010). Those conditions may vary seasonally based on the magnitude of seep discharge and evapotranspiration. Seeps may occur due to an abrupt change in topographic slope (Stein et al. 2004), also referred to as groundwater slope wetlands (Brinson 1993). Seeps may also occur due to a lateral or vertical change in subsurface sediment (Vidon and Hill 2004), soil and/or bedrock hydraulic properties, bedrock contacts, joints, fractures, and fault zones (West et al. 2001) (Figure 1).

There is limited work on groundwater seep classification systems. However, a framework exists for classifying springs based on spring hydrogeology and ecology (Springer and Stevens 2009). This spring characterization work can serve as guidance for further seep characterization efforts. Williams (2016) provided a classification of seeps into three general classes: helocrene (emerges from wetlands/marshy substrate); limnocrene (discharge into a pool); and rheocrene (flowing spring that emerges into channels) (Figure 1). Seeps and springs can also be categorized based on their magnitude of flow and flow permanence. However, since flow permanence assessment requires monitoring, many studies may not have enough data for accurate flow characterization. Williams (2016) recommended a flow characterization system for low flow: <0.01 m^3/s; medium flow: 0.01-0.5 m^3/s; and high flow: >0.5 m^3/s. Flow is an important variable for characterizing seeps and springs because of its influence on temperature and habitat. Seep discharge can influence the local ecology due to its controls on primary productivity, food supply (leaves and detritus), and influence on spring or seep-bed substrates (Williams 2016). Seep flow magnitude and timing can influence the extent of the seep habitat, disturbances, availability of food, temperature, moisture, and water quality. The invertebrate community that lives in and around the seeps is generally adapted to the range of common flow conditions (Williams 2016).

From an ecological perspective, seeps may have less diverse fauna than springs, but there may be genera found only in seeps (seep specialists) (Williams 2016). In addition to habitat for seep and spring specialists, seeps are important to groundwater-dependent ecosystems due to the groundwater inputs they provide and their influences on temperature, water chemistry, riverine biota, and in-stream processes (Boulton and Hancock 2006). Seeps can provide a wide range of ecosystem services (Figure 2) (Griebler and Avramov 2015). Seeps can serve as a linkage between the groundwater and surface water system and during summer base flows, may provide the dominant source of streamflow in some headwater catchments (Burns et al. 1998; West et al. 2001; O'Driscoll and DeWalle 2010; Morley et al. 2011). Seeps and other groundwater inputs are important to sustaining streamflows, as groundwater is the primary source of streamflow in many catchments across the globe (Winter 2007; Santhi et al. 2008; Beck et al. 2013; Miller et al. 2016).

Seeps can bestow water quality services by contributing to food webs (Williams 2016) and by attenuation of contaminants (O'Driscoll and DeWalle 2010). However, seeps can also act as net contaminant sources (Williams et al. 2014, 2015; Humphrey et al. 2018). From a water quality perspective, seeps can provide portholes to observe groundwater quality. When groundwater flowpaths transport contaminants to seeps, the discharge water quality can provide important insights into subsurface contaminant attenuation.

Although seeps may make up a relatively small extent of a catchment, they are important components of the watershed ecosystem because of their capability to translate groundwater

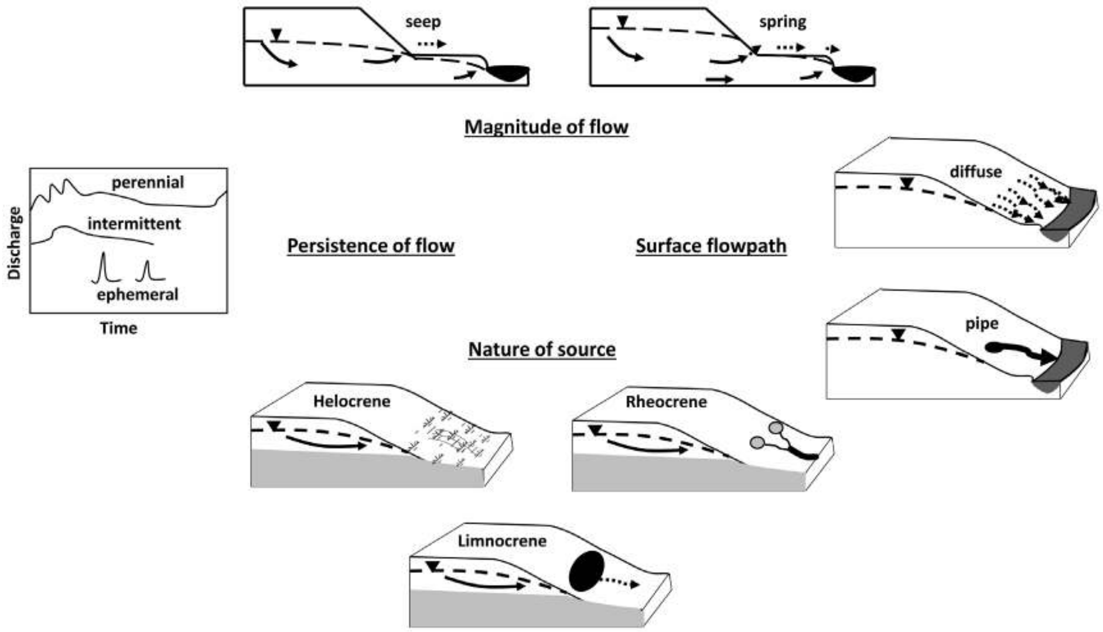


Figure 1. Variables that influence seep flow and their influence on downstream water quality (modified from Hill 1996; Shabaga and Hill 2010; Williams 2016).

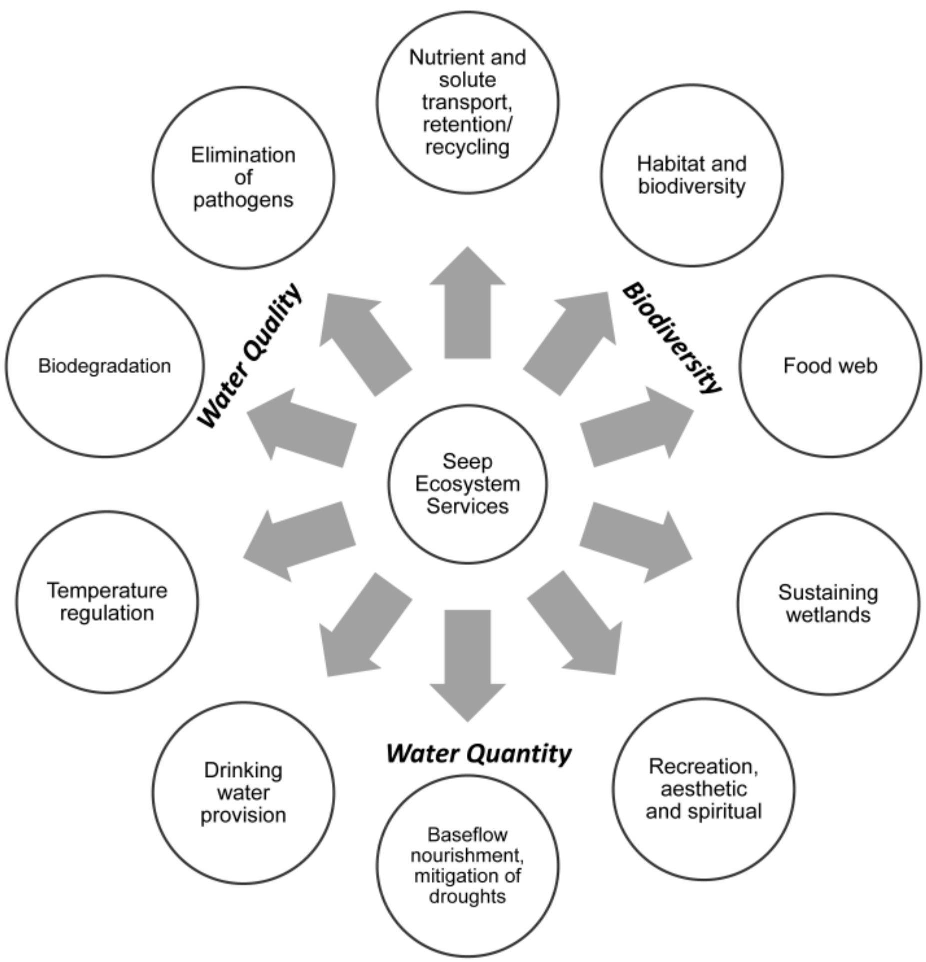


Figure 2. The variety of ecosystem services that seeps can provide including contributions to water quality, water quantity, and biodiversity (modified from Griebler and Avramov 2015).

contaminants to streams and wetlands (Williams et al. 2014; Humphrey et al. 2018) and act as nutrient cycling (McClain et al. 2003) and ecological diversity hotpots (Stevens and Meretsky 2008; Griebler and Avramov 2015; Williams 2016). Springs and seeps are key aquatic habitats because they exert a broad influence on regional ecosystem structure, function, and evolutionary processes (Stevens and Meretsky 2008). The next section will focus on seep water quality behavior across a gradient of watershed disturbance.

Seeps across a Gradient of Disturbance

In this study, examples of a range of seep water quality responses are provided from a series of seep water quality studies conducted across contrasting land-uses. The examples include a relatively undisturbed forested catchment in the Appalachian Plateau (PA), a rural Coastal Plain seep (NC), two suburban seeps in the Piedmont (NC), and an urban Coastal Plain seep (NC) (Figure 3). The seeps were sampled across several different studies, therefore the seep sampling timeframes did not overlap.

At the forested seep site at Baldwin Creek, PA, this Appalachian Plateau watershed was relatively undisturbed. Twenty-three seeps were identified and monitored monthly for a year (O'Driscoll and DeWalle 2010). Fifteen seeps flowed regularly and of these, thirteen were nitrate sinks on an annual basis (Figure 4). The results suggested that temperature (positively) and discharge (inversely) influenced the degree of seep nitrate attenuation. On an annual basis, seep nitrate concentrations declined by 31% along the seep surface flowpath (between the seep emergence point and where the seep flowed into the stream; seep flowpaths ranged from 20 - 400 m, with a median value of 150 m), suggesting that seeps generally acted as nitrate sinks. However, during winter and cooler periods, when discharge was elevated and water temperatures declined, the likelihood for seep nitrate bypass increased (O'Driscoll and DeWalle 2010) (Figure 4).

At a rural seep site in the Coastal Plain of NC (Craven County), surface seep versus subsurface flowpaths were compared for nitrogen attenuation. At this site there was a wastewater plume that was upwelling via a seep that drained to an adjacent

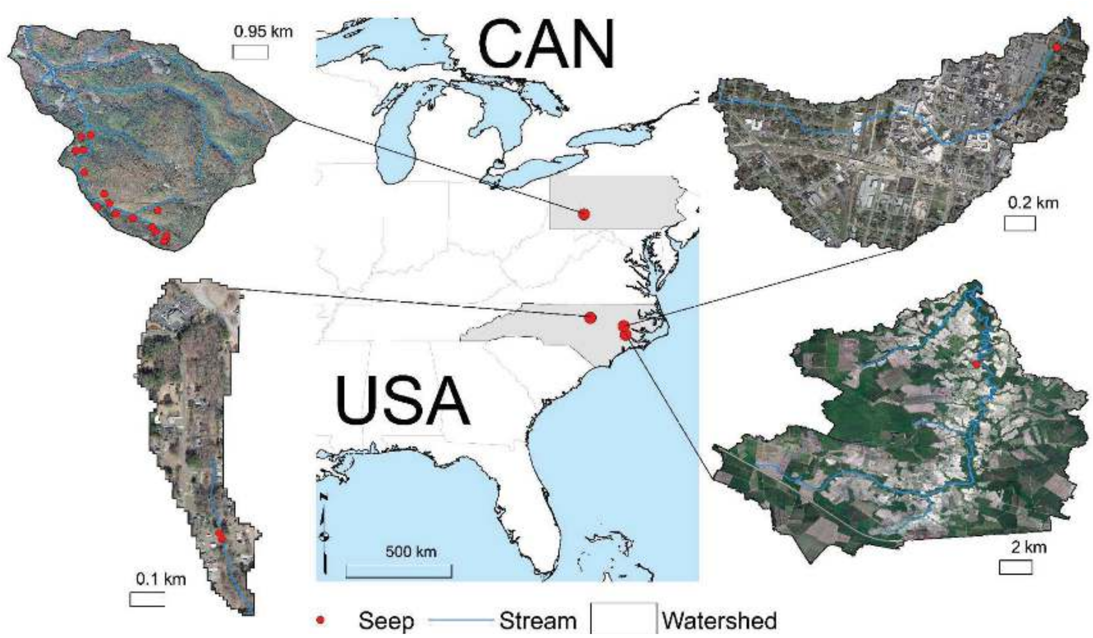
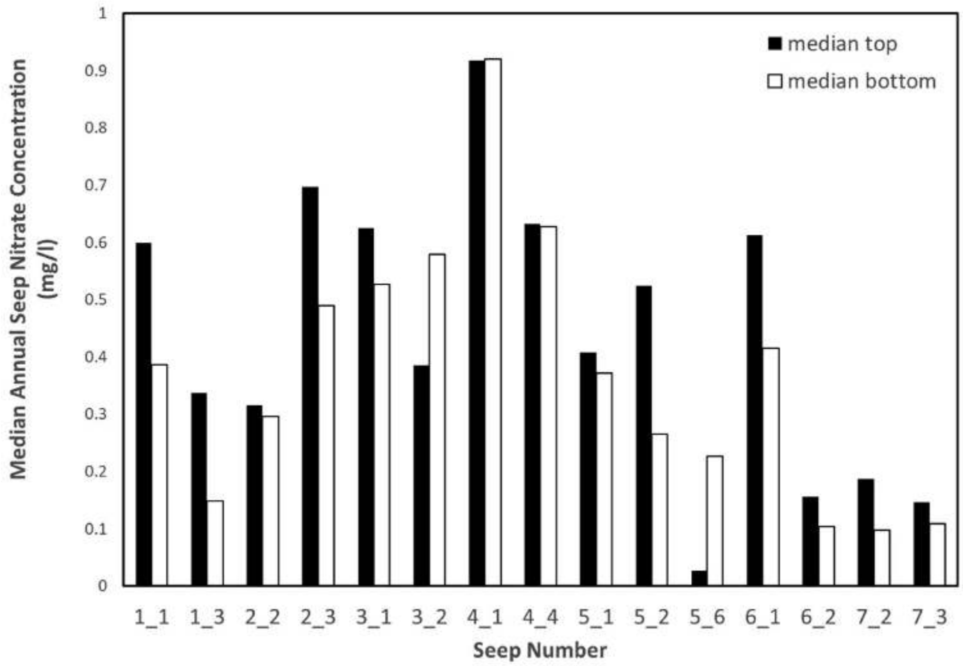


Figure 3. Maps and location information for the four seep water quality sites that occurred across a gradient of watershed disturbance. The sites include a relatively undisturbed forested catchment in the Appalachian Plateau (Baldwin Creek, PA), a rural Coastal Plain seep (Craven Co., NC), two suburban seeps in the Piedmont (Lick Creek, NC), and an urban Coastal Plain seep (Town Creek, NC).



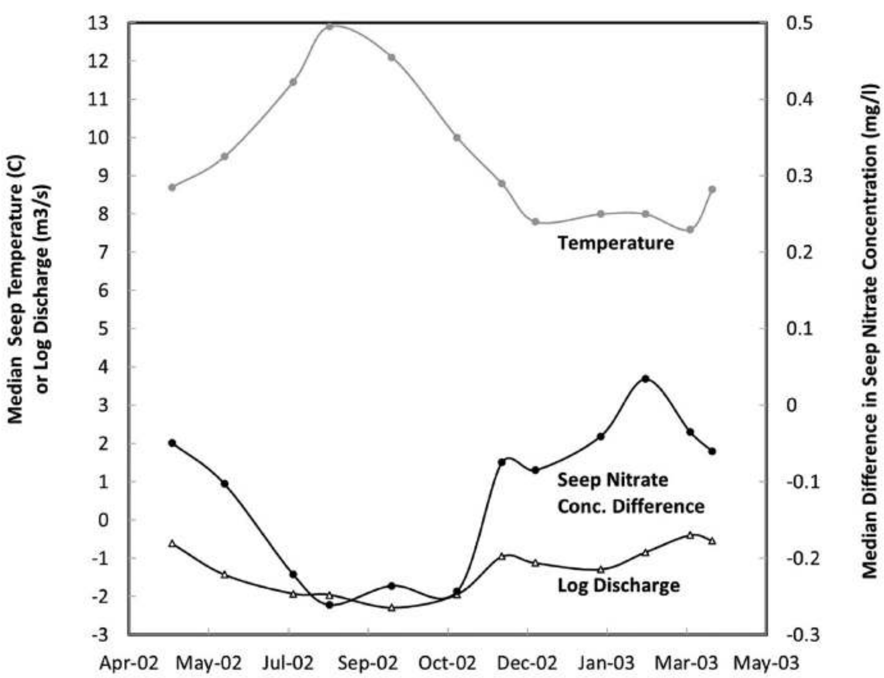


Figure 4. Seep water quality data for 15 seeps at Baldwin Creek, PA that were sampled monthly for a year. (Top) The seep nitrate concentrations typically declined from the seep emergence point to the location where the seep flowed into the stream, suggesting the seeps typically behaved as nitrate sinks. (Bottom) Water temperature and discharge data collected concurrently revealed a direct relationship between seep nitrate attenuation and temperature and an inverse relationship between seep nitrate attenuation and discharge.

stream. The wastewater plume at the site was delineated using electrical resistivity mapping, specific conductance, groundwater nitrogen concentrations, and groundwater chloride data (Humphrey et al. 2013). The seep downgradient of the plume was sampled periodically during 2012-2018 (16 seep sampling events) for comparison with groundwater quality data collected from piezometers. A comparison was made between the groundwater nutrient and chloride concentrations in the riparian buffer and the seep water. The piezometers located in the riparian buffer area had groundwater nitrogen and chloride data that indicated that the wastewater plume was upwelling in the riparian area, but at most riparian piezometers (except for piezometer 18, adjacent to the seep), the nitrogen attenuation in the surficial aquifer and riparian zone sediments was adequate to reduce groundwater total dissolved nitrogen (TDN) concentrations to background levels. However, the groundwater that upwelled at the seep contained elevated nitrogen concentrations associated with the wastewater inputs (Figure 5). A summary of all sampling dates revealed that median TDN declined by 93% (57.3 mg/l to 3.9 mg/l) from the wastewater tank to the riparian buffer wells. However, for the portion of the wastewater plume that upwelled at the seep and flowed into the channel, the decline in groundwater TDN from the tank to the seep was 79% (57.3 mg/l to 12.3 mg/l), suggesting lower nutrient attenuation due to the groundwater flowpath upwelling prior to flowing through the forested riparian buffer. In this case the seep was behaving as a nutrient source to the stream. This

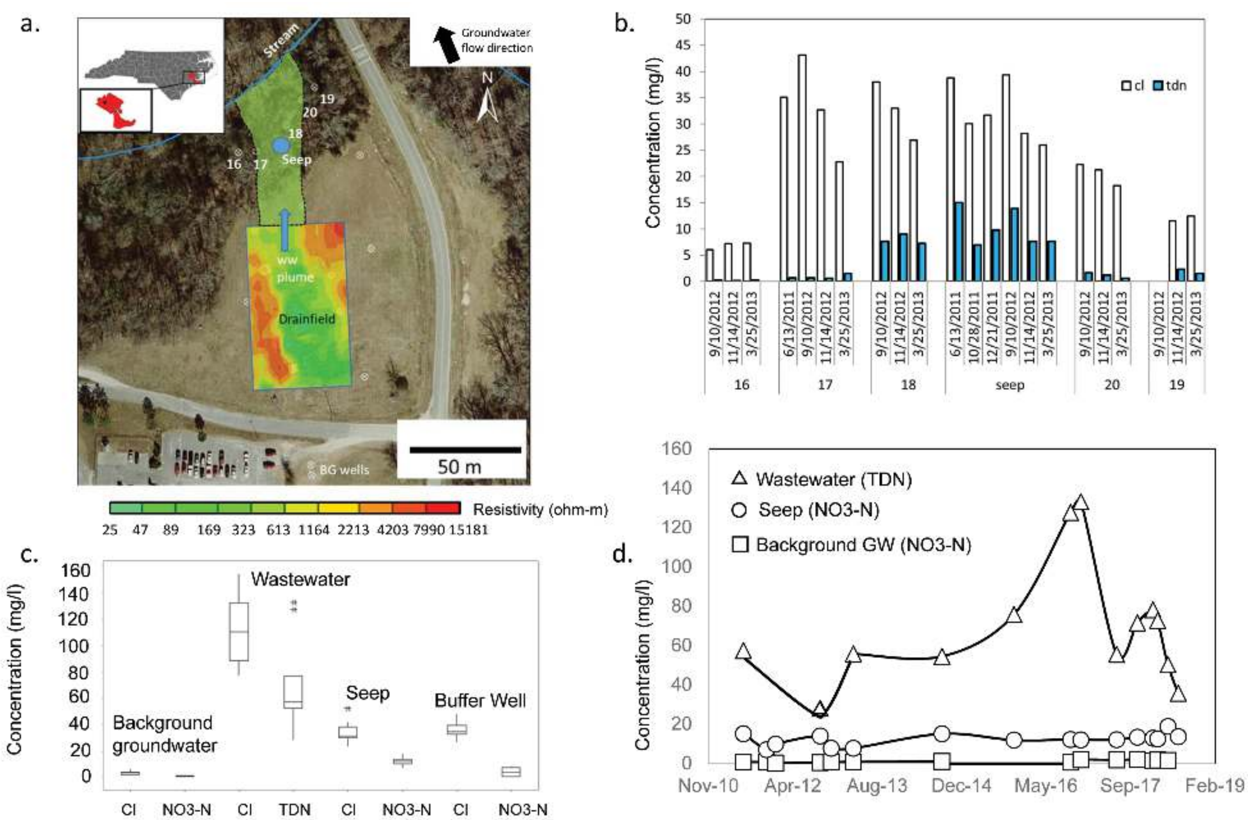


Figure 5. At an elementary school site in Craven Co., NC a seep that drained to a stream was found to be affected by a local wastewater plume. (a.) The wastewater plume at the site was delineated using electrical resistivity mapping and water quality data (Humphrey et al. 2013). (b.) Groundwater TDN and Cl concentrations in the riparian buffer and the seep water revealed that nitrogen attenuation was enhanced when groundwater flowpaths went through the riparian sediments, in contrast to the seep. (c.) Declines in nitrogen concentration between the tank, the adjacent riparian buffer well (MW-17), and the seep suggested enhanced nitrate attenuation in the riparian buffer sediments. (d.) Seep nitrate concentrations were elevated relative to background conditions, indicating wastewater-related nitrogen was being delivered to the seep.

example showed that the flowpath that groundwater takes to the stream can have a large influence on nitrogen delivery to the channel and seeps may act as nutrient (or other contaminant) sources.

At suburban seep sites in Durham County, NC, several seeps were identified at residential sites that drain to a tributary to Falls Lake. Falls Lake is a manmade reservoir that serves as a water supply for the City of Raleigh, NC. It also provides flood control and recreational opportunities. This reservoir has been classified as nutrient-sensitive since the early 1980s and was classified as eutrophic in the early 2000s. Recent nutrient management efforts have been implemented to improve water quality and use attainment (City of Durham 2012). Sampling was conducted to evaluate if the seeps were potentially transporting nutrients from onsite wastewater treatment systems to nearby creeks (Iverson et al. 2019).

Two intermittently flowing seeps were monitored from March 2017-June 2018 (seep 1, n=8; seep 2, n=5; the difference in n values occurred because seeps were not always flowing simultaneously) (Iverson et al. 2019). In an earlier study (Iverson et al. 2018) the median annual stream base flow TDN concentration was 0.97 mg/l for a nearby forested reference stream. Relative to these reference conditions, both seeps contained elevated concentrations of nutrients, but seep 1 had much greater concentrations (Figure 6). The elevated ammonium and TDN from seep 1 may be indicative of a septic system malfunction as raw wastewater generally contains elevated TDN, mostly in the form of ammonium or organic nitrogen (US EPA 2002). Septic system malfunctions can lead to transport of ammonium and/or organic nitrogen (O'Driscoll et al. 2014). It is possible that other sources could contribute elevated TDN and ammonium (e.g., fertilizers, pet and wildlife waste); however, based on other data collected, septic systems appear to be a likely source. Median chloride concentrations in seep 1 and seep 2 were 36.1 mg/l and 28.3 mg/l, respectively. A recent study showed wastewater chloride concentrations sampled from tanks in the study area were between 43.3 mg/l and 50.7 mg/l (Humphrey et al. 2016). The seep chloride concentrations were more similar to wastewater than background stream chloride concentrations in a nearby forested watershed (9.69 mg/l). Similarly, median specific conductance measured at seep 1 and 2 was 520 µs and 242 µs, respectively, and elevated relative to median background levels in a nearby forested stream (108 µs) (Iverson et al. 2018).

$\delta^{15}N_{\text{-nitrate}}$ and $\delta^{18}O_{\text{-nitrate}}$ samples were collected from seep 1 and seep 2. For seep 2, values for $\delta^{15}N_{\text{-nitrate}}$ and $\delta^{18}O_{\text{-nitrate}}$ were 23.6‰ and 11.7‰, respectively, which falls within the manure and septic effluent range of 8 to 23‰ and 0 to 14‰ for $\delta^{15}N_{\text{-nitrate}}$ and $\delta^{18}O_{\text{-nitrate}}$, respectively (Kendall and McDonnell 1998; Silva et al. 2002). However, seep 1 values were lower at 5.5‰ and 1.9‰ for $\delta^{15}N_{\text{-nitrate}}$ and

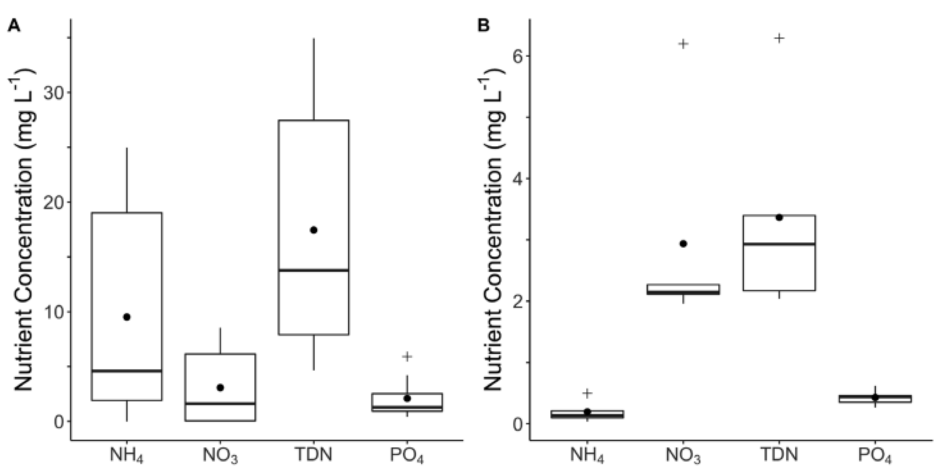


Figure 6. Nutrient concentration data for two residential seep sites along Lick Creek, Falls Lake watershed, NC. Boxplots of nitrogen [ammonium (NH_4-N), nitrate (NO_3-N), total dissolved nitrogen (TDN)] and phosphate (PO_4-P) concentrations for groundwater seep 1 (a.) and seep 2 (b.) Filled circles (•) denote mean values, while pluses (+) denote outliers.

$\delta^{18}O_{\text{-nitrate}}$, respectively, which fell slightly outside the wastewater range for $\delta^{15}N_{\text{-nitrate}}$ (Kendall and McDonnell 1998; Silva et al. 2002). That sample was collected during storm conditions and it is possible that organic, fertilizer, and/or wastewater sources of nitrate were mixed during storm events. These values are only based on one isotopic sample and more sampling would help confirm results. These watersheds contain mostly (> 90%) forest and residential land uses (Iverson et al. 2018), thus agricultural fertilizer is not a likely source of nitrogen. This example showed that seeps may act as conveyances for nutrients from wastewater, lawn fertilizer, and other anthropogenic sources in residential settings.

Seeps may also be affected by legacies of industrial chemical disposal and leaking underground petroleum tanks. Leaking petroleum can lead to BTEX (benzene, toluene, ethylbenzene, and xylene) compound transport to streams via groundwater. At an urban Coastal Plain site, a seep was monitored that was highly impacted by two or more leaking underground storage tanks. The tanks were leaking petroleum prior to the 1980s (Blackmon 2017; Humphrey et al. 2018). Benzene was upwelling with groundwater at the seep and influencing water and soil/air quality (S&ME, Inc. 2011) along Town Creek (Greenville, NC). Soil samples collected away from the seep had lower emissions of benzene in comparison to at the seep and when compared to an unimpaired seep draining the other side of the stream. The

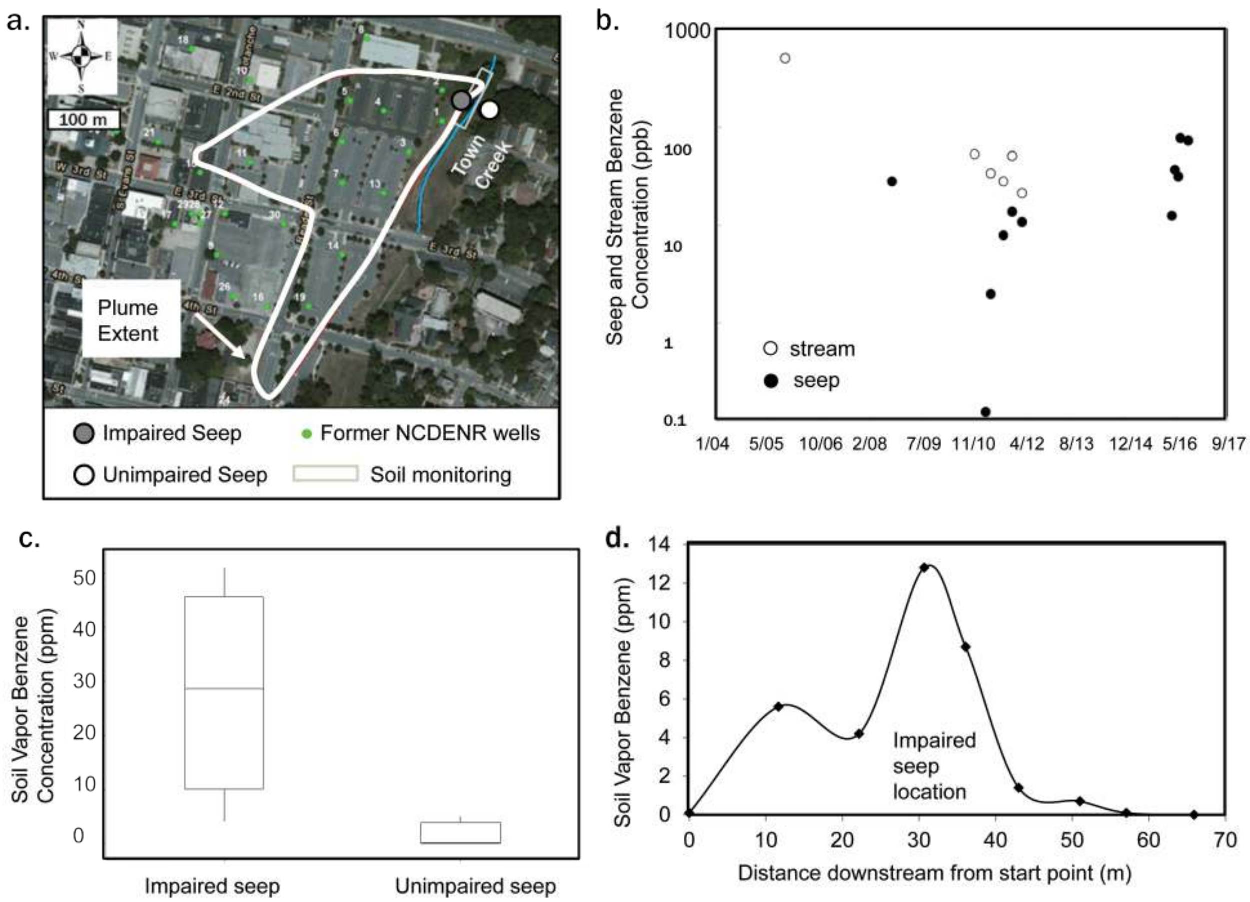


Figure 7. At an urban Coastal Plain site (Town Creek, Greenville, NC), a seep was monitored downgradient of at least two leaking underground petroleum storage tanks. (a.) the plume extent was approximated from an earlier study (NCDENR 1990). (b.) Stream and seep data from earlier studies indicated that benzene from the seep was affecting stream water quality (data source: S and ME 2011 and Humphrey et al. 2018). (c.) The impaired seep showed elevated soil benzene concentrations in contrast to a seep on the opposite side of the stream (sampled on four dates from 4/5/16 to 6/29/16, Blackmon 2017). (d.) Upstream and downstream of the impaired seep the soil benzene concentrations declined (longitudinal survey on 5/25/15).

petroleum-impaired seep was a pathway for benzene exposure via water and air (Figure 7). These data also showed that seep disturbances may originate at long distances from the actual seep, the leaking gas tanks that were the likely contaminant source were approximately 0.4 km or 0.25 miles upgradient of the seep (NCDENR 1990).

Overall, these examples showed that seeps can integrate the effects of upstream land-use disturbances and human activities on groundwater. When undisturbed and surrounded by forest canopy, seeps may be more likely to behave as contaminant sinks (particularly for nutrients), whereas when seep catchments or seeps are disturbed by a variety of human activities, seeps can serve as a conveyance to deliver a range of contaminants to the stream. The seeps that received elevated nutrient concentrations were associated with recent wastewater management activities and best management practices might reduce those inputs. In contrast, the urban seep contamination was associated with a legacy of leaking petroleum tanks; resolving that situation would require a more intensive groundwater remediation effort in the upgradient surficial aquifer. Understanding the nature of the groundwater flowpaths to seeps and associated contaminant sources can improve remedial efforts. In the peer-reviewed literature, there is a wide range of seep behavior documented. Next, the discussion will focus on previous studies on the topic of seep water quality and the factors that lead to seeps behaving as contaminant sinks or sources.

Seeps as Contaminant Sinks

Numerous studies suggest that seeps in forested catchments can act as nutrient sinks (Fisher and Acreman 2004; O'Driscoll and DeWalle 2010; Kaur et al. 2016), but seasonal variability in discharge and reduced biological activity during cooler months can lead to seeps behaving as nitrogen sources during cooler or wetter periods (O'Driscoll and DeWalle 2010; Shabaga and Hill 2010). Surface water – groundwater interactions, discharge, soil type, organic matter, moisture conditions, and vegetation all vary along seeps and their variability can influence the dominant mechanisms of nitrogen transformation and retention along seeps. Additionally, the seasonal and event variability of runoff, temperature, and soil moisture can lead to temporal variability in nitrogen attenuation. It has also been shown that the availability of phosphorus can influence the degree of nitrogen attenuation (Gibson et al. 2015).

In two forested catchments in VT, Kaur et al. (2016) found that seeps had gross nitrification rates approximately three times higher than those for upland soils and nitrate consumption was eight times higher in seep soils vs. upland soils. Overall, their work showed that seep soils can be hotspots for nitrification and denitrification, and the balance can determine if seeps behave as nitrogen sources or sinks. In Baldwin Creek, PA (as previously mentioned), nitrate concentrations in groundwater along a series of seeps declined suggesting that the forested seeps generally acted as nitrate sinks (O'Driscoll and DeWalle 2010). In a tracer study in New Zealand, Rutherford and Nguyen (2004) injected nitrate along seeps (also referred to as riparian swales) to quantify seep nitrate attenuation. They observed a 24% decline in nitrate concentration along a 1.5 m flowpath, indicating that seeps could act as nitrate sinks. Their work suggested that significant nitrate reductions downseep could be achieved when subsurface residence times were a day or longer. However, downseep nitrate concentration bypasses (or increases) likely occur when the surface flowpath dominates the seep discharge (Rutherford and Nguyen 2004). These seep bypasses can play an important role in influencing whether a seep is a nutrient source or sink over time.

Seep bypass can be defined as an occurrence when nitrogen concentrations of upwelling seep water remain constant or increase downseep (Gold et al. 2001; Rosenblatt et al. 2001). There is a portion of seep flow, predominantly surface, that is quickly transported downgradient. This rapid surface flow may not undergo substantial biotic uptake or denitrification (Gold et al. 2001). The mechanisms that can lead to reduced nitrogen attenuation during elevated seep discharge periods include a reduction in: particle settling, sediment–water contact times, nitrogen retention in sediment and/or vegetation (Seitzinger et al. 2002; Shabaga and Hill 2010), and increased flushing of nitrate from soils (Ocampo et al. 2006).

Seeps as Contaminant Sources

A wide variety of studies have documented seeps acting as contaminant sources to rivers. Seeps and/or springs have been documented to transport nutrients (Williams et al. 2015), pesticides (Van Stempvoort et al. 2016), wastewater and pharmaceuticals (Humphrey et al. 2013; Spoelstra et al. 2017), coal combustion products (Harkness et al. 2016), petroleum-related compounds (Humphrey et al. 2018), trichloroethylene (TCE) (Chapman et al. 2007), road salts (Foos 2003), landfill leachate (Atekwana and Krishnamurthy 2004), bacteria (Fisher et al. 2000; Baker et al. 2011), *Giardia* (Rose et al. 1991), and acid mine drainage (Brake et al. 2001; Johnston et al. 2017) to nearby streams and wetlands. Generally, these elevated seep contaminant inputs are related to land-use and human activities within the seep catchment that are associated with fertilizer and manure, pesticide, coal, oil, and gas activities, waste management, wastewater, and livestock, pet, and wildlife waste. However, in some cases forested catchments have also shown elevated nutrient and solute concentrations at seeps (Likens and Buso 2006; Zimmer et al. 2013). One potential explanation is that due to lag times between groundwater recharge and seep or spring discharge, summer base flow can originate from previous dormant seasons when nitrate in recharge is generally elevated (Burns et al. 1998).

Studies revealing seeps as nutrient sources have mainly been conducted in agricultural watersheds (Shabaga and Hill 2010; Williams et al. 2014, 2015, 2016) and are associated with upgradient fertilizer and manure applications. The most detailed work on seeps as nitrogen sources in agricultural watersheds has been performed at Mahantango Creek watershed in central PA by the USDA-ARS. In this agricultural watershed, Williams et al. (2014, 2015, 2016) performed a series of seep studies focused on improving the understanding of agricultural nitrogen transport to streams. In general their work showed that seeps can provide preferential flowpaths that convey nutrients from agricultural fields to streams and can lead to elevated nutrient transport to streams (Williams et al. 2014, 2015, 2016). They recommended to prioritize seep areas for enhanced management in agricultural catchments because they can be nutrient hotspots (Williams et al. 2014). In addition, their work indicated the importance of time-varying stream-groundwater interactions and the influence of seep presence on agricultural nutrient delivery to streams (Williams et al. 2016). In related work, a USGS study across a range of five agricultural watersheds (Tesoriero et al. 2009) looked at base flow and nutrient pathways to streams. They concluded nitrate transport has a high degree of spatiotemporal variability, and preferential flowpaths such as seeps can play a large role in nitrate transport to streams. These studies indicate the importance of detailed riparian groundwater and seep measurements to understand nitrogen delivery to streams.

The type of seep flow can also influence nutrient transport in agricultural watersheds. In Ontario, Canada, Shabaga and Hill (2010) found that the seep flow to the channel played a large role in nitrogen attenuation. They developed a conceptual model of the seep end-members of rivulet-pipe flow and diffuse surface flow. Overall, they found that nitrate removal along rivulet-pipe networks was inefficient, but when waters flowed diffusely through the riparian zone large nitrate declines could occur, particularly in the summer months.

Seeps in agricultural watersheds can also transport pesticides to streams. In a study in the Nottawasaga River Basin, ON, Canada, Van Stempvoort et al. (2016) studied glyphosate, a widely used pesticide that is expected to sorb to soil particles (Borggaard and Gimsing 2008). However, leaching may occur in settings where preferential flowpaths exist, such as groundwater seeps. They collected 153 samples of seep groundwater along the Nottawasaga River and found that 7.8% of those seep samples had detectable concentrations of glyphosate, with most detections occurring in the spring and summer. Shorter term seeps were more likely to have glyphosate since it is more likely to be transported along shorter residence time flowpaths where attenuation is minimal, and those ephemeral seeps may only be active during wetter periods. Overall, the results suggested that glyphosate could be transported from field application sites via groundwater flowpaths to seeps, and seeps that flow less regularly may drain shallower groundwater that is more likely to be contaminated by surface activities. Tang et al. (2012) looked more broadly at general pesticide transport mechanisms from agricultural fields and found that saturation excess runoff generation mechanisms could transport pesticides from field to stream. Upwelling groundwater at seeps flowing to the stream can serve as a transport mechanism. Saturated areas related to toe slopes where seeps may occur are generally more vulnerable to pesticide loss via overland flow than the rest of the catchment because of greater runoff generation in these areas (Tang et al. 2012).

In addition to nutrients and pesticides, agricultural watersheds have also been shown to transport bacteria to seeps. Livestock agriculture can be one of the major causes of bacterial contamination of surface and ground waters (Jamieson et al. 2002). In their review of fecal bacteria transport in agricultural soils and subsurface drainage, they documented the main factors influencing fecal bacteria survival, such as: soil type and moisture conditions, temperature, pH, rate of manure inputs, nutrient status, and microbial competition. Bacterial survival and transport is enhanced in cool conditions and when macropore flows occur, since the physical filtration through micropores is the main factor controlling bacteria mobility. Their work suggests that seep transport of bacteria from livestock operations may occur if seep flow is fed through macropores. Because livestock are generally drawn to water and shade during warmer months, they can often graze in riparian areas where seeps are more common and impacts can include soil compaction/erosion, devegetation, and water quality degradation (Agouridis et al. 2005). Approaches to protect riparian seep areas include riparian fencing, off-stream water sources, stream crossings, riparian buffers, and grazing management (Agouridis et al. 2005; Swanson et al. 2015). Although relationships with riparian pasture cover and increased *E. coli* have been documented (Scott et al. 2017), limited studies have evaluated seep *E. coli* transport in pasture lands (Collins and Rutherford 2004). Collins and Rutherford (2004) developed a model to simulate *E. coli* and used field measurements to illustrate elevated *E. coli* inputs from seepage areas accessed by cattle (10^4 to 10^8 MPN) during base flow and rain events. Although there are a range of studies on domesticated livestock impacts to riparian areas (Agouridis et al. 2005), less information is available on impacts by feral livestock. However, studies have shown impacts by feral hogs to seeps (FL) (Engeman et al. 2007) and feral horses to riparian areas (NV) (Beever and Brussard 2000).

In addition to bacteria, protozoa (*Cryptosporidium*) have been found to discharge at springs (Rose et al. 1991) and the authors suggested based on their results that upwelling groundwater that contains *Cryptosporidium* can present a risk of transmission of infections if the water is not treated. This and other studies suggest there is the possibility of spreading infections by groundwater seeps. For example, in Townsville, Australia, researchers found that groundwater seeps contained a bacterium linked to a fatal type of pneumonia (melioidosis) (Baker et al. 2011). They concluded that groundwater seeps may facilitate exposure to the bacterium and this may have contributed to the clustering of melioidosis in the area. This study revealed that seep exposure data may provide public health officials with guidance to implement management actions.

Another common source of contaminants to streams is wastewater (Humphrey et al. 2015), which can contain elevated concentrations of nutrients, bacteria, and pharmaceuticals. In rural settings where decentralized wastewater treatment results in wastewater inputs to the surficial aquifer, wastewater plumes that intersect and upwell at groundwater seeps may serve as a source of contaminants to seeps (Figures 5 and 6). Wastewater-impacted groundwater and its transport to seeps can deliver pharmaceutical and personal care products to adjacent surface waters. In a recent study in the Nottawasaga River Basin, ON, Canada, Spoelstra et al. (2017) evaluated groundwater wells and seeps along the banks of the river to evaluate if wastewater from local septic systems was discharging at the seeps or present in well water. They utilized four common artificial sweeteners as tracers and found those tracers in approximately 30% of the samples. For the seeps studied, 2 - 4.7% of the seeps had a septic effluent contribution of at least 1%. This study showed that pharmaceutical and personal care products associated with onsite wastewater effluent can be transported to surface waters via groundwater seeps (Spoelstra et al. 2017). In a similar effort in the Puget Sound watershed in WA, James et al. (2016) sampled approximately 20 seeps draining to the sound. They sampled seeps for a suite of emerging contaminants (including caffeine, ibuprofen, sucralose, atrazine, and others) and fecal bacteria. They found that the presence of sucralose in seep water could indicate a contribution of wastewater to the seep. At sites with known or presumed impacts by septic systems they found high detection frequencies of sucralose, acetaminophen, caffeine, ensulizole, and ibuprofen and indicated that these compounds could serve as indicators of wastewater and potential bacterial contamination. It was suggested to use more than one tracer due to the variability of septic inputs (James et al. 2016).

In urban and industrial areas, a range of organic chemicals have been found to discharge from seeps, particularly petroleum-related compounds (Humphrey et al. 2018), TCE (Chapman et al. 2007), and landfill leachate (Atekwana and Krishnamurthy 2004). Leaking underground petroleum tanks have led to BTEX compounds being transported to streams via seeps (Humphrey et al. 2018) (Figure 7). In addition, industrial solvent plumes have been shown to contaminate seeps. A detailed field study of a TCE plume at a former industrial facility in CT showed that TCE was discharging to the surface via seeps. TCE at seeps and in shallow groundwater may experience volatile organic carbon mass loss to the atmosphere, a mechanism that might also contribute to plume attenuation. TCE plume attenuation was enhanced prior to discharge to the river downgradient because of groundwater discharge to a pond and smaller streams, where some attenuation could be attributed to water-air exchange (Chapman et al. 2007).

Landfills have also been shown to contribute contaminants to seeps. Atekwana and Krishnamurthy (2004) investigated groundwater seepage to a stream adjacent to a landfill in Kalamazoo, MI. They used stable carbon isotopes (^{13}C) as a tracer for landfill leachate. Groundwater from the stream bank adjacent to the landfill and groundwater seepage into the stream showed evidence of dissolved inorganic carbon that was enriched in ^{13}C, associated with landfill leachate. This study suggested that the stream was likely affected by landfill leachate delivered via groundwater flowpaths. In another study in North Sea Harbor, NY, Gobler and Boneillo (2003) found groundwater seepage chemistry indicative of landfill leachate downgradient from an unlined municipal landfill. Groundwater seepage had elevated concentrations of ammonium, dissolved organic carbon, and low dissolved oxygen. The N-rich groundwater contributed approximately 80% of the inorganic nitrogen to the embayment. They concluded that landfill leachate upwelling at groundwater seepage areas could contribute to eutrophication (Gobler and Boneillo 2003).

Oil, gas, and coal production and use have led to seep contamination. Although naturally occurring petroleum and natural gas seeps occur in a variety of sedimentary basins (Donovan 1974; Philp and Crisp 1982; Schimmelman et al. 2018), in some cases the development activity can lead to groundwater contamination. Recent work by Woda et al. (2018) revealed that in Lycoming County, PA, leaking gas wells associated with shale gas development led to elevated methane concentrations in groundwater seeps, and they suggested that methane influx to the aquifer could lead to mobilization of groundwater contaminants such as arsenic. These and other studies suggest that greater monitoring of groundwater wells and seeps in areas of unconventional natural gas extraction may be called for (Jackson et al. 2013).

A recent study by Harkness et al. (2016) focused on evaluating the leakage from coal ash ponds in the southeastern U.S. They evaluated nine seeps adjacent to coal ash lagoons and found elevated concentrations of boron, strontium, and isotopic tracers indicative of coal combustion residuals. Overall, the seep data collected indicated that leaking coal ash ponds were impacting surface water quality. In a recent study, Brake et al. (2001) studied West Little Sugar Creek (IN) and the effects of acid mine reclamation associated with a coal mine. They found acidic seeps formed in the acid mine drainage reclamation area. The acidic effluent had low pH and several contaminants that exceeded state/or national water quality standards. They concluded that even after reclamation, the seeps and other inputs of acid mine drainage resulted in impaired aquatic ecology. Johnston et al. (2017) studied acid mine drainage from a gold and silver mine to a headwater stream in Empire, CO. They found that pH was inversely related to seep specific conductivity. Electrical resistivity imaging helped to identify seepage areas that were contributing acid mine drainage to the stream and these approaches may help to target remediation efforts. Another study in SC showed that seeps from a reject coal pile were responsible for creating high salinity, low pH conditions in adjacent soils. The low pH and high salinity resulted in vegetation dieback and limited the revegetation of the seep area (Carlson and Carlson 1994).

Other occurrences of saline seeps have been found to be naturally occurring as a result of groundwater upwelling from buried salt deposits (e.g., Manitoba, Canada; Grasby and Londry 2007) or caused by anthropogenic activities. Anthropogenic activities that can lead to elevated salinity at seeps include oil and gas activity, agricultural irrigation in arid regions, and road salt. In a study of 37 springs and seeps in Cuyahoga Falls, OH, it was found that road salt was the primary contributor to increased total dissolved solids at the springs and seeps (Foos 2003). In arid regions, salinity can be concentrated at seeps. For example, in Australia, sandplain seeps occur where salts from groundwater discharge are concentrated at the surface due to evaporation (George 1991). The salinity can affect agricultural use and vegetation growth in seep areas and a range of reclamation efforts have been attempted to reduce associated soil salinization, including interception drains and eucalyptus trees (George 1991). Overall, a wide range of studies have shown that groundwater seeps can be contaminated by a variety of agricultural, industrial, and urban contaminant sources. Recent legal cases have focused on water quality of seeps, springs, and seepage zones along navigable rivers because of their ability to transport contaminants to navigable surface waters regulated under jurisdiction of the CWA.

Emerging Legal and Policy Issues – Seeps, Groundwater, and the Clean Water Act

Since the turn of the century, numerous court cases (Table 1) have suggested that seeps with measureable impacts on adjacent surface water quality may fall under CWA jurisdiction if the contaminated groundwater that discharges at a spring or seep or through stream channel sediments is hydrologically connected to navigable waters. Numerous recent articles on the legal aspects of contaminated groundwater inputs to navigable streams have focused on recent case law and the applicability of the CWA to contaminated groundwater that is transported to navigable waters (Kvien 2015; Juilfs 2016; Smith 2016; William and Endres 2017). Although the CWA primarily regulates surface water quality (specifically point source contaminant inputs to navigable waters), courts have not ruled consistently on how to characterize the CWA's role in protecting water quality of groundwater and the relationship between groundwater and surface water (Kvien 2015; William and Endres 2017). There is an important legal question as to whether the CWA covers discharges of pollutants to groundwater that is hydrologically connected to navigable waters (Kvien 2015).

From the CWA perspective, the Environmental Protection Agency (EPA) defines a point source as "any discernible, confined and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, concentrated animal feeding operation, or vessel or other floating craft, from which pollutants are or may be discharged" (US EPA 2018a). The distinction between point source inputs from discrete conveyances and the more diffuse subsurface transport of contaminant inputs via groundwater flowpaths becomes important in cases of contaminated groundwater seeps and springs and their function of discharging contaminants to navigable waterways.

Kvien (2015) and William and Endres (2017) provided reviews of some recent legal cases that have considered how contaminated groundwater has recently been addressed under the CWA, and the range of opinions. Notable recent cases include the Northern California River Watch v. City of Healdsburg; the Yadkin Riverkeeper v. Duke Energy Carolinas LLC, and the Hawai'i Wildlife Fund v. County of Maui (Kvien 2015; William and Endres 2017) (Table 1). These cases showed that wastewater and coal ash contaminants that were stored or injected and had hydrologic connections

Table 1. Examples of recent legal cases that considered contaminated groundwater with hydrological connections to navigable surface waters to fall under Clean Water Act (CWA) jurisdiction (modified from Kvien 2015; William and Endres 2017).

Case (Year)	Basis
Idaho Rural Council v. Bosma (2001)	Unlined wastewater ponds leached contaminants into groundwater (GW) hydrologically connected to springs that were hydrologically connected to Clover Creek.
Northern California River Watch v. City of Healdsburg (2004)	Sewage from the city was discharged into a pond which was hydrologically connected to the Russian River.
Waterkeeper Alliance, Inc. v. US EPA (2005)	Challenged that EPA's Concentrated Animal Feeding Operations (CAFO) rule was unjustified because EPA does not have jurisdiction over GW. EPA agreed it can have jurisdiction when GW connects to navigable waters.
Hernandez v. Esso Standard Oil Co. (2009)	Leaky USTs leached gasoline to GW seeps hydrologically connected to a nearby stream.
Association Concerned Over Resources and Nature, Inc. v. Tennessee Aluminum Processors, Inc.(2011)	A dump polluted GW with Al, ammonium, Cl, Pb, and Mn. Contaminated GW eventually drained to a tributary of Quality Creek.
Raritan Baykeeper, Inc. v. NL Industries, Inc. (2013)	GW discharging into the Raritan River was found to contain elevated As, Cu, Pb, Ni, and Zn.
Hawaii Wildlife Fund v. County of Maui (2014)	Wastewater injection wells were shown to be connected to coastal waters via GW transport established by tracer dye study.
Yadkin Riverkeeper, Inc. v. Duke Energy Carolinas, LLC (2015)	Coal ash storage in unlined lagoons that were hydrologically connected to the nearby Yadkin River were considered point sources under the CWA.

to the nearby surface water should be considered as point source inputs under the CWA. In these cases, when groundwater flowpaths functioned similarly to discrete conveyances of point source pollutants, numerous courts ruled that those contaminated groundwater inputs should fall under the CWA (William and Endres 2017). Although numerous recent cases have shown that the CWA can cover groundwater contaminant inputs to navigable streams (Table 1), other cases have revealed differing opinions as to CWA coverage of groundwater contaminant transport to streams (Kvien 2015; William and Endres 2017). Most recently, in February 2019 the U.S. Supreme Court agreed to hear an appeal of the Hawai'i Wildlife Fund v. County of Maui case (Savage 2019).

In the future, related cases will come forward and the US EPA (US EPA 2018b) will likely clarify their position on how groundwater-transported pollution inputs may be subject to CWA regulation. An improved scientific understanding of seep-stream interactions can help provide guidance for legal and regulatory purposes. There are a range of hydrological questions that can help to better characterize the legal aspects of seeps and their influence on stream water quality (Kvien 2015). The questions can generally be grouped into two focus areas: the nature of the hydrologic connection between groundwater and navigable streams, and the nature of the contaminant transport and water quality effects (Figure 8).

Conclusions and Management Implications

A growing number of scientific and legal studies have focused on seep water quality and seep effects on stream water quality. In minimally

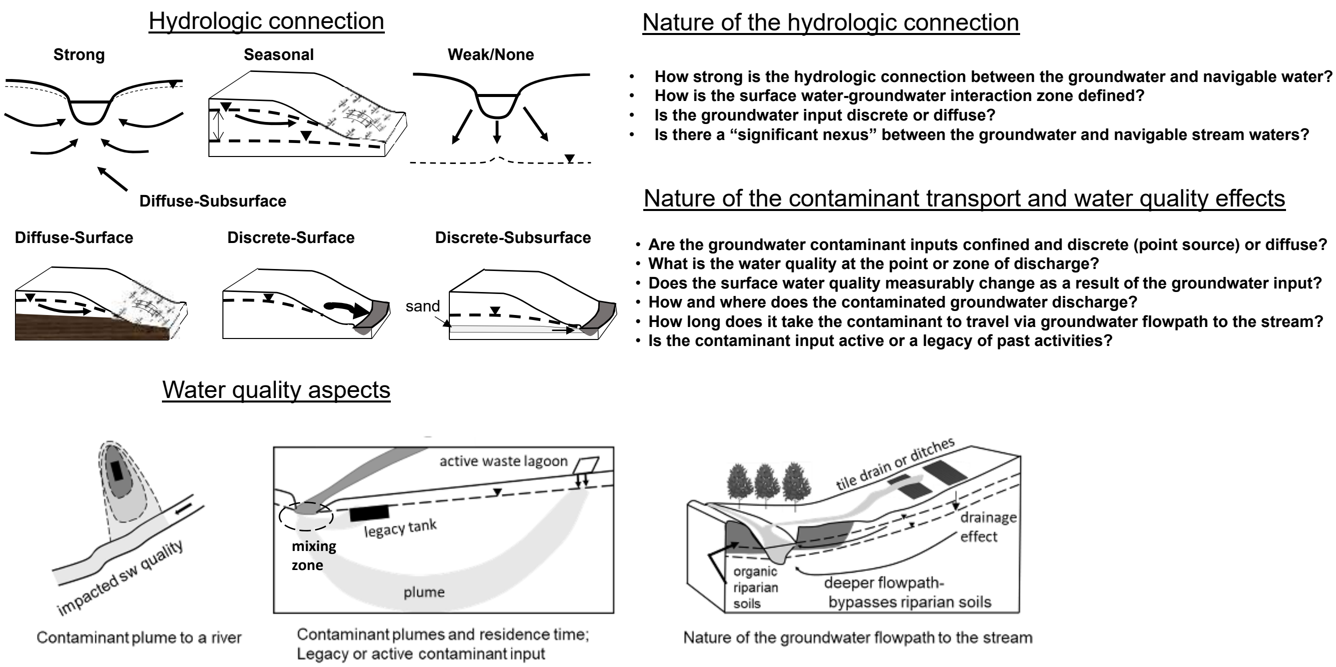


Figure 8. Hydrological questions that can help provide guidance for contaminated seeps and Clean Water Act jurisdiction. The questions can generally be grouped into two focus areas: the nature of the hydrologic connection between groundwater and navigable streams, and the nature of the contaminant transport and water quality (water quality figures modified from Heath 1983 and Puckett 2004).

disturbed forest catchments, seep attenuation of nutrients may improve downstream water quality. A wide range of studies were found that showed when human activities occur in the drainage area to the seep, seeps may act as conveyances for a number of inorganic, organic, and microbial contaminants associated with urban, wastewater, fossil fuel, and agricultural practices. In the worst-case scenarios, seeps can act as vectors for water-borne diseases and carcinogens. Although seep water quality was documented in a wide range of studies, less information was available on whether contaminated seeps measurably affected stream water quality downstream. Numerous upstream and near-seep activities may pose threats to seep flows and seep water quality including: upstream or near-seep water withdrawals, contaminant plumes, flow diversion or drainage of seep areas, and land disturbance of seep areas. Depending on the nature of groundwater flowpaths feeding the seeps, seep disturbances can be caused by land-use activities that occur far away from the immediate seep area, therefore delineating and understanding the temporal and spatial variability of the hydrological catchment area of the seep can be an important first step towards protecting the seep. Since seeps can deliver contaminants to streams, management efforts to protect water quality should consider seep setbacks to protect the upstream area draining to seeps (seep catchments) and near-seep zones.

As it has been shown that greater contaminant attenuation can occur for diffuse flow versus rivulet-pipe flow conditions, it may be possible in some settings to reduce the seep contaminant transport to the stream by using level spreaders (Winston et al. 2011) or other approaches to reduce rivulet-pipe flow conditions and enhance diffuse flow through riparian soils and vegetation. In cases where the groundwater contamination is fairly shallow, phytoremediation (Nichols et al. 2014) and forested riparian buffers (Mayer et al. 2007) may also protect the seep area and help to reduce the contaminants surfacing at the seep and flowing to adjacent streams.

More seep focused studies are needed and it is important to collect data on some of their basic properties, including the nature of their source (helocrene, limnocrene, rheocrene); discharge magnitude; water temperature; total dissolved solids; and persistence (Springer and Stevens 2009;

Williams 2016). The frequency and duration of seep flow and the diurnal and seasonal variability of seep water temperature may help distinguish seeps that have deeper, longer-term groundwater flowpaths from ephemeral seeps fed by shorter-term groundwater flowpaths. Those with shallower flowpaths may be more sensitive to local activities and climate change. An improved understanding of the nature of groundwater flowpaths to the seep may help characterize those that are vulnerable to impairment.

Although groundwater inputs often have a large influence on base flow water quality, groundwater data are not frequently included in surface water quality studies. Groundwater seeps provide an alternate low-cost method of monitoring groundwater quality when drilling monitoring wells is not practical (e.g., in mountainous terrain or wetlands) or within the budgetary constraints of a project (Soulsby et al. 2007). Additionally, seeps can serve as valuable educational tools as these are sites where groundwater is visible. Comprehensive seep and spring location information is available in a limited number of studies (e.g., Junghans et al. 2016), but in most regions seeps are not thoroughly mapped. Without watershed seep maps and baseline seep water quality and flow data, it will be challenging to understand changing conditions. Seep inventory, discharge, and water quality projects can provide a basis to evaluate shifting water quality and flow conditions over time. Several states such as Minnesota (Minnesota Spring Inventory 2019) and Kentucky (KGDR 2019) have spring mapping programs; similar efforts for seeps (including citizen science efforts) could be fruitful. The Springs Stewardship Institute has recently begun developing an online database and Springs Online program to help users locate and document springs and seeps (http://springstewardshipinstitute.org/). Because many seeps occur as wetlands, in some cases they may be mapped in the National Wetlands Inventory or other databases, such as the National Hydrography Database Plus (USEPA/USGS 2005). Field mapping might be improved using recent technologies; for example, thermal imaging may help to detect seeps (Roper et al. 2014).

From a legal and regulatory perspective, there are potentially a large number of contaminated seep sites where more research is needed to determine the hydrologic connection between the contaminated groundwater and the navigable surface water and evaluate if the groundwater inputs affect the surface water quality. These determinations can often be made with a range of field and modeling approaches including: nested piezometers, aquifer sampling and testing, water temperature and specific conductance logging, thermal imaging, geophysical surveying (ground penetrating radar, electrical resistivity, electromagnetic induction, seismic), seepage runs, nested water quality sampling, tracer studies, residence time and age dating, and surface water-groundwater modeling.

Future work on seep-stream interactions can improve understanding of the controls on: discrete vs. diffuse discharge; surface water/groundwater mixing zones; the degree of hydrologic connections; setback distances for seep protection; nature of groundwater discharge seeping into surface waters; water quality of groundwater discharge at the seep emergence point and at the point where seeps discharge to the navigable stream; magnitude and variability of groundwater residence time; seasonality of groundwater quality and discharge; influence of forested riparian buffers and hyporheic zones; and seep effects on stream water quality. Detailed seep water quality studies across a range of hydrogeological, meteorological, and land-use conditions can help improve the identification and characterization of seeps likely to convey contaminants to streams and affect stream water quality. In addition, improved understanding of seep water quality and disturbances can help in the development and testing of spring/seep ecosystem models (Springer et al. 2008; Stevens 2008; Lehosmaa et al. 2018). More work is needed to understand regional relationships between spring/seep ecological diversity and water quality (Stevens 2008).

Acknowledgments

This work was funded by the US EPA, NC Water Resources Research Institute, Pennsylvania State University, and East Carolina University. We thank Jim Watson, the ECU Environmental Research Lab, Jen Eismeier, Caitlin Skibiel, Matt Smith, Jeremy Robbins, Mark Akland, Sarah Hardison, and Jamil Blackmon for field and water quality support. Thanks to Craven Co. and private landowners for site access to seeps. We thank Tao Wen at Pennsylvania State University for providing recent information on natural gas development and the effects on groundwater seeps.

Author Bio and Contact Information

Dr. Michael O'Driscoll (corresponding author) is a hydrologist and has been a faculty member at East Carolina University (ECU) since 2004 and an adjunct Associate Professor at Duke University since 2018. Currently he is an Associate Professor in the Department of Coastal Studies. He has earned graduate degrees in Geology, Environmental Pollution Control, and Forest Resources (Penn State University). He teaches a variety of courses in hydrology, hydrogeology, water pollution, earth and environmental sciences. His research focuses on utilizing tracers and other hydrogeological, geochemical, and geophysical techniques to develop insights into the geological controls and land-use effects on surface water-groundwater interactions and contaminant transport, particularly in nutrient-sensitive Coastal Plain and Piedmont watersheds. He may be contacted at odriscollm@ecu.edu or by mail at Department of Coastal Studies, East Carolina University, 204 Graham Building, Greenville, NC 27858.

Dr. David DeWalle is a retired faculty member at Penn State University who formerly taught watershed management, snow hydrology, and forest micrometeorology courses and conducted research on a variety of environmental topics especially impacts of atmospheric deposition. He was educated in forestry and forest hydrology at the University of Missouri (BS and MS) and watershed management at Colorado State University (Ph.D.). At retirement he served as Associate Director of Penn State Institutes of the Environment and as Director of the Pennsylvania Water Resources Research Center.

Dr. Charles Humphrey Jr. is a NC Licensed Soil Scientist and Registered Environmental Health Specialist. He earned his BS (Ecosystems Assessment) and MS degrees (Soil Science) from North Carolina State University, and his Ph.D. (Coastal Resources Management) from East Carolina University (ECU). Dr. Humphrey is an Associate Professor of Environmental Health at ECU, where he conducts research on various water quality issues and teaches undergraduate and graduate classes on water and wastewater treatment. Prior to joining the faculty at ECU, he worked for seven years as an Environmental Agent with NC Cooperative Extension, and three years as an Environmental Health Specialist.

Dr. Guy Iverson recently completed his Ph.D. in Coastal Resources Management at East Carolina University and will begin as a faculty member in the Environmental Health Sciences Program at East Carolina University in August 2019. During his graduate studies, he has conducted research focusing on nutrient cycling in groundwater and surface water in nutrient-sensitive, water-supply watersheds in North Carolina's Coastal Plain and Piedmont.

References

Agouridis, C., S. Workman, R. Warner, and G. Jennings. 2005. Livestock grazing management impacts on stream water quality: A review. *Journal of the American Water Resources Association* 41(3): 591-606.

Alfaro, C. and M. Wallace. 1994. Origin and classification of springs and historical review with current applications. *Environmental Geology* 24(2): 112-124.

Atekwana, E. and R. Krishnamurthy. 2004. Investigating landfill-impacted groundwater seepage into headwater streams using stable carbon isotopes. *Hydrological Processes* 18: 1915-1926.

Baker, A., D. Tahani, C. Gardiner, K. Bristow, A. Greenhill, and J. Warner. 2011. Groundwater seeps facilitate exposure to Burkholderia pseudomallei. *Applied Environmental Microbiology* 77(20): 7243-7246.

Beck, H., A. van Dijk, D. Miralles, R. de Jeu, L. Bruijnzeel, T. McVicar, and J. Schellekens. 2013. Global patterns in base flow index and recession based on streamflow observations from 3394 catchments. *Water Resources Research* 49: 7843-7863.

Beever, E. and P. Brussard. 2000. Examining ecological consequences of feral horse grazing using exclosures. *Western North American Naturalist* 60(3): 236-254.

Blackmon, J. 2017. An evaluation of environmental health threats associated with stream discharge from Town Creek in Greenville, North Carolina. Masters Thesis in Environmental Health, East Carolina University, Department of Health Education and Promotion, Greenville, NC. Available at: http://thescholarship.ecu.edu/bitstream/handle/10342/6138/BLACKMON-MASTERSTHESIS-2017.pdf?sequence=1&isAllowed=y. Accessed April 19, 2019.

Borggaard, O. and A. Gimsing. 2008. Fate of glyphosate in soil and the possibility of leaching to groundwater and surface waters: A review. *Pest Management Science* 64: 441-456.

Boulton, A. and P. Hancock. 2006. Rivers as groundwater-dependent ecosystems: A review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* 54: 133-144.

Brake, S., K. Connors, and S. Romberger. 2001. A river runs through it: Impact of acid mine drainage on the geochemistry of West Little Sugar Creek pre- and post-reclamation at the Green Valley coal mine, Indiana, USA. *Environmental Geology* 40: 1471-1481.

Brinson, M.M. 1993. *A Hydrogeomorphic Classification for Wetlands*. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS, USA. Technical Report WRP-DE-4. Available at: https://wetlands.el.erdc.dren.mil/pdfs/wrpde4.pdf#view=fit&pagemode=none. Accessed April 19, 2019.

Burns, D.A., P.S. Murdoch, G.B. Lawrence, and R.L. Michel. 1998. Effect of ground water springs on NO_3^- concentration during summer in Catskill Mountain streams. *Water Resources Research* 34: 1987-1996.

Carlson, C.L. and C.A. Carlson. 1994. Impacts of coal pile leachate on a forested wetland in South Carolina. *Water, Air, and Soil Pollution* 72(1): 89-109.

Chapman, S., B. Parker, J. Cherry, R. Aravena, and D. Hunkeler. 2007. Groundwater–surface water interaction and its role on TCE groundwater plume attenuation. *Journal of Contaminant Hydrology* 91(3-4): 203-232.

City of Durham. 2012. Timeline of Water Quality Issues in Falls Lake. Available at: https://durhamnc.gov/DocumentCenter/View/2266/Timeline-of-Falls-Lake-Nutrient-Management-Strategy-Events-PDF?bidId=. Accessed April 19, 2019.

Collins, R. and K. Rutherford. 2004. Modelling bacterial water quality in streams draining pastoral land. *Water Research* 38(3): 700-712.

Donovan, T. 1974. Petroleum microseepage at Cement, Oklahoma: Evidence and mechanism. *The American Association of Petroleum Geologists Bulletin* 58(3): 429-446.

Engeman, R., A. Stevens, J. Allen, J. Dunlap, M. Dunlap, D. Teague, and B. Constantin. 2007. Feral swine management for conservation of an imperiled wetland habitat: Florida's vanishing seepage slopes. *Biological Conservation* 134(3): 440-446.

Fisher, D., J. Steiner, D. Endale, J. Stuedemann, H. Schomberg, A. Franzluebbers, and S. Wilkinson. 2000. The relationship of land use practices to surface water quality in the Upper Oconee Watershed of Georgia. *Forest Ecology and Management* 128: 39-48.

Fisher, J. and M.C. Acreman. 2004. Wetland nutrient removal: A review of the evidence. *Hydrology and Earth System Sciences* 8: 673-685.

Foos, A. 2003. Spatial distribution of road salt contamination of natural springs and seeps, Cuyahoga Falls, Ohio, USA. *Environmental Geology* 44: 14-19.

George, R.J. 1991. Management of sandplain seeps in the wheatbelt of western Australia. *Agricultural Water Management* 19: 85-104.

Gibson, C., C. O'Reilly, A. Conine, and S. Lipshutz. 2015. Nutrient uptake dynamics across a gradient of nutrient concentrations and ratios at the landscape scale. *Journal of Geophysical Research-Biogeosciences* 120: 326-340.

Gobler, C. and G. Boneillo. 2003. Impacts of anthropogenically influenced groundwater seepage on water chemistry and phytoplankton dynamics within a coastal marine system. *Marine Ecology Progress Series* 255: 101-114.

Gold, A.J., P.M. Groffman, K. Addy, D.Q. Kellogg, M. Stolt, and A.E. Rosenblatt. 2001. Landscape attributes as controls on ground water nitrate removal capacity of riparian zones. *Journal of the American Water Resources Association* 37: 1457-1464.

Grasby S. and K. Londry. 2007. Biogeochemistry of hypersaline springs supporting a mid-continent marine ecosystem: An analogue for martian springs? *Astrobiology* 7(4): 662-683.

Griebler, C. and M. Avramov. 2015. Groundwater ecosystem services: A review. *Freshwater Science* 34(1): 355-367.

Harkness, J., B. Sulkin, and A. Vengosh. 2016. Evidence for coal ash ponds leaking in the southeastern United States. *Environmental Science and Technology* 50(12): 6583-6592.

Heath, R.C. 1983. Basic Ground-water Hydrology: U.S. Geological Survey Water-Supply Paper 2220. Available at: https://pubs.usgs.gov/wsp/2220/report.pdf. Accessed April 19, 2019.

Hill, A. 1996. Nitrate removal in stream riparian zones. *Journal of Environmental Quality* 25:743-755.

Humphrey, C., M. O'Driscoll, D. Mallinson, and S. Hardison. 2013. *Geophysical and Water Quality Characterization of On-site Wastewater Plumes*. North Carolina Water Resources Research Institute Report No. 445. Available at: https://repository.

lib.ncsu.edu/bitstream/handle/1840.4/8292/NC-WRRI-445.pdf?sequence=1&isAllowed=y. Accessed April 19, 2019.

Humphrey, C., A. Finley, M. O'Driscoll, A. Manda, and G. Iverson. 2015. Groundwater and stream *E. coli* concentrations in Coastal Plain watersheds served by onsite wastewater and a municipal sewer treatment system. *Water Science and Technology* 72(10): 1851-1860.

Humphrey, C., J. Jernigan, G. Iverson, B. Serozi, M. O'Driscoll, S. Pradhan, and E. Bean. 2016. Field evaluation of nitrogen treatment by conventional and single-pass sand filter onsite wastewater systems in the North Carolina Piedmont. *Water, Air, and Soil Pollution* 227: 255.

Humphrey, C., J. Blackmon, T. Kelley, M. O'Driscoll, and G. Iverson. 2018. Environmental health threats associated with drainage from a coastal urban watershed. *Environment and Natural Resources Research* 8(1): 52-60.

Iverson, G., C. Humphrey, M. O'Driscoll, C. Sanderford, J. Jernigan, and B. Serozi. 2018. Nutrient exports from watersheds with varying septic system densities in the North Carolina Piedmont. *Journal of Environmental Management* 211: 206-217.

Iverson, G. 2019. Nutrient contributions from septic systems in nutrient-sensitive watersheds: Quantifying nutrient inputs, reduction methods, and economic feasibility. PhD Dissertation, Coastal Resources Management, East Carolina University, Greenville, NC.

Jackson, R., A. Gorody, B. Mayer, J. Roy, M. Ryan, and D. Van Stempvoort. 2013. Groundwater protection and unconventional gas extraction: The critical need for field-based hydrogeological research. *Ground Water* 51(4): 488-510.

James, C., J. Miller-Schulze, S. Ultican, A. Gipe, and J. Baker. 2016. Evaluating contaminants of emerging concern as tracers of wastewater from septic systems. *Water Research* 101: 241-251.

Jamieson, R., R. Gordon, K. Sharples, G. Stratton, and A. Madani. 2002. Movement and persistence of fecal bacteria in agricultural soils and subsurface drainage water: A review. *Canadian Biosystems Engineering* 44: 1.1-1.9.

Johnston, A., R. Runkel, A. Navarre-Sitchler, and K. Singha. 2017. Exploration of diffuse and discrete sources of acid mine drainage to a headwater mountain stream in Colorado, USA. *Mine Water Environment* 36: 463-478.

Juilfs, T. 2016. Muddy waters: Why polluted groundwater infiltrating navigable waters should not be excluded from National Pollutant Discharge Elimination System permitting. *Northern Illinois University Law Review* 7(2): 30-62.

Junghans, K., A. Springer, L. Stevens, and J. Ledbetter. 2016. Springs ecosystem distribution and density for improving stewardship. *Freshwater Science* 35(4): 1330-1339.

Kaur, A., D. Ross, J. Shanley, and A. Yatzor. 2016. Enriched groundwater seeps in two Vermont headwater catchments are hotspots of nitrate turnover. *Wetlands* 36: 237-249.

Kendall, C. and J. McDonnell. 1998. *Isotope Tracers in Catchment Hydrology*, First edition. Elsevier Science B.V., Amsterdam, 839 pp.

KGDR. 2019. Kentucky Groundwater Data Repository Water Well and Spring Location Map. University of Kentucky and Kentucky Geological Survey. Available at: https://kgs.uky.edu/kgsmap/KGSWater/viewer.asp. Accessed April 19, 2019.

Kvien, A. 2015. Is groundwater that is hydrologically connected to navigable waters covered under the CWA?: Three theories of coverage and alternative remedies for groundwater pollution. *Minnesota Journal of Law, Science, and Technology* 16(2): 957-1010.

Lehosmaa, K., J. Jyväsjärvi, J. Ilmonen, P. Rossi, L. Paasivirta, and T. Muotkaa. 2018. Groundwater contamination and land drainage induce divergent responses in boreal spring ecosystems. *Science of the Total Environment* 639: 100-109.

Likens, G.E. and D.C. Buso. 2006. Variation in steam water chemistry throughout the Hubbard Brook Valley. *Biogeochemistry* 78: 1-30.

Mayer, P., S. Reynolds, M. McCutchen, and T. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. *Journal of Environmental Quality* 36: 1172-1180.

McClain, M., E. Boyer, C. Dent, et al. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. *Ecosystems* 6(4): 301-312.

Miller, M., S. Buto, D. Susong, and C. Rumsey. 2016. The importance of base flow in sustaining surface water flow in the Upper Colorado River Basin. *Water Resources Research* 52: 3547-3562.

Minnesota Spring Inventory. 2019. Minnesota Department of Natural Resources. Available at: https://www.dnr.state.mn.us/waters/groundwater_section/springs/msi.html. Accessed April 19, 2019.

Morley, T.R., A.S. Reeve, and A.J.K. Calhoun. 2011. The role of headwater wetlands in altering streamflow and chemistry in a Maine, USA catchment. *Journal of the American Water Resources Association* 47(2): 337-349.

NCDENR. 1990. North Carolina Department of Environment and Natural Resources. A Report on the Investigation of the Town Creek Incident, Greenville, Pitt County- Incident #3212.

Nichols, E., R. Cook, J. Landmeyer, B. Atkinson, D. Malone, G. Shaw, and L. Woods. 2014. Phytoremediation of a petroleum-hydrocarbon contaminated shallow aquifer in Elizabeth City, North Carolina, USA. *Remediation* 24(2): 29-46.

Ocampo, C.J., C.E. Oldham, M. Sivapalan, and J.V. Turner. 2006. Hydrological versus biogeochemical controls on catchment nitrate export: A test of the flushing mechanism. *Hydrological Processes* 20: 4269-4286.

O'Driscoll, M. and D. DeWalle. 2010. Seeps regulate stream nitrate concentration in a forested Appalachian catchment. *Journal of Environmental Quality* 39: 420-431.

O'Driscoll, M.A., C.P. Humphrey, Jr., N.E. Deal, D.L. Lindbo, and M. Zarate-Bermudez. 2014. Meteorological influences on nitrogen dynamics of a coastal onsite wastewater treatment system. *Journal of Environmental Quality* 43: 1873-1885.

Philp, R. and P. Crisp. 1982. Surface geochemical methods used for oil and gas prospecting: A review. *Journal of Geochemical Exploration* 17: 1-34.

Puckett, L. 2004. Hydrogeologic controls on the transport and fate of nitrate in ground water beneath riparian buffer zones: Results from thirteen studies across the United States. *Water Science and Technology* 49(3): 47-53.

Röper, T., J. Greskowiak, and G. Massmann. 2014. Detecting small groundwater discharge springs using handheld thermal infrared imagery. *Ground Water* 52(6): 936-942.

Rose, J., C. Gerba, and W. Jakubowski. 1991. Survey of potable water supplies for *Cryptosporidium* and *Giardia*. *Environmental Science and Technology* 25(8): 1393-1400.

Rosenblatt, E., A. Gold, H. Stolt, P. Groffman, and D. Kellogg. 2001. Identifying riparian sinks for watershed nitrate using soil surveys. *Journal of Environmental Quality* 30: 1596-1604.

Rutherford, J.C. and M. Nguyen. 2004. Nitrate removal in riparian wetlands: Interactions between surface flow and soils. *Journal of Environmental Quality* 33: 1133-1143.

Santhi, C., P. Allen, R. Muttiah, J. Arnold, and P. Tuppad. 2008. Regional estimation of base flow for the conterminous United States by hydrologic landscape regions. *Journal of Hydrology* 351: 139-153.

Savage, D. 2019. Supreme Court to Decide if Clean Water Act Limits Hawaii's Underground Wastewater Dumping. Los Angeles Times, February 19, 2019. Available at: https://www.latimes.com/politics/la-na-pol-court-clean-water-hawaii-20190219-story.html. Accessed April 19, 2019.

S&ME, Inc. 2011. Surface Water Sampling Report: Town Creek Downtown Greenville. North Carolina Department of Environment and Natural Resources.

Schimmelmann, A., S. Ensminger, A. Drobniak, M. Mastalerz, G. Etiope, R. Jacobi, and C. Frankenberg. 2018. Natural geological seepage of hydrocarbon gas in the Appalachian Basin and Midwest USA in relation to shale tectonic fracturing and past industrial hydrocarbon production. *Science of the Total Environment* 644: 982-993.

Scott, E., D. Mansoor, K. Leh, and B. Haggard. 2017. Spatiotemporal variation of bacterial water quality and the relationship with pasture land cover. *Journal of Water and Health* 15(6): 839-848.

Seitzinger, S., R. Styles, E. Boyer, R. Alexander, G. Billen, R. Howarth, B. Mayer, and N. Van Breemen. 2002. Nitrogen retention in rivers: Model development and application to watersheds in the northeastern USA. *Biogeochemistry* 57: 199-237.

Shabaga, J. and A. Hill. 2010. Groundwater-fed surface flow path hydrodynamics and nitrate removal in three riparian zones in southern Ontario, Canada. *Journal of Hydrology* 388: 52-64.

Silva, S., P. Ging, R. Lee, J. Ebbert, A. Tesoriero, and E. Inkpen. 2002. Forensic applications of nitrogen and oxygen isotopes of nitrate in an urban environment. *Environmental Forensics* 3: 125-130.

Smith, B. 2016. Pollution problems in paradise: Does the Clean Water Act apply to groundwater pollution in Maui? *Journal of Environmental and Sustainability Law* 22(2): 292-309.

Soulsby, C., D. Tetzlaff, N. van den Bedem, I. Malcolm, P. Bacon, and A. Youngson. 2007. Inferring groundwater influences on surface water in montane catchments from hydrochemical surveys of springs and streamwaters. *Journal of Hydrology* 333(2-4): 199-213.

Spoelstra, J., N. Senger, and S. Schiff. 2017. Artificial sweeteners reveal septic system effluent in rural groundwater. *Journal of Environmental Quality* 46(6): 1434-1443.

Springer, A., L. Stevens, D. Anderson, R. Parnell, D. Kreamer, L. Levin, and S. Flora. 2008. Chapter 4: A comprehensive springs classification system: Integrating geomorphic, hydrogeochemical, and ecological criteria. In: *Aridland Springs in North America- Ecology and Conservation*, L. Stevens and V. Meretsky (Eds.). University of Arizona Press, Tucson, AZ, pp. 49-75.

Springer, A. and L. Stevens. 2009. Spheres of discharge of springs. *Hydrogeology Journal* 17: 83-93.

Stein, E.D., M. Mattson, A.E. Fetscher, and K.J. Halama. 2004. Influence of geologic setting on slope wetland hydrodynamics. *Wetlands* 24: 244-260.

Stevens, L. 2008. Chapter 17: Every last drop: Future springs ecosystem ecology and management. In: *Aridland Springs in North America- Ecology and Conservation*, L. Stevens and V. Meretsky (Eds.). University of Arizona Press, Tucson, AZ, pp. 332-346.

Stevens, L. and V. Meretsky. 2008. Chapter 1: Springs ecosystem ecology and conservation. In: *Aridland Springs in North America- Ecology and Conservation*, L. Stevens and V. Meretsky (Eds.). University of Arizona Press, Tucson, AZ, pp. 3-10.

Swanson, S., S. Wyman, and C. Evans. 2015. Practical grazing management to maintain or restore riparian functions and values on rangelands. *Journal of Rangeland Applications* 2: 1-28.

Tang, X., B. Zhu, and H. Katou. 2012. A review of rapid transport of pesticides from sloping farmland to surface waters: Processes and mitigation strategies. *Journal of Environmental Sciences* 24(3): 351-361.

Tesoriero, A.J., J.H. Duff, D.M. Wolock, N.E. Spahr, and J.E. Almendinger. 2009. Identifying pathways and processes affecting nitrate and orthophosphate inputs to streams in agricultural watersheds. *Journal of Environmental Quality* 38: 1892-1900.

US EPA and USGS. 2005. U.S. Environmental Protection Agency and the U.S. Geological Survey (USGS) National Hydrography Dataset Plus – NHDPlus 1.0. Available at: http://www.horizon-systems.com/nhdplus. Accessed April 19, 2019.

US EPA. 2002. Onsite Wastewater Treatment Systems Manual. Office of Water, Office of Research Development, United States Environmental Protection Agency. EPA/625/R-00/008.

US EPA. 2018a. Clean Water Act Secton 502: General Definitions. Available at: https://www.epa.gov/cwa-404/clean-water-act-section-502-general-definitions. Accessed April 19, 2019.

US EPA. 2018b. Clean Water Act Coverage of "Discharges of Pollutants" via a Direct Hydrologic Connection to Surface Water. Available at: https://www.federalregister.gov/documents/2018/02/20/2018-03407/clean-water-act-coverage-of-discharges-of-pollutants-via-a-direct-hydrologic-connection-to-surface. Accessed April 19, 2019.

Van Stempvoort, D., J. Spoelstra, N. Senger, S. Brown, R. Post, and J. Strugera. 2016. Glyphosate residues in rural groundwater, Nottawasaga River Watershed, Ontario, Canada. *Pest Management Science* 72: 1862-1872.

Vidon, P.G.F. and A.R. Hill. 2004. Landscape controls on nitrate removal in stream riparian zones. *Water Resources Research* 40(W03201): 1-14.

West, A.J., S.E. Findlay, D.A. Burns, K.C. Weathers, and G.M. Lovett. 2001. Catchment-scale variation in the nitrate concentration of ground water seeps in the Catskill Mountains, New York, USA. *Water Air and Soil Pollution* 132: 389-400.

William, R. and A.B. Endres. 2017. Bridging the divide: Incorporating interflow into legal discourse on surface water-groundwater interactions. *Ecology Law Currents* 44(1): 101-108.

Williams, M., A. Buda, H. Elliott, J. Hamlett, E. Boyer, and J. Schmidt. 2014. Groundwater flow path dynamics and nitrogen transport potential in the riparian zone of an agricultural headwater catchment. *Journal of Hydrology* 511: 870-879.

Williams, M.R., A.R. Buda, H.A. Elliott, A.S. Collick, C. Dell, and P.J.A. Kleinman. 2015. Linking nitrogen management, seep chemistry, and stream water quality in two agricultural headwater watersheds. *Journal of Environmental Quality* 44: 910-920.

Williams, D.D. 2016. Chapter 11: Invertebrates in groundwater springs and seeps. In: *Invertebrates in Freshwater Wetlands: An International Perspective on their Ecology*, D. Batzer and D. Boix (Eds.). Springer, Switzerland, pp. 357-410.

Williams, M.R., A.R. Buda, K. Singha, G.J. Folmar, H.A. Elliott, and J.P. Schmidt. 2016. Imaging hydrological processes in headwater riparian seeps with time-lapse electrical resistivity. *Ground Water* 55(1): 136-148.

Winston, R., W. Hunt III, D. Osmond, W. Lord, and M. Woodward. 2011. Field evaluation of four level spreader–vegetative filter strips to improve urban storm-water quality. *Journal of Irrigation Drainage Engineering* 137(3): 170-182.

Winter, T. 2007. The role of ground water in generating streamflow in headwater areas and in maintaining base flow. *Journal of the American Water Resources Association* 43(1):15-25.

Woda, J., T. Wen, D. Oakley, D. Yoxtheimer, T. Engelder, M. Castrod, and S. Brantley. 2018. Detecting and explaining why aquifers occasionally become degraded near hydraulically fractured shale gas wells. *Proceedings of the National Academy of Sciences* (*PNAS)*: 201809013. DOI: 10.1073/pnas.1809013115. Available at: www.pnas.org/cgi/doi/10.1073/pnas.1809013115. Accessed April 19, 2019.

Zimmer, M., S. Bailey, K. McGuire, and T. Bullen. 2013. Fine scale variations of surface water chemistry in an ephemeral to perennial drainage network. *Hydrological Processes* 27: 3438-3451.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 166, Pages 79-94, April 2019

Cultural Narratives on Constraints to Community Engagement in Urban Water Restoration

*Amit Pradhananga[1], Mae Davenport[1], and Emily Green[2]

[1]Department of Forest Resources, University of Minnesota, St. Paul, MN
[2] Center for Changing Landscapes, University of Minnesota, St. Paul, MN
*Corresponding Author

Abstract: Natural resource professionals increasingly recognize that water protection and restoration efforts require not only technical solutions, but also the active engagement of stakeholders who live and work in the local community. People of color, and those of lower income brackets, are frequently underrepresented in water-related programming or decision-making, although they are often disproportionately affected by water problems. Effective engagement of diverse community members in water programs and projects requires understanding and addressing constraints to action. We conducted 25 interviews with community members who live or work in a highly urbanized Minnesota watershed to explore perceived obstacles to community engagement in local water resource protection and restoration. Based on self-reported race, ethnicity, and general community engagement level, interviewees were assigned to one of three "stakeholder groups" for comparative analysis: formal decision-makers, active white community members, and active community members of color. Qualitative analysis of responses revealed perceived constraints to engagement common to all three groups: inaccessibility and invisibility of water, lack of local leadership in water issues, and limited community dialogue about water problems and solutions. Additional constraints were perceived uniquely by community members of color: cultural constraints around water uses, recreation, action, and inequities or disenfranchisement in community decision-making processes and water programming. Study findings suggest partnership building is needed for collaboration in designing civic engagement programs and improving water protection and restoration projects.

Keywords: *watershed protection, stakeholder engagement, public participation, social disparities*

Healthy lakes and streams can greatly benefit urban communities by fostering community identity, boosting local economies, and improving residents' quality of life. Urban water resource managers increasingly recognize that protecting and restoring healthy water requires not only careful land and water management, but also the engagement of community stakeholders to support funding and implement plans. Unfortunately, fostering meaningful and inclusive community engagement in planning *processes* has been a challenge for water and land resource managers (National Research Council 2008). Moreover, the populations most vulnerable to environmental risks are also least likely to be engaged and represented in natural resource decision-making processes (Sarokin and Schulkin 1994; Moraes and Perkins 2007; Larson and Lach 2010; Phadke et al. 2015). Not surprisingly, research shows that people within dominant social groups (e.g., men, middle aged, homeowners, and higher income and education levels) are more engaged in water issues than their counterparts (Koehler and Koontz 2008).

Research shows public participation in water resource planning and management can have multiple ecological and cultural benefits. Participatory water resource management enhances implementation of water plans (Lubell 2005; Sabatier et al. 2005), increases community support for long-term planning (Selfa and Becerra 2011), bolsters public funding for water programs (Larson and Lach 2008), and builds social capital, or networks of community influence (Prokopy and Floress 2011). Public participation in water planning can increase public trust in and perceived legitimacy of planning processes (Trachtenberg and Focht 2005). Participatory processes also have diffused community tensions around environmental problems and policy interventions (Fraser et al. 2006). Questions persist around what communities are *excluded* from or underrepresented in planning processes and why. Planning processes that treat the public as having a singular unified interest fail to recognize different voices, empower diverse leaders, or inspire collective and sustained action (Lane 2005). In the case of urban water planning and management, narratives of the cultural constraints to civic engagement have been largely absent from the literature.

Research shows that communities of color and low-income communities face unique cultural constraints to engagement in environmental issues. The environmental justice literature points to a broader set of socio-political and institutional constraints to racial and ethnic minority community members' engagement in environmental issues, including the separation of "environmental" from "social" issues (Di Chiro 2008). Communities facing pressing social issues (e.g., employment, poverty, housing, immigration) commonly prioritize those issues over environmental problems (e.g., Gibson-Wood and Wakefield 2013), especially if institutions separate environmental and social issues.

The structure and method of a public participation opportunity may constrain diverse community engagement. Conventional methods of public participation (e.g., formal meetings) may exclude marginalized communities. For example, a study of environmental participation among communities of color in the United Kingdom found that the formality of facilitated, local sustainability meetings was a constraint to public involvement. This same study found that people of color were more involved in community-oriented events, rather than environment-oriented events (Clarke and Agyeman 2011). Another study focused on the engagement of Hispanic communities found that formal approaches to public participation were not accessible to the broader Hispanic community (Gibson-Wood and Wakefield 2013). Participants may also lack the confidence to express themselves in formal settings, and their contributions may be viewed as unrelated and unhelpful (Pothier et al. 2019). Further, participation also involves real costs (e.g., transportation, childcare costs to attend meetings) that may differentially affect lower income community groups (Wakefield and Poland 2005).

Closer to our study area, researchers investigated water-related perceptions and behaviors in Minnesota's Hmong community (MWMO and City of Minneapolis 2007). Findings suggest that the Hmong community faces multiple institutional and communication barriers when it comes to accessing water use information. These barriers inhibit community members' awareness of environmental problems and risks, as well as their causes, consequences, and solutions. Conventional modes of water communication (e.g., print materials, websites) often do not take into account cultural preferences for communication (e.g., oral, inter-personal). Language barriers emerged as a major obstacle for Minnesota's Hmong community members.

More recently, the concept of recognition has gained prominence in the environmental justice literature. Recognition of whose experiences and knowledge is included and excluded in the way the environmental values and problems are defined or prioritized can also be a constraint to marginalized communities and their engagement in water programs or projects (e.g., Schlosberg 2004, 2007). Lack of recognition denies an equal voice to those who define and experience the environment in ways that are different from the dominant culture (see Gibson-Wood and Wakefield 2013).

Study Context

Multiple waterways in the Minneapolis-St. Paul Metropolitan area (Twin Cities) of Minnesota have been shown to be seriously impaired or at risk (U.S. EPA 2018). The natural hydrology of the area was profoundly altered during the mid-20th

century building boom, resulting in substantially increased vulnerabilities to flooding and pollution (MCWD 2017, 2018). The 22-mile Minnehaha Creek experienced serious impairments stemming from industrial, residential, and transportation development within the watershed. Land use changes, building construction, and increased impervious surfaces within the watershed have led to creek channeling, habitat loss, and decreased base flow, limiting many of the stream's ecosystem services, especially cultural services (e.g., spiritual, aesthetic, recreational, educational, human health, and social cohesion) (MCWD 2018). The creek is listed on the state's Impaired Waters list (U.S. EPA 2018) for excess chloride, fecal coliform, and biotic community impairments.

The Minnehaha Creek Watershed District (MCWD) is a local unit of government with taxing authority. It is charged with the management and protection of water resources within the watershed. The MCWD has made significant investments to protect, enhance, and restore water quality through large-scale capital improvement projects including habitat restoration. Over the last decade, the MCWD has remeandered the mainstem stream channel, restored adjacent wetlands, and constructed new stormwater management facilities (MCWD 2018). Yet the MCWD acknowledges that engineering alone is not sufficient to achieve watershed-scale protection and restoration. Recent comprehensive plans emphasize integrated approaches to management, including the need for "an informed and engaged constituency" to support their water protection strategies (MCWD 2018). Given this prioritization, the MCWD sought insight on how to better engage the diverse community members who live and work in the watershed so as to inform their efforts to achieve implementation goals.

In 2012, the researchers collaborated with the MCWD to assess community capacities for, and constraints to, engagement in watershed protection and restoration projects along the highly urbanized Reach 20 segment of the Minnehaha Creek. Reach 20 spans three municipalities: St. Louis Park, Hopkins, and Edina. Our specific study objective was to explore community member perspectives on constraints to community engagement in water resource protection and restoration.

Methods

Study Area

The Minnehaha Creek watershed encompasses eight major creeks, 129 lakes, and thousands of wetlands; it spans 178 square miles from Lake Minnetonka to downtown Minneapolis. The watershed is divided into 11 subwatersheds, and partially or wholly contains 27 municipalities and two townships. The region includes several water bodies of recreational and cultural significance, including Minnehaha Creek, Lake Minnetonka, the Minneapolis Chain of Lakes, and the iconic Minnehaha Falls, one of the state's most visited attractions (Figure 1) (MCWD 2018), and a sacred site within the ancestral lands of the Ocheti Sakowin (Dakota) People (MPRB 2019). The watershed population is estimated at more than 300,000 with a projected growth of 24% in the next two decades (Metropolitan Council 2012). Population densities are highest in the lower reaches of the watershed, which include Minneapolis's urban core. The lower watershed's population is more racially and ethnically diverse with significant clusters of Hispanic, Hmong, Somali, Ethiopian, and other non-Hispanic ethnic groups (e.g., Asian Indian, Chinese). Municipalities in the upper watershed have higher median household incomes (e.g., Shorewood and Minnetrista exceed $100,000) than municipalities in the urbanized lower watershed (e.g., Hopkins is less than $50,000) (U.S. Census Bureau 2010).

Data Collection and Analysis

We gathered data through 24 key informant interviews with 25 community stakeholders. An initial list of stakeholders, including water resource professionals, government officials, and community actors (i.e., people with leadership roles in community organizations or businesses) within the communities of St. Louis Park, Hopkins, and Edina, was developed through internet searches and discussions with MCWD staff. We then used a chain referral sampling technique (Miles and Huberman 1994) to expand and diversify the sampling frame. Participants were contacted by phone or email and were offered a $50 cash incentive for participation. First, we recruited formal decision-makers (FD) (e.g., government

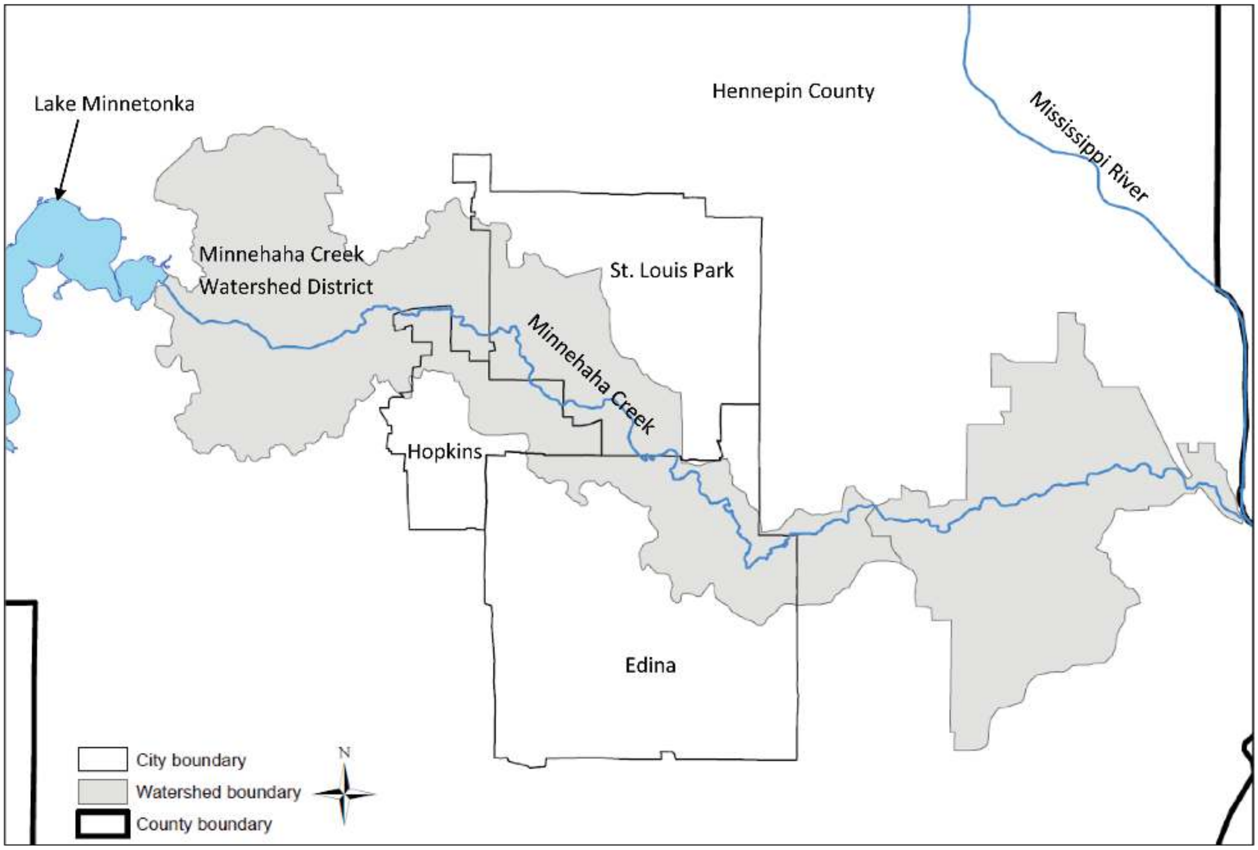


Figure 1. Study sites.

officials) engaged in (or responsible for) water resource protection and restoration activities in the study area and community members active in water resource and other community issues, often from local organizations and businesses. After preliminary analysis, it was clear that the sample underrepresented community members of color (CMC), a population that had been historically excluded from watershed planning. Thus, we intentionally recruited CMC who were active in community organizations or participated in community meetings and events.

Interviews were conducted at participants' homes, places of work, and in public spaces (e.g., coffee shops, libraries) and ranged from 45 minutes to two hours. Standard procedures of informed and voluntary consent were used to protect participants (University of Minnesota IRB #0609E92806). Interviews were audio-recorded and transcribed verbatim. Interviews were semi-structured (Brinkman and Kvale 2015) with the interviewer following scripted questions, including 21 primary questions (Appendix 1), but also allowing unscripted probing for clarity and meaning. Participants also were asked to complete a short background survey consisting of basic sociodemographic questions (e.g., age, gender, occupation, race, education, organizational membership). Sampling was limited by funding resources, though it continued until we reached what we believed was sufficient theoretical saturation (Charmaz 2006; Corbin and Strauss 2008) around our research questions. While new theoretical insights may have been gained from further data collection, we determined the richness of our existing data and diversity of narratives captured would offer water managers and community actors with important insights.

Data were analyzed using an adapted grounded theory approach consistent with Charmaz (2006). First, we assigned labels or codes to all meaning units including words, sentences, or paragraphs that represent a distinct idea or belief. Next, we organized the codes into broader themes or categories (Saldana 2009). The themes were used to develop sets of participant narratives. Analysis was performed using QSR International's Nvivo 10 software. Constant comparison was conducted between stakeholder groups to identify common and unique perspectives on community engagement in water resource protection. Theme and stakeholder group attribution were tracked throughout analysis.

Results

Participants' age, years of residence in the watershed, formal education, and occupation varied. Participants' roles in the community included government officials or employees, business owners/operators, community organization leaders, civically active residents, and educators. Nineteen of the 25 interviewees were residents of St. Louis Park, Hopkins, or Edina (Table 1). For comparative analysis of water narratives, participants were assigned to one of three "stakeholder groups" based on reported race and ethnicity, and engagement in water or community issues: 1) FD (n=7), 2) active white community members (WCM) (n=11), or 3) active CMC (n=7) (Table 2). Participants in the FD group described their connection to the community through their professional roles in local government (e.g., city manager, planner). FD participants generally described a high level of engagement in water resource protection and restoration activities. Active WCMs described being connected to the community through the work they do in community organizations, neighborhood associations, (e.g., block leader, school board member), or local businesses. WCMs were engaged in water resource protection and restoration through local organizations and neighborhood associations. Active CMCs described their connection to the community as associated with their ethnic group, the work they do in the area through organizations, and as residents participating in local events or meetings (e.g., community organization leaders, educators). Although involved in other community activities, CMC participants had limited engagement in water resource protection and restoration activities.

We present study findings on constraints to community engagement in water resource protection along five predominating narratives (Table 3). Narratives 1 and 2 were conveyed by all stakeholder groups, narrative 4 by FDs and active WCMs only, and narratives 3 and 5 were unique to CMCs.

Narrative 1: The Community Lacks Awareness about Local Water Issues

Participants from all stakeholder groups spoke about a perceived widespread lack of awareness of water problems and limited connections to local water resources as key constraints to community engagement; some also referenced this as a personal challenge. Several opined that local water issues receive little attention because there is no perceived connection or threat to drinking water supply. A CMC explained, "I cannot tell whether [the community is] really facing water problems here, because as long as [drinking water is fine], no one will know."

A FD suggested that many community members have little awareness of the "impact of water quality on their lives." Several participants contemplated why awareness is low. One FD asserted that the "ways in which water quality affects people is often invisible." Another FD communicated their sense of the broader community's oblivion to serious local water quality impairments: "the actual levels of the chlorides in the creeks and the ponds, if they understand how bad it is getting, it's getting to the point where it's killing fish and making water stagnant." Meanwhile, a WCM admitted that water quality is a personally "very intimidating subject," suggesting that the complexity of the topic may hinder interest and awareness.

Some participants bemoaned water inaccessibility in their communities. Though the Minnehaha Falls are a locally prominent and beloved water feature, the creek is not a perceptible landscape feature in the Reach 20 area. A FD conceded, "Right now in this area, you don't even know where Minnehaha Creek is. You can't see it from any of the roads. It's back behind a lot of industrial-commercial businesses." Similarly, several participants described the creek as "covered up." Participants also agreed that despite being a water-rich region, water is not "central to the community identity" in the Reach 20 corridor. A FD added, "Besides a couple small lakes, water doesn't make up as big of a proportion, as visible of

Table 1. Study participant profile.

Sociodemographic characteristic		
Gender	Male	13
	Female	12
Race	White	18
	African American	1
	Somali	3
	Ethiopian	1
	Indian	1
	Chinese	1
Age	Minimum	26
	Maximum	61
Years of local residence	Minimum	Non-resident
	Maximum	52
Formal education	Completed high school	1
	Associate degree or vocational degree	1
	College bachelor's degree	6
	Completed graduate degree (Masters or Ph.D.)	10
	JD	1
Occupation	Government	7
	Business	3
	Organization/Association	5
	Resident- apartment	7
	School/Education	3
City/County	St. Louis Park	11
	Hopkins	9
	Edina	2
	Others	3

Table 2. Stakeholder group characteristics.

	Formal decision-makers	**White community members**	**Community members of color**
No. of participants	7	11	7
Ethnicity	White	White	Somali, African American, Chinese, Ethiopian, Indian
Primary connection to community	Professional	Organizations and associations	Participation in community events
Role/Position	Water resource professionals, government officials	Resident, business owner, leadership positions in organizations	Community advocate, resident
Engagement in water resource issues	Engaged in professional capacity	Engaged through organization activities	Limited engagement

Table 3. Constraints to community engagement in water resource protection. FD = formal decision-makers. WCM = white community members. CMC = community members of color.

Theme *Descriptors*	**Stakeholder Group** **FD**	 **WCM**	 **CMC**
Narrative 1: Water is an invisible and inaccessible community resource			
Lack of awareness of water issues *Community members lack awareness of water resource problems, impacts of water pollution, consequences of their actions on local water resources, and their own connections to water.*	x	x	x
Complexity of water resource problems *Water quality is difficult to define and can be an "intimidating subject."*	x	x	
Limited visibility and accessibility of water resources *Water resources are not a visible and central part of the community's landscape; Negative perceptions of the creek (i.e., as "a swamp").*	x	x	
Narrative 2: Water discourse lacks community relevance			
Ineffective communication about water issues *Water resource issues are not discussed in the community; community leaders do not address water resource issues; water resource issues are not linked to other community issues.*	x	x	x
Language barriers *Language barriers exist in communicating issues with the community.*		x	x
Narrative 3: Culture shapes water uses, values, and civic engagement			
Recreation styles *Recreational use of water resources varies across cultural groups. Boating, swimming, or fishing for recreation (e.g., non-subsistence fishing) may not be common practices in certain ethnic groups.*			x
Communication styles *Some community members of color are not outspoken because of cultural differences in communication styles or language barriers.*			x
Cultural integration *Adapting to new cultural norms around water takes time. Perceptions of water and water issues vary based on cultural uses, water conditions in country of origin.*			x
Strained intercultural relationships *Lack of understanding and trust between community members of different racial/ethnic identities affects engagement.*			x
Narrative 4: Water management is complex and uncoordinated			
Multiple authorities/property owners *There are too many organizations and too many rules around water resources; lack of clarity exists in property ownership along the creek.*	x	x	
Lack of coordination *Lack of coordination between multiple jurisdictions in addressing water resource issues.*	x		
Narrative 5: Community members of color are disempowered in decision-making			
Civic engagement not inclusive *Water plans, projects, and programs are not inclusive of community members of color.*			x
Community needs not addressed *Needs of communities of color (e.g., transportation, child care, basic cultural differences) are not addressed in civic engagement efforts.*			x
Lack of decision-making power *Community members of color are underrepresented in organizations with decision-making authority or with influence on decision-making.*			x

a proportion, of the [geographic] community. And, [it's] just not as central to the community identity as some of the lakeside communities [nearby]." Several participants linked the physical and visual inaccessibility of water to reduced awareness of water problems. A FD participant reflected:

> *If you're not an outdoors person and you don't live on the creek in St. Louis Park, a lot of people might not even know it's there. They don't really see it on a day-to-day basis. So that's probably the biggest issue, awareness of what types of runoff impact the quality of water and how that filters into the system. I think that's better than it was 20 years ago, but I'm sure there's a lot of people that don't get that connection between fertilizer running into the storm sewers and that ultimately getting to the creek.*

Narrative 2: Water Discourse Lacks Community and/or Personal Relevance and Investment by Local Leaders

Participants in each of the stakeholder groups characterized communication about water resource issues by local leaders as ineffective and a constraint to community engagement. When asked about community engagement in water resource protection, several participants expressed concern about the lack of community leaders who are engaged in water issues. A WCM believed community leaders should play a more active role in guiding community dialogue:

> *I think we need to engage our leaders to be addressing [local water quality goals] more. I don't think that it's talked about much. I think it should be something that we can have upfront like at community gatherings, such as the Raspberry Days, things like that…have booths or something where you're interacting with the public.*

CMCs expressed similar concerns about a lack of community discussions around water. One CMC stated, "I never see [community leaders] talk about water. They never talk about water." To illustrate how important local leaders are in guiding community member engagement, a CMC used an analogy of a school principal's role in setting the tone of a school's environment: "It's kind of from-up-to-down thing. So if the [school] principal doesn't care, we don't care as well."

The way in which water issues are framed also appears to influence community engagement. One WCM stressed that when "the issue of water resource or pollution… is presented in a way that doesn't connect with [community members'] lives, it will be hard to make progress on that issue." Similarly, other participants emphasized the need to make water communications personally relevant to people. A FD elaborated:

> *What isn't helpful is when we hear about a certain species that no one's ever engaged with. Trout, that would be a species that we could all get behind, but if it's a slimy mud flea or whatever, and we just don't have enough of them, the biotic integrity just isn't there, that's hard for people to understand. It might be the right move. It might be a natural resource service and the habitat side that we want to get…but man, when you come at them with the chemistry equations, and you come at them with the scientific names of the little bugs that you don't see in the stream because it's not a healthy one, I think people just kind of glaze over.*

Beyond message framing, language barriers were a distinct and significant challenge in water communication for several CMC participants. A CMC participant offered an example of typical communications they receive about upcoming meetings: "If you knock the door and say 'Hey, this is a letter, it's a project, you need to come attend this meeting,' maybe I don't understand English and I don't understand you, I just took the letter and say 'oh, thank you.'"

Narrative 3: Culture Shapes Water Uses, Values, and Civic Engagement for Community Members of Color

CMCs explicitly identified cultural factors as constraints to their own engagement in water resource protection. For many of their community members, cultural heritage and experiences shape their interactions within their communities and their connections to water. CMC participants identified their primary use of water is for household purposes, including drinking, cooking, and washing. Several CMC participants stressed that water-related recreation is not consistent with their cultural traditions, practices, or lifestyles. When asked about use of the creek, a CMC said succinctly, "No, I don't go down the creek in a canoe. It's not part of my culture." Another CMC suggested that her upbringing has influenced her use of the creek. She explained, "If you didn't have water around you growing up maybe, you haven't developed that culture." Similarly, adjusting to new cultural norms in water recreation can be particularly difficult for women and for older generations. A CMC explained,

> *You wouldn't see a Somali person diving in, especially women because we have not learned to swim into lakes. You don't have that training as a kid, and back home you may take a chance to swim [in an area that has rainfall], but you're not going to drown. ...But here because everything has to be structured, you have to learn how to swim, wear the better dress, better swimming suits. Somalis will not, most of them, my generation will not wear a swimming suit and go into the lake.*

CMC participants also referenced cultural factors as constraining their participation in public water protection dialogue. CMC participants characterized their communities as not "vocal" about water issues. A CMC member attributed limited engagement in water issues to her "cultural upbringing":

> *Ethiopians in general… our culture, I believe hinders us. If you take the Somali culture, they're more [out]spoken, they're more visible. Whereas Ethiopians are more subdued and kind of in the background. And, I attribute that to our cultural upbringing. So maybe that has to do with that, of us not standing up and facing those issues and resolving it, maybe.*

A lack of engagement is further fueled by strained intercultural relationships. Participants portrayed community members' distrust in the dominant culture as a result of the dominant culture's limited intercultural understanding and history of oppression. A CMC participant explained:

> *It's trust, and that trust comes in with… "You hear what my needs are, and I want you to help me get there," or "Let's partner." "Don't just use me to get your agenda across." So then there is that kind of suspicious thing in our area, which is, I think, something normal. When you're a minority of the area and people don't understand who you are, they have their own little bias, so we have ours as well.*

Narrative 4: Water Management is Complex and Uncoordinated

Participants from the FD and active WCM stakeholder groups believed that a lack of clarity around water management in the watershed is a constraint to community engagement. WCM participants noted that they felt put off by the complexity of management and strategies, as multiple agencies, organizations, and businesses appear to have varying responsibilities, goals, and interests in water. In addition, the Minnehaha Creek flows through several municipalities and several participants expressed uncertainty about "who owns the land" and "who has jurisdiction." Balancing the interests of multiple agencies and organizations is a clear challenge. A WCM participant described this in the context of a nearby lake (outside of study area) and that lake's management:

> *The most challenging aspects are just the sheer number of agencies and organizations that have their fingers in the lake. Lake Minnetonka is probably the most highly managed or highly…regulated lake in the state of Minnesota. It's got several state agencies like all lakes do- Minnesota Pollution Control Agency, Department of Natural Resources, Department of [Agriculture], and probably a few others that I'm not thinking of…whose programs and regulations affect the lake. There are 14 cities around the lake, a couple of park districts and many businesses and non-profits all with similar interests most of the time, but many with competing or opposing interests as well. And balancing all that to get things done is challenging.*

Some FD participants recognized that the state of Minnesota has an "organizational infrastructure" in place through city, county, and watershed-wide plans. However, they also lamented the lack of cross-jurisdictional coordination and collaboration to address water resource issues. A

FD questioned the value of having multiple plans and organizations in addressing problems in an expansive geographical area:

We have 11 organizations in Hennepin County, and they don't talk to each other very much, and we have cities, they're in four different watershed organizations… We have a system where everybody's generating plans. We've got 11 watershed management plans, we have all these local water plans, and still we're not addressing the fact, well how do you? Over a larger geographical area, how do you set priorities? How do you implement? How do you allocate resources?

Narrative 5: Community Members of Color are Disempowered in Decision-making

According to CMC participants, lack of representation in community decision-making processes generally, is a significant constraint to their water engagement. Participants emphasized that a strong motivation to be engaged in community issues exists in communities of color. A CMC participant noted her community's strong desire to be engaged while acknowledging feeling outside the decision-making "circle":

We actually know what we want to do. We actually know where our needs are. I want to be able to be in the circle where decisions are made, and I will help you make the decision… ones best for us… I think some people call it discrimination, but I call it…a challenge. But one of these days we'll get through it. Somebody has to do it, right?

Several CMC participants expressed ongoing frustration that their communities are not taking part in the water dialogue. A CMC observed, "We get water, we drink it…it's not been part of our dialogue, it's never been. But I think it should be." Another CMC participant stressed the importance of engaging CMCs as program planners and designers rather than simply end users:

People get used to telling us what to do, or bring in programs into our doorstep, but we're never are part of the planning. So then if you're not part of the planning, nobody knows how you… your feedback's not there. Your ideas [are] not there. Then if you don't have the conversation …we're not part of the dialogue. So that's the biggest barrier.

According to CMC participants, not being meaningfully engaged in dialogue has led to weak programs or disparities in resource distribution. For example, a CMC noted that multiple requests from the Somali community for a community center have been ignored:

We ask a lot of times, many times to have a center for the community, Somali community… to learn the culture or whatever, teach kids language. They don't answer. So that's why everybody say "Oh no, they're same thing." So last five years …they ask us something, used to ask us, then when they say "What do you want as a community, what do you need?" and then we never see something.

Fueled by frustrations over historic oppression, many CMCs may reject any new programming that is not designed specifically for their community:

[Agency or organization leaders] start the intervention, and the intervention does not fit us because we're not the community that that program was developed [for]. Then immediately the rejection happens, and that's why everything that's happening is ineffective because the program is not catered to us. It was not for us, it was for the general population, and we don't fit that category.

Discussion

In this study, we interviewed 25 community members in the MCWD regarding their views on water engagement and we documented five key narratives on engagement constraints. Narratives 1 and 2 were conveyed by participants from all stakeholder groups, narrative 4 by FDs and active WCMs only, and narratives 3 and 5 were conveyed uniquely by CMCs:

1. The community lacks awareness about local water issues.
2. Water discourse lacks community and/or personal relevance and investment by local leaders.
3. Culture shapes water uses, values, and civic engagement for community members of color.

4. Water management is complex and uncoordinated.
5. Community members of color are disempowered in decision-making.

These narratives are significant because they serve not only as cultural stories, but also as cultural worldviews that frame and impede water action. They reflect varying water beliefs, social and cultural norms, attitudes, and behaviors. Comparative analysis of comments by participants from all three stakeholder groups (FDs, active WCMs, and active CMCs) identified areas of convergence as well as areas of clear divergence in perceptions and lived experiences associated with water and community engagement.

Common ground emerged around water communication and community awareness of water issues. Specifically, lack of awareness about local water resource problems and ineffective communication about water by local leaders were common themes across the three stakeholder groups. According to participants from all stakeholder groups, there is a need for local leaders to put greater focus on water issues. Respected leaders in the community have the ability to stimulate community member engagement and activate a currently absent dialogue about water issues among community members. Participants also stressed the need to focus water discourse on dimensions that connect to the real issues and values of community members, such as drinking water. FD and WCM participants also perceived that community members are not motivated to engage in water protection because local water is largely unseen and inaccessible. FD and WCM participants believe that the complexity of water management, including roles and jurisdictions, has stymied public participation in water planning and priorities.

In our view, the emergent FD narratives reflect the archetypal "urban water manager" or synoptic planner who frames public participation as a matter of raising awareness and educating citizens about expert-driven water goals. Lane (2005) characterizes this approach to public participation as tokenistic and a product of assumptions that the public interests are homogenous. In our study, FDs located constraints to community engagement as being 1) within the community: the community is physically and intellectually disengaged from water, or 2) within the nature of water management: water management is too complex and confusing for the community to be engaged. Though participants from all three stakeholder groups stressed that the community lacks awareness of water issues, CMC participants were forthcoming about institutional barriers in water communication, cultural insensitivity of participation opportunities, and historic oppression of people of color in decision-making. CMC narratives were tied to broader socio-economic and cultural context and programmatic inequities.

Two emergent narratives were unique to CMCs: the role of culture in shaping community-water interactions, and inequities in decision-making that specifically disadvantage or disempower CMCs. Culture was central to CMC participants' discussion of community engagement constraints including cultural differences in water-based recreation, heterogeneity within and across ethnic groups, the challenge of adapting to new cultural norms for recent immigrants, and limited cross-cultural understanding and competencies of the dominant culture. Similar work in Minnesota has shown that language barriers, limited access to culturally relevant water recreation, and cultural differences in water recreation are barriers to engaging some communities of color in water management (e.g., MWMO and City of Minneapolis 2007; Davenport et al. 2016). Research has shown high levels of engagement in social issues such as housing, employment, health, and immigration among CMCs (e.g., Mohai and Bryant 1998; Clarke and Agyeman 2011) and lower levels of engagement in environmental issues. This trend was echoed in narratives captured in this study. Water management efforts that lack cultural or social relevance are less likely to be successful (Di Chiro 2008).

Finally, CMC participants referenced the lack of representation in community decision-making or leadership as a significant constraint to their community's engagement in water issues. Participants spoke candidly about the exclusion of their communities in programmatic design or project planning, limiting their sense of ownership in water programs and projects, and fueling frustration and detachment from water issues. While CMCs acknowledged community willingness to engage in issues, they also want to be part of the decision-making process, and not mere recipients of programs. In watershed planning, perceived fairness in the decision-making process enhances trust among stakeholders (Leach and Sabatier 2005), increases perceived legitimacy of planning processes (Trachtenberg and Focht 2005), and leads to greater satisfaction with and acceptance of decisions and confidence in decision-makers (Lind and Tyler 1988). Study findings suggest that lack of representation and decision-making power is a significant constraint to the engagement of diverse, underrepresented groups in water resource protection. As one CMC participant in this study explained, the lack of representation and decision-making power can lead communities of color to become disengaged and to reject community programs.

In addition to issues of procedural fairness, this study also shows that the lack of recognition (Schlosberg 2004) of the experiences, values, and voices of marginalized communities can be significant constraints to their engagement. Lack of recognition denies an equal voice to communities of color in community planning and decision-making, and can fuel their frustration with the planning process. This "frustration effect" (Lawrence et al. 1997) among CMC participants stems from past experiences with attempting engagement in community events and meetings in which their needs and concerns were not taken seriously.

While this study documents important constraints to community engagement for communities of color, it is important to note here that "communities of color" are not a homogenous group. There could be critical differences among ethnic groups that this study does not capture. While examining interethnic differences in water engagement is beyond the scope of this study, it is an important area for future research.

Conclusion

We believe several important recommendations can be drawn from the narratives that could improve water protection. Chief among them is to re-envision the approach to community engagement, from a top-down, agency-driven approach to a community-driven approach. Active forms of public participation create community partnerships, and allow for greater levels of community involvement in decision-making (Arnstein 1969). This is particularly important when engaging traditionally underrepresented communities. CMCs expressed a willingness to engage in water issues. However, they also want their voices represented in community decision-making. Thus, the community should drive engagement process design and definitions of success. Of utmost importance is to listen carefully to CMC concerns, and to take active steps to address those concerns, even if those concerns are not perceived to be "environmental" or "water-related" by resource managers.

CMCs should be included early on in the engagement process in defining local community problems, rather than being informed about and asked to participate in community interventions that do not represent their perspectives and concerns. As one CMC participant explained, negative experiences with agency-driven community interventions can lead to rejection of community programming and a general distrust of agencies. There is a need to build and regain trust. An important step in a new community engagement approach will be to build trusting relationships with communities of color through trusted and respected minority group leaders and existing community institutions such as community centers and places of worship.

While CMCs were not highly engaged in water issues, they were engaged in other community issues (e.g., health, education). Water managers should reflect on the linkages between water and expressed community needs around housing, transportation, immigration, workforce development, youth mentoring, or parks and trails access. Which community-based organizations are having success in these areas and how might water managers best partner with these organizations to build mutual capacity? As past research suggests, the segregation of environmental from social issues (e.g., Di Chiro 2008) can be a barrier for community engagement among CMCs. Strategies that connect water issues with broader community issues are more likely to resonate with local communities, particularly CMCs. In a community-driven approach, rather than defining and leading engagement efforts, managers could play the role of supporting culturally inspired and community-led public events to help build collaborative relationships and trust. Building trust is a long-term commitment. Managers should prioritize and incentivize relationship building within their institutions, and commit to relationship building beyond specific project timelines.

Finally, findings suggest the need to increase the visibility and accessibility of water resources in the urban corridor. Water managers may want to consider daylighting streams and creating more community-water access points, but above all proactively engaging community members in dialogues on community values and needs related to water access.

Acknowledgments

The authors would like to acknowledge and thank the Minnehaha Creek Watershed District (MCWD) for funding this research.

Author Bio and Contact Information

Amit Pradhananga (corresponding author) is a research associate in the Center for Changing Landscapes and the Department of Forest Resources at the University of Minnesota. His research focuses on the human and policy dimensions of natural resource management, particularly water resources, invasive species, parks and trails management, and climate change adaptation. He is especially interested in investigating constraints and strategies to engage racial and ethnic community members in natural resource issues. He has a PhD in Natural Resource Science and Management from the University of Minnesota. He may be contacted at prad0047@umn.edu or by mail at 37 McNeal Hall, 1985 Buford Ave., St. Paul, MN 55108

Mae Davenport is the director of the Center for Changing Landscapes and professor in the Department of Forest Resources at the University of Minnesota. Her research interests are focused on the human dimensions of natural resource management, specifically land use planning, community-based ecosystem management, recreation planning; as well as human beliefs, attitudes, and behaviors associated with landscape change. She has a PhD in Natural Resource Science and Management from the University of Minnesota.

Emily Green is an editor and researcher in the Center for Changing Landscapes at the University of Minnesota. Her primary interests are in supporting natural resource protection, climate change understanding, sustainable land use, and community engagement in environmental protection and conservation. She has an M.S. degree in Conservation Biology from the University of Minnesota.

References

Arnstein, S. 1969. A ladder of citizen participation. *Journal of the American Institute of Planners* 35(4): 216-224.

Brinkman, S. and S. Kvale. 2015. *Interviews: Learning the Craft of Qualitative Research Interviewing* (3rd ed.). Sage Publications, Thousand Oaks, CA.

Charmaz, K. 2006. *Constructing Grounded Theory: A Practical Guide through Qualitative Analysis.* Sage Publications, Thousand Oaks, CA.

Clarke, L. and J. Agyeman. 2011. Is there more to environmental participation than meets the eye? Understanding agency, empowerment and disempowerment among black and minority ethnic communities. *Area* 43(1): 88-95.

Corbin, J. and A. Strauss. 2008. *Basics of Qualitative Research: Techniques and Procedures for Developing Grounded Theory.* Sage Publications, Thousand Oaks, CA.

Davenport, M., V. Perry, A. Pradhananga, and J. Shepard. 2016. *A Community Capacity Assessment for Stormwater Management in Three Twin Cities Metro Area Watersheds: A Social Science-Based Assessment*. Final technical report prepared for Ramsey Washington Metro Watershed District, Capitol Region Watershed District, and Mississippi Watershed Management Organization. University of Minnesota, St. Paul, MN. Available at: https://www.forestry.umn.edu/sites/forestry.umn.edu/files/metro_stormwater_full_technical_report_final.pdf. Accessed March 25, 2019.

Di Chiro, G. 2008. Living environmentalisms: Coalition politics, social reproduction, and environmental justice. *Environmental Politics* 17(2): 276-298.

Fraser, E.D.G., A.J. Dougill, W. Mabee, M.S. Reed, and P. McAlpine. 2006. Bottom up and top down: Analysis of participatory processes for sustainability indicator identification as a pathway to community empowerment and sustainable environmental management. *Journal of Environmental Management* 78(2): 114-127.

Gibson-Wood, H. and S. Wakefield. 2013. "Participation", white privilege and environmental justice: Understanding environmentalism among Hispanics in Toronto. *Antipode* 45(3): 641-662.

Koehler, B. and T. Koontz. 2008. Citizen participation in collaborative watershed partnerships. *Environmental Management* 41(2): 143-154.

Lane, M. 2005. Public participation in planning: An intellectual history. *Australian Geographer* 36(3): 283-299.

Larson, K. and D. Lach. 2008. Participants and non-participants of place-based groups: An assessment of attitudes and implications for public participation in water resource management. *Journal of Environmental Management* 88(4): 817-830.

Larson, K. and D. Lach. 2010. Equity in urban water governance through participatory, place-based approaches. *Natural Resources Journal* 50(2): 407-430.

Lawrence, R., S. Daniels, and G. Stankey. 1997. Procedural justice and public involvement in natural resource decision making. *Society and Natural Resources* 10(6): 577-589.

Leach, W. and P. Sebastier. 2005. To trust an adversary: Integrating rational and psychological models of collaborative policymaking. *American Political Science Review* 99(4): 491-503.

Lind, E.A. and T.R. Tyler. 1988. *The Social Psychology of Procedural Justice.* Plenum Press, New York, NY.

Lubell, M. 2005. Do watershed partnerships enhance beliefs conducive to collective action? In: *Swimming Upstream: Collaborative Approaches to Watershed Management*, P.A. Sabatier, W. Focht, M. Lubell, Z. Trachtenberg, A. Vedlitz, and M. Matlock (Eds.). The MIT Press, Cambridge, MA, pp. 201-232.

Metropolitan Council. 2012. 2030 Regional Development Framework – Revised Forecasts as of January 1, 2012. Available at: https://metrocouncil.org/Planning/Planning/2030-Regional-Development-Framework/RDF-Forecasts-January-2012.aspx. Accessed March 25, 2019.

Miles, M. and A.M. Huberman. 1994. *Qualitative Data Analysis: An Expanded Sourcebook.* Sage Publications, Thousand Oaks, CA.

Minneapolis Park and Recreation Board (MPRB). 2019. Minnehaha Regional Park. Available at: https://www.minneapolisparks.org/parks_destinations/parks__lakes/minnehaha_regional_park/. Accessed March 25, 2019.

Minnehaha Creek Watershed District (MCWD). 2017. MCWD 50th Anniversary: Celebrating 50 Years of Protecting Some of our State's Most Iconic Resources. Available at: https://www.minnehahacreek.org/sites/minnehahacreek.org/files/attachments/MCWD%2050th%20Publication%20Final_compressed_0.pdf. Accessed March 25, 2019.

Minnehaha Creek Watershed District (MCWD). 2018. Minnehaha Creek Watershed District Comprehensive Water Resources Management Plan 2007-2017. Available at: http://www.minnehahacreek.org/sites/minnehahacreek.org/files/attachments/Comprehensive%20Plan_amended%20thru%208-15-13.pdf. Accessed March 25, 2019.

Mississippi Watershed Management Organization (MWMO) and City of Minneapolis. 2007. Hmong Water Research Project Kev Cob Qhia Zej Tsoom Hmoob Txog Dej. Assessing Attitudes, Perceptions and Behavior about Water in Minnesota's Hmong Community. Prepared by Katherine Barton and Associates.

Mohai, P. and B. Bryant. 1998. Is there a 'race' effect on concern for environmental quality? *Public Opinion Quarterly* 62(4): 475-505.

Moraes, A. and P. Perkins. 2007. Women, equity, and participatory water management in Brazil. *International Feminist Journal of Politics* 9(4): 485-493.

National Research Council. 2008. *Public Participation in Environmental Assessment and Decision Making*, T. Dietz and P. Stern (Eds.). The National Academies Press, Washington, D.C.

Phadke, R., C. Manning, and S. Burlager. 2015. Making it personal: Diversity and deliberation in climate adaptation planning. *Climate Risk Management* 9: 62-76.

Pothier, M., N. Zewge-Abubaker, M. Cahuas, C. Borstad Klassen, and S. Wakefield. 2019. Is "including them" enough? How narratives of race and class shape participation in a resident-led revitalization initiative. *Geoforum* 98: 161-169.

Prokopy L. and K. Floress. 2011. Measuring the citizen effect: What does good citizen involvement look like? In: *Pathways for Getting to Better Water Quality: The Citizen Effect,* L.W. Morton and S.S. Brown (Eds.). Springer-Verlag, New York, NY, pp. 83-94.

Sabatier, P., W. Leach, M. Lubell, and N. Pelkey. 2005. Theoretical frameworks explaining partnership success. In: *Swimming Upstream: Collaborative Approaches to Watershed Management*, P.A. Sabatier, W. Focht, M. Lubell, Z. Trachtenberg, A. Vedlitz, and M. Matlock (Eds.). The MIT Press, Cambridge, MA, pp. 173-200.

Saldana, J. 2009. *The Coding Manual for Qualitative Researchers* (3rd ed.). Sage Publications, Thousand Oaks, CA.

Sarokin, D. and J. Schulkin. 1994. Environmental justice: Co-evolution of environmental concerns and social justice. *The Environmentalist* 14(2): 121-129.

Schlosberg, D. 2004. Reconceiving environmental justice: Global movements and political theories. *Environmental Politics* 13(3): 517-540.

Schlosberg, D. 2007. *Defining Environmental Justice: Theories, Movement and Nature*. Oxford University Press, New York.

Selfa, T. and T. Becerra. 2011. Upstream, downstream: Forging a rural-urban partnership for shared water governance in central Kansas. In: *Pathways for Getting to Better Water Quality: The Citizen Effect,* L.W. Morton and S.S. Brown (Eds.). Springer, New York, NY, pp. 121-132.

Trachtenberg, Z. and W. Focht. 2005. Legitimacy and watershed collaborations: The role of public participation. In: *Swimming Upstream: Collaborative Approaches to Watershed Management*, P.A. Sabatier, W. Focht, M. Lubell, Z. Trachtenberg, A. Vedlitz, and M. Matlock (Eds.). The MIT Press, Cambridge, MA, pp. 53-82.

U.S. Census Bureau. 2010. American FactFinder. Available at: http://factfinder.census.gov. Accessed March 25, 2019.

U.S. EPA. 2018. Impaired Waters and TMDLs in Region 5. Available at: https://www.epa.gov/tmdl/impaired-waters-and-tmdls-region-5. Accessed March 25, 2019.

Wakefield S. and B. Poland. 2005. Family, friend or foe? Critical reflections on the relevance and role of social capital in health promotion and community development. *Social Science and Medicine* 60(12): 2819-2832.

Appendix 1. Minnehaha Creek Assessment Interview Guide

First, I have some questions about you and your connection to this community.

1. How would you define community?
2. How would you describe your connection to this community?
3. What has been your role as [position] in this community?
4. What would you say are the best things about the work you do in this community?
5. What have been some of the most challenging things about the work you do in this community?

Next, I have some general questions about community assets and needs.

6. What would you say are the biggest assets of the community?
   a. What makes these assets important?
7. What do you believe are the most pressing needs in the community?
   a. What makes these needs important?
8. In the past 5 years, what would you say have been the most significant problems the community has faced?
9. How effective has the community been at responding to or managing these problems?
   a. What made it effective/ineffective? Can you provide examples?

Now, I have some specific questions about community planning and water resources in the [X] watershed, which intersects the community [Map: point to watershed boundaries on map].

10. How important are water resources such as local streams and lakes to quality of life for residents in this community?
11. Is the community actively engaged in land use planning in this watershed?
    a. What success has it experienced? Please explain.
    b. What challenges or setbacks has it experienced? Please explain.
12. Is the community actively engaged in water resource protection and restoration in this watershed?
13. What success has the community had related to water resource protection? Please explain.
    a. What has contributed to these successes? (e.g., leadership, funding, citizen groups, etc.)
14. What challenges or setbacks has the community had related to water resource protection? Please explain.
    a. What has contributed to these challenges?
15. As you may know, certain streams and lakes in the area have been identified as polluted or impaired with respect to water quality and aquatic habitat. How concerned are you about the quality of water resources in the community? Please explain.
    a. Are there any issues that you are most concerned about?
16. If the community was going to be more effective at addressing these types of water resource problems…
    a. What would it need to do?
    b. How would it do this?
    c. What resources would it need to accomplish this?
17. What do you see as the 3 biggest barriers to better engage this community in water resource protection and restoration?
18. What do you see as the 3 most promising opportunities to better engage this community in water resource protection and restoration?
19. Is there anything else you would like to share with me about the community or water resources in this area?

Finally, I would like to get some recommendations from you as we proceed with this project.

20. What other community representatives (e.g., from government, organizations or interest groups) could give us an important perspective on community assets and needs on water resources in this area? (Those with similar or very different perspectives than you.)
    a. What makes them a key representative (organizations they are involved in, how are they involved in watershed management in this area)?
    b. May we tell them you recommended them?
21. We would like to identify representatives willing to provide input, receive information and serve as community liaisons for the duration of this project. Would you be interested? ___Yes ___No

UCOWR UNIVERSITIES COUNCIL ON WATER RESOURCES

NIWR The National Institutes for Water Resources

2019 UCOWR/NIWR Annual Water Resources Conference

June 11-13, 2019

Snowbird, UT

Register now at https://ucowr.org/2019-conference/registration/.

Scale new heights as we come both west and up in the spectacular Wasatch Mountains in Snowbird, Utah just outside of Salt Lake City. The challenges facing the water community are constantly increasing in number and growing in complexity. In facing an uncertain future related to water availability in the West, water overabundance and quality in the East and Midwest, and water damage in the Southeast, the need for communicating our research and ideas is ever more important. UCOWR and NIWR invite you and your colleagues to join leading researchers, educators, water managers, and other professionals from across the country to address some of the most compelling and important challenges facing our profession. This year's conference is unique because, in addition to being both a scientific conference and an exploration of how universities help to meet societal goals, it will highlight the many unmet challenges in a newly uncertain cultural and regulatory climate.

Presentations will be made on these and other topics:

- Water-Energy Nexus
- Climate Change
- Extremes in Water Resources
- Water Economics
- Water Conservation Strategies
- Groundwater Management
- Hydrologic Connectivity
- Water Resources Policy
- Extension Education
- Colorado River Basin
- Agriculture and Water Use
- Rural Water Issues in Developing Countries
- Science Communication
- Engaging the Public on Water Issues
- Water Education
- Water Quality Monitoring
- Emerging Contaminants
- Agricultural BMPs
- Bacteria Sources, Fate, and Transport
- Columbia River Basin
- Water Infrastructure
- Water Sensors
- Data Management and Modeling
- Harmful Algal Blooms
- Wastewater Treatment
- Urban Water Conservation and Security
- Diversity in Water Research
- Drought and Water Scarcity
- Remote Sensing
- GIS

For more info, visit www.ucowr.org. General questions about the conference can be directed to Karl Williard (williard@siu.edu), Executive Director of UCOWR, or Staci Eakins (ucowr@siu.edu), Administrative Assistant.

Contents

2019 UCOWR / NIWR
Annual Water Resources Conference
Snowbird, UT
June 11-13, 2019

UCOWR
UNIVERSITIES COUNCIL ON WATER RESOURCES

Universities Council on Water Resources
Mail Code 4526
Southern Illinois University Carbondale
1231 Lincoln Drive
Carbondale, IL 62901
www.ucowr.org