Journal of Contemporary

Water Research & Education

Issue 158
August 2016







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

CO-EDITORS

Karl W. J. Williard
Southern Illinois University
Carbondale, Illinois 62901
williard@siu.edu

Jackie F. Crim
Southern Illinois University
Carbondale, Illinois 62901
crimjac@siu.edu

ASSOCIATE EDITORS

Kofi Akamani
Policy and Human Dimensions
Southern Illinois University
k.akamani@siu.edu

Natalie Carroll
Education
Purdue University
ncarroll@purdue.edu

James Heaney
Water Use and Efficiency
University of Florida
heaney@ufl.edu

M.S. Srinivasan
Hydrology
National Institute of Water and
Atmospheric Research, New Zealand
MS.Srinivasan@niwa.co.nz

Kevin Wagner
Water Quality and Watershed Management
Texas A&M University
klwagner@ag.tamu.edu

Jonathan Yoder
Natural Resource Economics
Washington State University
yoder@wsu.edu

TECHNICAL EDITORS

Elaine Groninger
Southern Illinois University
Carbondale, Illinois 62901
egroninger@siu.edu

Shelly Williard
Southern Illinois University
Carbondale, Illinois 62901
swilliard@siu.edu

EDITORIAL STAFF

Melissa Pind
Southern Illinois University
Carbondale, Illinois 62901
mpind@siu.edu

ISSN 1936-7031

Cover photos: Top: *Lower Cache River access in southern Illinois*, Credit: Jennifer Behnken; Middle: *Catfish pond on the campus of Samaru College of Agriculture*, *Zaria, Nigeria*, Credit: John Groninger; Bottom: *Generating an artificial rain event using a mobile unit of rainfall simulators, Clear Creek, IA*, Credit: Praveen Kumar
Back cover & inside back cover photos: Courtesy of Colorado State University

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.

UCOWR is not responsible for the statements and opinions expressed by authors of articles in the *Journal of Contemporary Water Research & Education.*

Printed by the authority of the State of Illinois. August 2016. 458 copies. Order Number 170282.

Journal of Contemporary Water Research & Education

Issue No. 158 **August 2016**

Letter from the Editors
Karl W.J. Williard and Jackie F. Crim 1

Adaptive Water Governance: Integrating the Human Dimensions into Water Resource Governance
Kofi Akamani 2

Institutional Constraints to Collaborative Ecosystem Management within a Wetlands Conservation Partnership
Jennifer A. Behnken, John W. Groninger, and Kofi Akamani 19

The Effects of Policy Implementation Uncertainty on Groundwater Extraction
Dacheng Bian, Ryan B. Williams, Aaron Benson, and Eduardo Segarra 34

Climate Change, Agriculture and Water Resources in the Southwestern United States
Emile Elias, Al Rango, Ryann Smith, Connie Maxwell, Caiti Steele, and Kris Havstad 46

Irrigation Technology Choice as Adaptation to Climate Change in the Western United States
George Frisvold and Ting Bai 62

Building Watershed Management Capacity in Nigeria: Expanding the Role of Agriculture Colleges
John W. Groninger 78

Heard it through the Grapevine - Social Network Analysis of a Hydrology Class
E. Annette Hernandez and Venkatesh Uddameri 85

Economic Value of Nature and Ecosystems in the Delaware River Basin
Gerald J. Kauffman 98

Water and Society: Interdisciplinary Education in Natural Resources
Laszlo J. Kulcsar, Joseph A. Aistrup, Thomas Bulatewicz, Jeffrey M. Peterson, Stephen M. Welch, and David R. Steward 120

Deliberating on Missouri River Water Diversions in Congressional Committee Hearings
Sarah Michaels and Mike Gruszczynski 132

Dynamic Assessment of Current Management in an Intensively Managed Agroecosystem
Christopher G. Wilson, Kenneth M. Wacha, A.N. (Thanos) Papanicolaou, Heather A. Sander, Violet B. Freudenberg, Benjamin K.B. Abban, and Chang Zhao 148

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Page 1, August 2016

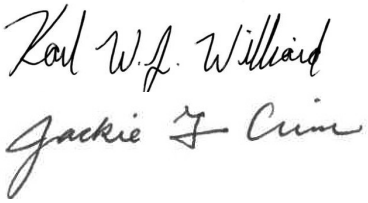
Letter from the Editors

It is our pleasure to introduce our first open solicitation issue of the *Journal of Contemporary Water Research and Education* (*JCWRE*). Since our first issue in 1964, *JCWRE* (formerly *Water Resources Update*) has focused on publishing special themed issues, generally three per year. Looking through our archives, *JCWRE* has covered current water resources issues across a broad range of fields including, but not limited to policy, planning, watershed management, education, hydrology, economics, water supply, climate change, water infrastructure, groundwater, trans-boundary issues, water rights, and the energy-water nexus.

Each themed issue has an Issue Editor who develops the theme, invites potential authors, and solicits external reviews for the submitted manuscripts. To help us in the review process of our open issues, we formed a group of Associate Editors in our primary publication areas. We are pleased to announce our inaugural group of Associate Editors and their specialty areas: Jonathan Yoder, Washington State University, Natural Resource Economics; Kofi Akamani, Southern Illinois University Carbondale, Policy and Human Dimensions; Natalie Carroll, Purdue University, Education; M.S. Srinivasan, National Institute of Water and Atmospheric Research, New Zealand, Hydrology; James Heaney, University of Florida, Water Use and Efficiency; and Kevin Wagner, Texas A&M University, Water Quality and Watershed Management. Our current plan is to devote one of our issues per year (August) to an open solicitation issue.

JCWRE will continue to build on our historic strength as an outlet for applied work in transdisciplinary water resources research, education, and outreach. We seek to highlight work that focuses on developing solutions for the complex water challenges of our generation. Please contact us if you have any ideas for a themed issue or a manuscript for our next open issue in August 2017. We look forward to hearing from you.

Sincerely,



Karl W.J. Williard and Jackie F. Crim
Co-Editors, *Journal of Contemporary Water Research and Education*

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 2-18, August 2016

Adaptive Water Governance: Integrating the Human Dimensions into Water Resource Governance

Kofi Akamani

Department of Forestry, Southern Illinois University, Carbondale, IL

Abstract: The command-and-control paradigm of water resource management that dominated 20th century water policy has resulted in vulnerabilities due to its inadequate consideration of the human dimensions as well as its failure to prioritize the need for learning and adapting to change and uncertainties. In response to these shortfalls, recent years have seen the emergence of alternative water policy frameworks, notably integrated water resource management (IWRM) and adaptive management. However, while IWRM broadens the planning goals and scope of water resource governance through integration and coordination across scales and sectors, it does not adequately emphasize learning to deal with uncertainty. On the other hand, adaptive management emphasizes the need to prioritize learning to deal with uncertainty in resource management through monitoring and experimentation, but places less emphasis on the human dimensions required for its successful implementation. This paper discusses adaptive water governance – collaborative, flexible, and learning-based institutions that connect state and non-state actors across multiple levels for ecosystem-based management of land and water resources – as a promising institutional mechanism for integrating the human dimensions into water resource governance while building the capacity of water resource systems to learn and adapt to change. The paper highlights four attributes of adaptive water governance: (1) reintegrating humans into nature; (2) integrating diverse sources and types of knowledge; (3) promoting adaptive and integrative management goals; and (4) using polycentric institutions and analytic deliberation processes. Through these mechanisms, adaptive governance can contribute to sustainability, good governance, conflict management, and social-ecological resilience in water resource systems.

Keywords: *adaptive management, climate change, resilience, social-ecological systems, vulnerability, water policy*

In recent decades, the uncertainties and complexity associated with water resource systems have been gaining widespread recognition (Pahl-Wostl et al. 2007; Huitema et al. 2009; Akamani and Wilson 2011). The anticipated impacts of climate change, coupled with the interactions among the multiple social, ecological, and technological components of water resource systems add to this complexity (Pahl-Wost 2007b; Pahl-Wostl et al. 2009). In this context, the concepts of resilience and vulnerability are increasingly receiving attention in efforts to enhance the sustainable governance of water resources in the face of uncertainties from climate change and other drivers of change. Resilience refers to the capacity of a system to absorb disturbances while maintaining its structures, processes, and feedbacks (Berkes 2007). Vulnerability, on the other hand, is the degree of susceptibility of a system to harm from exposure to hazards (Turner et al. 2003). The conventional command-and-control water management paradigm that dominated much of 20th century water policy was premised on a belief in the ability to control and predict water resource systems emphasis on the pursuit of economic efficiency, reliance on positivist science, dominance of rational-comprehensive planning processes, and top-down institutional mechanisms (Pahl-Wostl et al. 2007; Raadgever et al. 2008; Molle 2009). Two shortfalls associated with this paradigm are the neglect of the human dimensions in water management (Floress et al. 2015), and the lack of emphasis on the need for learning and adaptation (Pahl-Wostl 2007b). These shortfalls have led to the erosion of resilience and increased vulnerability in water resource systems (Pahl-

Wostl 2007a; Sendzimir et al. 2008).

Through recent transitions in the governance of water resources toward integrated water resource management (IWRM) and adaptive management, the shortfalls in the conventional command-and-control paradigm of water resource management are being addressed in diverse ways. The concept of IWRM broadens the planning goals and scope of water resource governance through integration and coordination across scales and sectors, but does not adequately emphasize learning to deal with uncertainty (Biswas 2004; Medema et al. 2008; Gain et al. 2013). The concept of adaptive management also emphasizes the need to prioritize learning to deal with uncertainty in resource management through monitoring and experimentation, but emphasizes less on the human dimensions (e.g., stakeholder values, interests, and decision-making processes) required for its successful implementation (McLain and Lee 1996; Walters 2007). Thus, ongoing efforts aimed at integrating the human dimensions into water management processes and goals have not adequately accounted for the need to adapt to uncertainties, whereas those efforts focused on reducing uncertainties have not given adequate consideration to the human dimensions. Through the reconciliation of adaptive management and integrated resource management, the emerging literature on adaptive governance of social-ecological systems offers promise for integrating the human dimensions into water resource management as well as preparing linked social and ecological systems to learn and adapt to gradual and episodic changes. The pursuit of adaptive governance could contribute to enhancing good governance, conflict management, and building social-ecological resilience for sustainable water resource management (Dietz et al. 2003; Folke et al. 2005; Akamani and Wilson 2011). Yet, these potentials remain largely unexplored. Existing knowledge gaps in the adaptive governance literature include poor understanding of the conditions and processes for the emergence of adaptive governance regimes (Österblom and Folke 2013) and inadequate understanding of factors influencing success and failures in such regimes (Karpouzoglou et al. 2016). Methodological tools for assessing the outcomes of adaptive governance across multiple scales also remain largely undeveloped, although progress is being made in this direction (e.g., Schultz et al. 2015; Morris and de Loe 2016).

In this paper, I examine the prospects of adaptive governance as a conceptual foundation for adaptive and integrated water resource management, thereby providing a framework for integrating the human dimensions into the governance of water resources while simultaneously dealing with uncertainties. According to Jacobson and McDuff (1998, 265), the human dimensions encompasses "the beliefs, attitudes, values, behaviors, and socioeconomic, demographic, and organizational characteristics of the stakeholders involved in natural resource conservation issues." As such, the field of human dimensions borrows from a wide range of disciplines in the social sciences and humanities, including philosophy, economics, political science, anthropology, sociology, and social psychology to understand, explain, predict, and guide policies on human-environment interactions. The integration of the human dimensions in natural resource management is increasingly seen as an essential part of efforts aimed at promoting sustainable human-environment interactions (Endter-Wada et al. 1998; Mascia et al. 2003; Manfredo et al. 2014). For the purposes of this paper, I refer to the human dimensions as the social considerations in the assumptions, goals, knowledge, and institutional structures and processes that inform resource management. The next section of the paper presents an overview of transitions in the water governance literature by discussing the concepts of IWRM and adaptive management. Next, adaptive governance and its potential contributions to the human dimensions of water resources governance under dynamic conditions will be discussed. Concluding remarks will then be offered in the final section of the paper.

Evolving Approaches to Water Resource Governance

Integrated Water Resource Management (IWRM)

The concept of IWRM is one of the most widely discussed water policy frameworks in recent decades. The need for IWRM is born out of realization of the increasing interconnectedness between the water sector and other development

sectors such as agriculture, energy, industry, transportation, health, environment, and rural development across multiple scales from the local to the international (Biswas 2004). The complexity of these interactions is further compounded by the onslaught of economic and cultural globalization, climate change, and other global processes. Within this context, managing the different aspects of water resources in isolation from one another threatens long term sustainability (Engle et al. 2011). Realization of water policy goals will require the explicit and simultaneous consideration of potential trade-offs between the water sector and related sectors at various spatial and temporal scales (Biswas 2004; Pahl-Wostl 2007a; Pahl-Wostl et al. 2007). The concept of IWRM (Biswas 2004; Molle 2009) emerged in the 1990s to address these management challenges.

In an oft-quoted definition, the Global Water Partnership (GWP 2000, 22) defines IWRM as "a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems." More recently, Gain et al. (2013, 14) define IWRM as "an integrated form of sustainably managing water and associated resources at the river basin scale involving specific governance forms that guarantee multi-stakeholder participation, equitable access and efficient use of water resources." IWRM is a holistic resource management approach that primarily aims at promoting the coordinated and integrated management of land and water resources to enhance water resource sustainability (Medema et al. 2008). Through this holistic approach, IWRM has the potential to overcome the shortfalls associated with the conventional approach to managing water resources in isolation from other sectors (Blomquist and Schlager 2005; Pahl-Wostl et al. 2007). IWRM represents an important step toward the integration of the human dimensions into water resource governance through its emphasis on the integration of multiple disciplinary and stakeholder perspectives across multiple scales, as well as its emphasis on fairness, participation, and decentralized governance systems (Medema et al. 2008; Engle et al. 2011; Plummer et al. 2011; Gain et al. 2013).

In spite of its promise, the concept of IWRM has seen limited success in practical implementation (Blomquist and Schlager 2005). Medema et al. (2008) have identified three conditions for the successful implementation of IWRM programs: availability of enabling legislation and policy environment; appropriate institutional frameworks; and availability of instruments for assessments, as well as gathering of data and information. However, these ideal conditions rarely exist. The implementation of IWRM programs is often constrained by a range of challenges, including the lack of clarity and consensus in the meaning of the concept, capacity and institutional constraints in the implementation of integrated approaches across multiple sectors, and the lack of appropriate indicators for assessing success in IWRM programs (Biswas 2004; Medema et al. 2008). Using the case of a small watershed in Southern California, Blomquist and Schlager (2005) highlight the political challenges entailed in defining the boundaries of watersheds, choosing appropriate decision-making structures, and enhancing accountability in IWRM. Importantly, IWRM has not adequately addressed the uncertainties entailed in the management of water resource systems in the face of climate change impacts and other drivers of change (Pahl-Wostl et al. 2007; Engle et al. 2011; Gain et al. 2013).

Adaptive Management

Since the 1970s, adaptive management has been receiving attention as an approach for dealing with the complexity and uncertainties entailed in the resource management process (Holling 1978; Walters 1986). The concept has been particularly embraced in the field of water resource management (Pahl-Wostl et al. 2007; Huitema et al. 2009; Engle et al. 2011). Adaptive management is a resource management approach that is based on the assumption that resource management occurs in the face of incomplete knowledge and unpredictable social and ecological interactions (Ringold et al. 1996; Lee 1999; Walters 2007). As a consequence, adaptive management aims at enhancing the resilience and flexibility of management systems to cope with uncertainty (Allen et al. 2011; Allen and Gunderson 2011). Pahl-Wostl (2007a, 567) defines adaptive management as "a systematic process

for continually improving management policies and practices by learning from the outcomes of implemented management strategies." While definitions vary, adaptive management generally refers to a structured process of learning by doing in the face of uncertainty (Walters 1997). Structured decision making provides a systematic and transparent process for identifying and evaluating policy alternatives in the adaptive management process (Allen et al. 2011). However, not all forms of adaptive management follow such processes.

Based on Walters and Holling (1990), Allan and Curtis (2005) identify three types of adaptive management: evolutionary; passive; and active adaptive management. Evolutionary adaptive management occurs in the form of trial and error learning, i.e. random experience. For instance, the adaptive component of locally evolved resource management systems that occur via experiential learning is gaining recognition (Berkes et al. 2000; Olsson and Folke 2001). Passive adaptive management entails the implementation of an informed policy option based on the best available data, followed by monitoring of the implementation process (Allan and Curtis 2005; Gregory et al. 2006). Passive adaptive is relatively simple and less expensive due to its non-experimental nature (Medema et al. 2008). As such, it is largely favored by government representatives in the policy process (Gregory et al. 2006). Active adaptive management facilitates learning by treating the resource management process as an experiment to test competing hypotheses, as well as emphasizing monitoring and integration of new knowledge into subsequent management actions (Gregory et al. 2006; Allen and Gunderson 2011; Benson and Garmestani 2011). Active adaptive management bridges the gap between science and policy through its experimental resource management approach (Medema et al. 2008; Allen et al. 2011; Benson and Garmestani 2011; Smith 2011). While active adaptive management is generally considered the most superior form of adaptive management from the standpoint of enhancing learning (Benson and Garmestani 2011), there is little evidence of its successful implementation (Gregory et al. 2006; Allen and Gunderson 2011).

The challenges in the conceptualization and implementation of adaptive management processes have received considerable attention in the literature. Walters (1997) discusses four impediments to adaptive management of riparian and coastal ecosystems: technical difficulties in the development of accurate models representing complex social-ecological relationships; perceived risks and costs associated with large scale adaptive management experiments; opposition to experimentation by vested interests; and conflicts in values and interests among stakeholders in the adaptive management process. In his assessment of the Everglades of Florida, Gunderson (1999) also cautioned that adaptive management cannot succeed where there is a loss of resilience in the ecological system and/or a lack of flexibility in the social system. The literature on land and water resources increasingly highlights the social and institutional nature of barriers to adaptive management success, including the absence of enabling regulatory and policy environments, lack of leadership, declining resources and capacity for monitoring and experimentation, reluctance of policy makers to embrace the uncertainty assumptions of adaptive management, lack of adequate engagement of stakeholders, neglect of nonscientific forms of knowledge, and inadequate consideration of the broad range of social values in adaptive management processes (McLain and Lee 1996; Stankey et al. 2003; Walters 2007; Allen and Gunderson 2011). In view of these insights, it has been argued that successful adaptive management will require institutional mechanisms that adequately promote stakeholder involvement, as well as integration of diverse values and knowledge systems (McLain and Lee 1996; Folke et al. 2002). This call for the integration of the human dimensions into adaptive management of water resources requires an adaptive governance approach (Folke 2006; Gunderson and Light 2006; Walker 2012; Aytur et al. 2015; Morris and de Loe 2016).

Adaptive Water Governance

Adaptive governance of social-ecological systems (Dietz et al. 2003; Folke et al. 2005) is a promising alternative to the command-and-control paradigm of water resource management, and it holds promise as the foundation for adaptive and integrated management of water and other

natural resources. Adaptive governance addresses the social and institutional context that enables adaptive ecosystem-based management (Folke et al. 2005; Folke 2006). Österblom and Folke (2013) contend that adaptive governance is suitable for dealing with change and complexity in a context characterized by diverse actors with differences in values, interests, and power that give rise to conflicts. As such, adaptive governance has the potential to address the dual challenge of integrating social considerations into the resource management process, as well as dealing with uncertainties in complex social-ecological systems. Adaptive governance is a collaborative, flexible, learning-based approach to ecosystem management that relies on a diverse network of institutions and organizations across multiple levels from the local to the international (Hughes et al. 2005; Olsson et al. 2006). The adaptive governance framework also provides a medium for linking science and decision-making in an ongoing process of dealing with change (Gunderson and Light 2006). Adaptive governance is closely related to adaptive management (Holling 1978; Allen et al. 2011), as well as other mechanisms for governing common pool resources, such as co-management (Berkes 2009) and adaptive co-management (Olsson et al. 2004; Plummer 2009). However, adaptive governance is particularly suited for situations involving abrupt and transformational change in social-ecological systems (Olsson et al. 2006; Walker 2012). Transformational changes entail the crossing of critical thresholds, resulting in a shift from one regime to another (Folke et al. 2010; Walker 2012). In such situations, adaptive management and adaptive co-management have limited applicability, as they mainly address the capacity to deal with change while staying within an existing social-ecological regime (Olsson et al. 2014).

Transformation toward adaptive governance for ecosystem-based resource management is influenced by many factors, including the availability of enabling legislation, presence of economic and non-economic incentives, bridging organizations, windows of opportunity, and leadership (Folke 2007; Olsson et al. 2008; Folke et al. 2010; Gelcich et al. 2010; Österblom and Folke 2013). While the burgeoning literature on adaptive governance is quite disparate, key attributes of adaptive governance include the following: embracing complexity and resilience through recognition of the interdependence between social and ecological systems; promoting adaptive, integrated resource management goals; utilizing diverse sources of knowledge; and relying on polycentric institutional structures and analytic deliberation processes for managing conflicts (Dietz et al. 2003; Hughes et al. 2005; Gunderson and Light 2006; Nelson et al. 2008). These attributes do not exist in isolation but seem to operate as interconnected aspects of adaptive water governance as a resource regime. The ensuing section discusses how these four attributes of adaptive governance I identify in the existing literature can contribute to integrating the human dimensions into water resource management assumptions, goals, knowledge systems, and institutions while preparing the coupled social-ecological system to deal with uncertainties.

Reintegrating Humans into Nature

At the philosophical level, conventional resource management has been critiqued for its flawed assumption of nature-culture dichotomy whereby social and ecological systems are treated as separate from each other (Folke et al. 2002; Berkes 2004). Alberti et al. (2003, 1170) have noted that "Despite dominating Earth's ecosystems, humans remain conspicuously excluded as subjects of much ecological thinking and experimentation. Traditional ecological research investigates ecosystems in terms of biophysical, ecological, and evolutionary processes unaffected by human influences." Similarly, social science fields, such as common pool resources research, have typically focused on the social system without adequate consideration of the ecological system on which the social system depends (Hughes et al. 2005). The roots of the separation of social systems from the natural world have been traced to multiple sources, notably the balance of nature paradigm (Cole and Yung 2010), the Judeo-Christian religious tradition (White 1967), and enlightenment thought (Cortner and Moote 1999). Closely related to the assumption of nature-culture dichotomy, another flawed assumption of conventional resource management is its conceptualization of ecosystem

dynamics as relatively stable and predictable, fluctuating around a single equilibrium (Folke 2006). Working together, these twin assumptions have impeded the accumulation of knowledge on the dynamic and reciprocal interactions between social and biophysical systems, thus leading to misinformed policies (Hughes et al. 2005; Folke 2007; Liu et al. 2007).

Over the last two decades, these older assumptions have begun to be replaced with alternative conceptualizations of social and ecological systems as coupled social-ecological systems, intricately interconnected and co-evolving across multiple spatial and temporal scales (Anderies et al. 2004; Folke 2007). The simultaneous study of the reciprocal interactions between social and ecological systems has revealed that coupled social-ecological systems exhibit the attributes of complex adaptive systems, such as nonlinearity, thresholds, emergence, surprise, scale, and path-dependency (Berkes 2004; Hughes et al. 2005; Folke 2006, 2007; Liu et al. 2007). Holling's (2001) panarchy theory models the dynamic cross-scale interactions among the various levels of social-ecological systems. Contrary to the assumption of ecosystems in a stable equilibrium, panarchy is a representation of the dynamics of social-ecological systems as a nested set of adaptive cycles in constant flux (Folke et al. 2010). Adaptive cycles represent the four phases of growth, conservation, collapse, and reorganization that each level of a social-ecological system constantly passes through (Carpenter et al. 2001). The sustainable management of such complex social-ecological systems calls for building resilience rather than the pursuit of efficiency and output maximization (Holling and Meffe 1996). These theoretical concepts that serve as the foundation for adaptive governance provide an opportunity for the reintegration of humans into the biosphere for ecosystem-based management of land and water resources (Chapin et al. 2009; Folke et al. 2011; Valman et al. 2015). For instance, in their recent analysis of adaptive governance regimes in the Kristianstads Vattenrike in Southern Sweden, the Great Barrier Reef in Australia, and fisheries in the Southern Ocean, Schultz et al. (2015, 7371) highlighted that "a mental shift, a reframing of the human-nature relation" was an essential component of the transition toward adaptive governance mechanisms that prioritized managing multiple ecosystem services and dealing with change across scales.

Integration of Diverse Sources and Types of Knowledge

The traditional separation between the social and ecological sciences has led to the marginalization of social issues in conventional resource management (Jacobson and McDuff 1998; Redman et al. 2004). The resource management process is often dominated by resource managers, scientists, and other professionals trained in the natural sciences (Endter-Wada et al. 1998). In these management processes, preference is given to positivist and reductionist scientific methods for generating quantitative data for the purpose of generalization and prediction of natural phenomena (Berkes 2003; Nelson et al. 2008). As a consequence, social science disciplines and non-scientific forms of knowledge have been virtually overlooked (Cortner and Moote 1999; Clark 2002). For instance, the field of water resources management has traditionally been dominated by hydraulic engineers (Raadgever et al. 2008), thus offering little space for social scientists. However, insights from the complex systems perspective highlight the fact that knowledge about such systems varies according to the scale of observation and is dispersed among different stakeholders. As such, no single analytical perspective or stakeholder group can yield adequate knowledge for making sense of the complexity of the whole system (Berkes 2004; Armitage et al. 2012). Each perspective offers a legitimate but incomplete account of the whole. Reductionist scientific methods are particularly ill-suited for understanding the non-linear relationships and emergent properties of complex systems (Costanza 1996; Nelson et al. 2008; Armitage et al. 2009).

As a response to the shortfalls of the dominance of reductionist and positivist science in conventional resource management, the adaptive governance approach calls for the integration of different knowledge systems (Nelson et al. 2008; West and Schultz 2015). In view of the inadequacy of single disciplinary approaches in the study of complex social-ecological systems, cross-disciplinary

approaches are gaining recognition in the study of land and water resource systems (Lélé and Norgaad 2005; Eigenbrode et al. 2007). For instance, the emerging field of socio-hydrology moves beyond the dominance of hydrology to integrate the social sciences in water resources research (Sivapalan et al. 2011). According to Sivapalan et al. (2014, 226), "Socio-hydrology is aimed at uncovering the dynamic cross-scale interactions and feedbacks between the natural and human processes that may give rise to water sustainability challenges that we face in the Anthropocene." Unlike multi-disciplinary and interdisciplinary approaches, the transdisciplinary field of socio-hydrology offers a deeper understanding of the dynamics of socio-hydrologic systems through its explicit emphasis on the reciprocal feedbacks between social and hydrologic systems (Sivapalan et al. 2011, 2014). Such cross-disciplinary approaches offer opportunities for utilizing knowledge from the social and biophysical sciences in the sustainable management of complex water resource systems.

Knowledge integration in adaptive governance occurs not only among academic disciplines across the social and biophysical sciences, but also includes the recognition of local and traditional ecological knowledge (Hughes et al. 2005; Cosens 2013; Morris and de Loe 2016). Berkes et al. (2000, 1252) define traditional ecological knowledge as "a cumulative body of knowledge, practice, and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment." The authors contend that traditional ecological knowledge is typically associated with societies in which there is a historical continuity in resource use. Traditional ecological knowledge entails not only the observational knowledge derived from experience, but also includes the worldviews, beliefs, institutions, and practices within which such knowledge is nested (Berkes et al. 2000; Houde 2007). Traditional ecological knowledge differs from local ecological knowledge, which comprises a mix of scientific knowledge and practical knowledge (Olsson and Folke 2001).

Adaptive governance mechanisms also require the promotion of social learning (Folke et al. 2005; Bark et al. 2012). Reed et al. (2010) define social learning as "change in understanding that goes beyond the individual to become situated within wider social units or communities of practice through social interactions between actors within social networks." The social learning approach to knowledge differs from the reductionist approach that focuses on learning at the individual level. As Armitage et al. (2011) have noted, a focus on the individual ignores the social and institutional context that shapes learning processes. Social learning is collective learning that occurs through social interactions. Social interactions contribute to social learning through the direct transmission of information or through argumentation processes that lead to changes in ideas, worldviews, and rule systems (Reed et al. 2010). Social learning in adaptive governance processes is closely related to social memory – the reservoir of knowledge, practices, values, and worldviews that individuals and organizations have accumulated from past adaptations which is essential for dealing with future surprises (Adger et al. 2005). Such knowledge co-production requires decision-making processes that offer opportunities for meaningful participation and for bridging the epistemological differences among scientists, resource managers, resource users, and other stakeholders in the construction and interpretation of knowledge (Armitage et al. 2012). The vertical and horizontal linkages in the institutional structures and decision-making processes in the adaptive governance framework (discussed in subsequent sections) offer an enabling environment for knowledge co-production processes as well as the integration of science into policy (Bark et al. 2012; Aytur et al. 2015).

Promoting knowledge co-production and social learning in water resource management through adaptive governance may be enhanced using participatory action research methods, such as photovoice and photo-elicitation (Wang 1999; Bugos et al. 2014), participatory GIS (Brown et al. 2014), and participatory social network analysis (Fuller et al. 2012). These participatory approaches contribute to trust building, knowledge integration and sharing, capacity building, empowerment, and the flexibility to deal with change in water resource systems (Aytur et al. 2015; Madsen and O'Mullan 2016). For instance, Aytur et al. (2015) discuss the application of adaptive governance principles in

the development of an integrated climate change adaptation strategy for the town of Exeter, New Hampshire. Through the use of participatory action research methods, the project sought to promote regular interactions among diverse stakeholders at the local and state levels, and also to connect various epistemic communities including scientists and community groups. Results from an evaluation of the project showed that participants had become better informed about climate change issues (e.g., flooding and water quality degradation) and also had an enhanced ability to analyze climate change issues from multiple perspectives.

Adaptive and Integrated Management Goals

The emphasis of the command-and-control resource management paradigm on the maximization of economic benefits from selected components of biophysical systems that are assumed to be stable has been critiqued for eroding the resilience of social and ecological systems (Hughes et al. 2005; Holling 2012; Cosens 2013). Adaptive governance shifts the focus of resource management away from the pursuit of efficiency and optimization (Green et al. 2013), and instead offers an ecosystem-based framework for adaptive and integrated management of land and water resources (Engle et al. 2011; Bark et al. 2012; Valman et al. 2015; Akamani et al. 2016).

Adaptive governance embraces the holistic focus of ecosystem-based resource management on sustainability and resilience (Folke et al. 2005, 2011). Ecosystem-based management seeks to satisfy the range of social, economic, and ecological values desired by stakeholders while sustaining the capacity of the social-ecological system to produce those goods and services that sustain human well-being in the face of uncertainty (Hughes et al. 2005; Gunderson and Light 2006; Chapin et al. 2009). The pursuit of such holistic management goals provides an opportunity for integrating the diverse values and interests held by stakeholders, including the interests of traditionally marginalized groups in water resource governance, such as indigenous communities (Bark et al. 2012; von der Porten and de Loe 2014). While conventional integrated resource management has failed to address value conflicts through expert-driven rational-comprehensive planning approaches (Blomquist and Schlager 2005), the adaptive governance approach promotes the pursuit of integrated management goals by providing opportunities for negotiated solutions to conflicting values and knowledge uncertainties (Dietz et al. 2003; Nelson et al. 2008) while enhancing the coordination of responsibilities among diverse actors across scales and jurisdictions (Bark et al. 2012).

Besides the pursuit of an integrative agenda, another central component of adaptive governance is adaptive management (Gunderson and Light 2006; Walker 2012). Adaptive governance builds on the focus of adaptive management on dealing with uncertainties and surprises in social-ecological systems through learning (Karpouzoglou et al. 2016). However, adaptive governance broadens the scope of adaptive management by accounting for the wider social context within which ecosystem-based resource management occurs (Folke et al. 2005). As such, adaptive governance addresses the inadequate consideration of social and institutional issues that has impeded success in the implementation of adaptive management processes (Gunderson and Light 2006). In all, the adaptive governance approach provides enabling institutional structures and processes that facilitate integration within and across scales, as well as preparing coupled social-ecological systems to deal with uncertainties through adaptive management (Bark et al. 2012; Cosens 2013). Green et al. (2013) present an interesting case study on integrated and adaptive management in the Okavango River Basin of southern Africa. Through the coordinated efforts of actors across multiple levels from the local to the international, an integrated approach to addressing socio-economic and ecological concerns is being pursued through the Integrated River Basin Management Project. The authors contend that although the adaptive capacity of actors in the river basin has not recently been tested, there are indications of preparations for change and uncertainty through monitoring and information sharing, as well as opportunities for meaningful participation among actors.

Institutional Structures and Processes

The top-down institutional structures of the conventional water resource management paradigm have received widespread criticisms, including the

lack of fit between policy interventions and the scales at which problems occur (Lebel et al. 2008), lack of flexibility in adapting to change (Healey 1998), difficulty of integrating local knowledge (Nelson et al. 2008), and limited capacity for dealing with social-ecological complexity (Karkkainen 2005). Adaptive governance is a response to the shortfalls of both centralized and community-based institutional mechanisms. In their seminal paper on adaptive governance, Dietz et al. (2003) identify the institutional requirements of adaptive governance: availability of information, mechanisms for conflict management, inducement of rule compliance, provision of infrastructure, and preparedness for change. The authors propose three strategies for meeting these requirements: analytic deliberation, nesting, and institutional variety. These three strategies constitute the institutional structures and processes entailed in adaptive governance mechanisms that enable adaptive and integrated resource management.

Analytic Deliberation. Using public deliberation to inform scientific analysis is an effective way to integrate facts and values for good decision-making (Dietz 2013). The concept of analytic deliberation refers to a "Well-structured dialogue involving scientists, resource users, and interested publics, and informed by analysis of key information about environmental and human-environment systems" (Dietz et al. 2003, 1910). The wisdom in analytic deliberation is that it combines scientific analysis with public deliberation to inform better policies (Dietz and Stern 1998; Akamani and Wilson 2011). Whereas analysis entails the systematic application of rigorous and replicable procedures for data collection and interpretation, deliberation encompasses the formal and informal processes for communication and collective decision-making among scientists and other stakeholders (Tuler and Webler 1999; Webler and Tuler 1999). Analytic deliberation offers a more effective mechanism for dealing with the wicked problems that arise from the value conflicts and knowledge uncertainties in complex social-ecological systems than expert-driven rational-comprehensive planning processes (Ballint et al. 2011; Akamani et al. 2016). Analytic deliberation also offers opportunities for the integration of different types and sources of knowledge held by stakeholders (Corfee-Morlot et al. 2011), thereby enhancing prospects for the recognition of local and traditional ecological knowledge in adaptive governance processes. Such deliberation processes also meet the conditions for the emergence of social learning (Reed et al. 2010; Akamani and Wilson 2011), as well as the enhancement of trust and social capital among stakeholders (Dietz et al. 2003). In all, analytic deliberation addresses the need for inclusive and integrative institutional mechanisms for the transparent and evidence-based negotiation of trade-offs among stakeholders in the water governance process (Dietz and Stern 1998; Pahl-Wostl et al. 2013). Case studies from the broader literature on participation and social learning highlight that outcomes of such processes may be shaped by the patterns of interactions among participants (Crona and Parker 2012), the organizational structures guiding the process (Muro and Jeffrey 2012), and the duration of the process (Measham 2013). In the specific case of water resource management, Webler and Tuler (1999) offer a number of helpful guidelines for enhancing effective analytic deliberation processes. These include: ensuring a balanced representation of all stakeholders; motivating stakeholders to participate; creating opportunities for participation; addressing unequal access to the process, resources, information, and skills; use of facilitators; providing a safe environment for dialogue; and ensuring legitimacy of the entire process.

Nested Institutions. Dietz et al. (2003) also highlighted the need for nested institutions in the adaptive governance of social-ecological systems. As has been noted previously, a representation of social-ecological systems from the perspective of panarchy theory illustrates that social-ecological systems are composed of inter-dependent adaptive cycles that are nested across multiple levels and are constantly engaged in cross-scale and cross-level interactions (Holling 2001). Within such a multi-level system, top-down institutional arrangements that ignore these cross-scale issues often result in scale mismatches or the lack of fit between policy interventions and social-ecological problems (Cash et al. 2006; Folke et al. 2007; Olsson and Galaz 2009). The nested institutional structure of adaptive governance is an appropriate response to the scale issue in complex social-

ecological systems (Akamani and Wilson 2011). Nested institutions closely relate to the widely discussed concept of polycentric institutions (Marshall 2009). Ostrom (2010, 552) defines polycentric institutions as those "characterized by multiple governing authorities at differing scales rather than a monocentric unit." Each level within the polycentric system retains some autonomy to make and enforce rules (Marshall and Smith 2010), and responsibilities among levels of the institutional structure are allocated according to the principle of subsidiarity whereby responsibilities are devolved to the lowest appropriate level (Marshall 2009). In addition to addressing problems associated with scale mismatch, the polycentric institutional structure of adaptive governance offers several advantages, including the promotion of learning and experimentation at lower levels (Ostrom 1999), creating space for engagement of local communities in multi-level processes (Akamani and Wilson 2011; Bark et al. 2012), and providing redundancies that enhance system performance in the face of uncertainty (Dietz et al. 2003; Olsson and Galaz 2009). The vertical and horizontal linkages in such nested polycentric institutions could also contribute to integrated resource management goals by reducing or eliminating conflicts and competition among alternative uses (Marshall and Smith 2010; Pahl-Wostl et al. 2013). The governance mechanism in the Okavango River Basin provides a good illustration of nested institutions. Community-based conservation efforts at the local level, protection by constitutions and government ministries at the national level, a river basin commission at the watershed level, and international transboundary agreements on the management of the shared water resource provide multiple, overlapping levels of governance (Green et al. 2013).

Institutional Variety. The strategy of institutional variety addresses the need for diversity in the institutional mechanisms for adaptive governance processes (Dietz et al. 2003). Following Hardin's (1968) publication of "The tragedy of the commons," states and markets have usually been posited as panaceas in conventional resource management. However, a key insight from the common pool resource literature is that no single type of institution, whether states, markets, or communities, can be expected to function effectively, efficiently, and sustainably all the time (Ostrom et al. 1999). Since success and failures are associated with each of these types of institutions, the use of a diversity of institutions is an effective strategy for reducing the vulnerability of social-ecological systems (Berkes 2007). The issue of diversity is of central importance to resilience and adaptive governance. From an ecological perspective, functional and response diversity is an essential requirement for resilience (Hughes et al. 2005). Similarly, from the social perspective, institutional diversity enhances the integration of diverse perspectives from the different institutional approaches, thereby promoting the use of new ideas and techniques, as opposed to reliance on a single approach (Pahl-Wostl et al. 2013). This diversity enhances the overall resilience and effectiveness of the system. As Dietz et al. (2003, 1910) have noted, "Innovative rule evaders can have more trouble with a multiplicity of rules than with a single type of rule." For example, the governance of the Okavango River Basin involves community-based organizations, government representatives of the riparian states, and international non-governmental organizations in promoting adaptive and integrated management (Green et al. 2013). Similarly, the Blackfoot Challenge in Montana, a collaborative mechanism which emerged to address various social and ecological challenges in the 1.5 million acre Blackfoot watershed in the 1990s, comprised diverse actors, including private landowners, state representatives, federal agencies, and conservation organizations, such as the Nature Conservancy (Weber 2012; Wyborn and Bixler 2013).

Challenges in Adaptive Water Governance

The previous section of this paper highlighted the key features of adaptive water governance and their potentials for integrating the human dimensions into the governance of water resources in the face of uncertainty. However, a number of critical challenges will need to be overcome in order to realize the potentials of adaptive water governance (Akamani 2014).

At the philosophical level, differences among scientists, resource managers, and other stakeholders on fundamental issues, such as the value of nature

and the place of man in nature constitute potential stumbling blocks to adaptive water governance. For instance, lessons from ecosystem management suggest that differences in philosophical perspectives, ranging from anthropocentric to biocentric or ecocentric perspectives account for conflicts in the management process (Endter-Wada et al. 1998; Yaffee 1999; Behnken et al. 2016). Similarly, the limited implementation of active adaptive management and adaptive governance has been attributed to the lack of political will by decision-makers to embrace uncertainty (Walters 2007; Valman et al. 2015). In this regard, Dietz (2013) highlights the cognitive limitations in human ability to understand uncertainties and non-linear relationships in complex adaptive systems.

With regard to knowledge, the integration of scientific and non-scientific knowledge, as well as the promotion of social learning in adaptive governance, promise to offer a holistic understanding of complex social-ecological systems. However, epistemological differences in the foundations of western science and traditional knowledge, as well as differences in worldviews among stakeholders in social learning processes can constrain learning and knowledge integration (Berkes 2009; Reed et al. 2010). Also, institutional challenges, such as power imbalances among stakeholders, often lead to the marginalization of local and traditional knowledge (Houde 2007; Armitage et al. 2011). Finally, attitudinal issues, such as the lack of trust and mutual respect for local and traditional knowledge, need to be overcome (Berkes 2009; Armitage et al. 2012).

The institutional structures and processes of adaptive governance provide a promising mechanism for integrating diverse stakeholder values across scales, as well as learning and adapting to change and uncertainties. However, transitions toward such adaptive governance institutions can be constrained by the path-dependent effects of the dominant top-down institutional mechanisms associated with the command-and-control paradigm (Pahl-Wostl et al. 2007, 2009; Valman et al. 2015). The complexity associated with the diversity of interests and values across the multiple scales of adaptive governance also presents challenges for collective decision-making (Ostrom et al. 1999; Huitema et al. 2009; Dietz 2013). Importantly, Koontz et al. (2015) have cautioned that adaptive governance institutions could potentially yield undesirable social and ecological outcomes, such as widening inequalities and resource degradation. Contextual factors, such as the influence of vested interests, as well as pre-existing inequalities in access to power, information, resources, and opportunities could shape the outcomes of adaptive water governance. For instance, Mostert (2012) analyzed polycentric water governance on the small Dutch Island of IJsselmonde over a thousand-year period and found that contrary to the expected flexibility and adaptability of such systems, change was often slow due to the role of vested interests, the lack of willing and capable leaders at higher levels, and emotional attachments to old institutions, among other constraints.

Conclusion

The shortfalls associated with the command-and-control paradigm of water resource management have created awareness about the need to integrate the human dimensions and to prepare for uncertainties in the governance of water resources. In this regard, this paper has argued that adaptive water governance addresses these challenges as it provides opportunities for integrating the human dimensions into the assumptions, management goals, knowledge systems, and institutional mechanisms for water resource governance, while also allowing flexibility in the face of uncertainty. Expected benefits of such integration include enhancing good governance and sustainability and resilience in water resource systems. In spite of these benefits, it is worthwhile to note that adaptive governance is not a panacea (Koontz et al. 2015). The meaning of adaptive water governance remains ambiguous, its implementation may be constrained by several factors, and outcomes may not always be equitable or sustainable. Greater theoretical and methodological contributions from researchers across various social science disciplines, as well as enhanced awareness, willingness, and capacity of decision-makers, resource managers, and the general public is needed to enrich our understanding of adaptive water governance and to enhance its successful implementation.

Acknowledgements

This work was supported by the USDA National Institute of Food and Agriculture McIntire Stennis grant, project 1006249. I would also like to acknowledge the helpful comments from the two anonymous reviewers that helped in enriching the discussion of the major themes in the manuscript.

Author Bio and Contact Information

Kofi Akamani is Assistant Professor of Forest Recreation and Conservation Social Science in the Department of Forestry at Southern Illinois University. His teaching and research involves the integration of concepts from social-ecological systems research and rural sociology among other fields with the aim of gaining a theoretical understanding of human-environment interactions across multiple scales and informing policies that promote human well-being and ecosystem health across the rural urban continuum. He is particularly interested in understanding and enhancing the resilience of forest-dependent communities and the adaptive governance of social-ecological systems. He can be reached at k.akamani@siu.edu, or Department of Forestry, Southern Illinois University, 1205 Lincoln Drive, Mail Code 4411, Carbondale, IL 62901, USA.

References

Adger, W.N., T.P. Hughes, C. Folke, S.R. Carpenter, and J. Rockström. 2005. Social-ecological resilience to coastal disasters. *Science* 309(5737): 1036-1039.

Akamani, K. 2014. Challenges in the transition toward adaptive water governance. In: *Water Conservation: Practices, Challenges and Future Implications,* M.A. Imtaez (Ed.). Nova Science Publishers, Hauppauge, New York, pp. 165-178.

Akamani, K., E.J. Holzmueller, and J.W. Groninger. 2016. Managing wicked environmental problems as complex social-ecological systems: The promise of adaptive governance. In: *Landscape Dynamics, Soils and Hydrological Processes in Varied Climates,* A. Melesse and W. Abtew (Eds.). Springer, New York, pp. 741-762.

Akamani, K. and I.P. Wilson. 2011. Toward the adaptive governance of transboundary water resources. *Conservation Letters* 4(6): 409-416.

Alberti, M., J.M. Marzluff, E. Shulenberger, G. Bradley, C. Ryan, and C. Zumbrunnen. 2003. Integrating humans into ecology: Opportunities and challenges for studying urban ecosystems. *BioScience* 53(12): 1169-1179.

Allan, C. and A. Curtis. 2005. Nipped in the bud: Why regional scale adaptive management is not blooming. *Environmental Management* 36(3): 414-425.

Allen, C.R., J.J. Fontaine, K.L. Pope, and A.S. Garmestani. 2011. Adaptive management for a turbulent future. *Journal of Environmental Management* 92(5): 1339-1345.

Allen, C.R. and L.H. Gunderson. 2011. Pathology and failure in the design and implementation of adaptive management. *Journal of Environmental Management* 92(5): 1379-1384.

Anderies, J.M., M.A. Janssen, and E. Ostrom. 2004. A framework to analyze the robustness of social-ecological systems from an institutional perspective. *Ecology and Society* 9(1): 18.

Armitage, D., F. Berkes, A. Dale, E. Kocho-Schellenberg, and E. Patton. 2011. Co-management and the co-production of knowledge: Learning to adapt in Canada's Arctic. *Global Environmental Change* 21(3): 995-1004.

Armitage, D., R. de Loe, and R. Plummer. 2012. Environmental governance and its implications for conservation practice. *Conservation Letters* 5(4): 245-255.

Armitage, D.R., R. Plummer, F. Berkes, R.I. Arthur, A.T. Charles, I.J. Davidson-Hunt, A.P. Diduck, N.C. Doubleday, D.S. Johnson, M. Marschke, P. McConney, E.W. Pinkerton, and E.K. Wollenberg. 2009. Adaptive co-management for social-ecological complexity. *Frontiers in Ecology and the Environment* 7(2): 95-102.

Aytur, S.A., J.S. Hecht, and P. Kirshen. 2015. Aligning climate change adaptation planning with adaptive governance: Lessons from Exeter, NH. *Journal of Contemporary Water Research and Education* 155: 83-98.

Balint, P.J., R.E. Stewart, and A. Desai. 2011. *Wicked Environmental Problems: Managing Uncertainty and Conflict.* Island Press, Washington, D.C.

Bark, R.H., D.E. Garrick, C.J. Robinson, and S. Jackson. 2012. Adaptive basin governance and the prospects for meeting indigenous water claims. *Environmental Science & Policy* 19: 169-177.

Behnken, J.A., J.W. Groninger, and K. Akamani. 2016. Institutional constraints to collaborative ecosystem management within a wetlands conservation partnership. *Journal of Contemporary Water Research & Education* (in press).

Benson, M.H. and A.S. Garmestani. 2011. Embracing panarchy, building resilience and integrating

adaptive management through a rebirth of the National Environmental Policy Act. *Journal of Environmental Management* 92(5): 1420-1427.

Berkes, F. 2003. Alternatives to conventional management: Lessons from small-scale fisheries. *Environments* 31(1): 5-19.

Berkes, F. 2004. Rethinking community-based conservation. *Conservation Biology* 18(3): 621-630.

Berkes, F. 2007. Understanding uncertainty and reducing vulnerability: Lessons from resilience thinking. *Natural Hazards* 41(2): 283-295.

Berkes, F. 2009. Evolution of co-management: Role of knowledge generation, bridging organizations and social learning. *Journal of Environmental Management* 90(5): 1692-1702.

Berkes, F., J. Colding, and C. Folke. 2000. Rediscovery of traditional ecological knowledge as adaptive management. *Ecological Applications* 10(5): 1251-1262.

Biswas, A.K. 2004. Integrated water resources management: A reassessment. *Water International* 29(2): 248-256.

Blomquist, W. and E. Schlager. 2005. Political pitfalls of integrated watershed management. *Society and Natural Resources* 18(2): 101-117.

Brown, G., M.F. Schebella, and D. Weber. 2014. Using participatory GIS to measure physical activity and urban park benefits. *Landscape and Urban Planning* 121: 34-44.

Bugos, E., R. Frasso, E. FitzGerald, G. True, A.M. Adachi-Mejia, and C. Cannuscio. 2014. Practical guidance and ethical considerations for studies using photo-elicitation interviews. *Preventing Chronic Disease* 11(E189): 1-9.

Carpenter, S.R., B. Walker, M.J. Anderies, and N. Abel. 2001. From metaphor to measurement: Resilience of what to what. *Ecosystems* 4: 765-781.

Cash, D.W., W.N. Adger, F. Berkes, P. Garden, L. Lebel, P. Olsson, L. Pritchard, and O. Young. 2006. Scale and cross-scale dynamics: Governance and information in a multilevel world. *Ecology and Society* 11(2): 8.

Chapin, F.S., S.R. Carpenter, G.P. Kofinas, C. Folke, N. Abel, W.C. Clark, P. Olsson, D.M.S. Smith, B. Walker, O.R. Young, F. Berkes, R. Biggs, J.M. Grove, R.L. Naylor, E. Pinkerton, W. Steffen, and F.J. Swanson. 2009. Ecosystem stewardship: Sustainability strategies for a rapidly changing planet. *Trends in Ecology & Evolution* 25(4): 241-249.

Clark, T.W. 2002. *The Policy Process: A Practical Guide for Natural Resource Professionals*. Yale University Press, New Haven, CT.

Cole, D.N. and L. Yung. 2010. *Beyond Naturalness: Rethinking Park and Wilderness Stewardship in an Era of Rapid Change*. Island Press, Washington, D.C.

Corfee-Morlot, J., I. Cochran, S. Hallegatte, and P.J. Teasdale. 2011. Multilevel risk governance and urban adaptation policy. *Climatic Change* 104(1): 169-197.

Cortner, H. and M.A. Moote. 1999. *The Politics of Ecosystem Management*. Island Press, Washington, D.C.

Cosens, B.A. 2013. Legitimacy, adaptation, and resilience in ecosystem management. *Ecology and Society* 18(1): 3.

Costanza, R. 1996. Ecological economics: Reintegrating the study of humans and nature. *Ecological Applications* 6(4): 978-990.

Crona, B.I. and J.N. Parker. 2012. Learning in support of governance: Theories, methods, and a framework to assess how bridging organizations contribute to adaptive resource governance. *Ecology and Society* 17(1): 32.

Dietz, T. 2013. Bringing values and deliberation to science communication. *Proceedings of the National Academy of Sciences* 110(3): 14081-14087.

Dietz, T., E. Ostrom, and P.C. Stern. 2003. The struggle to govern the commons. *Science* 302(5652): 1907-1912.

Dietz, T. and P.C. Stern. 1998. Science, values, and biodiversity. *Bioscience* 48(6): 441-444.

Eigenbrode, S.D., M. O'Rourke, J. Wulfhorst, D.M. Althoff, C.S. Goldberg, K. Merrill, W. Morse, M. Nielsen-Pincus, J. Stephens, L. Winowiecki, N.A. Bosque-Perez. 2007. Employing philosophical dialogue in collaborative science. *BioScience* 57(1): 55-64.

Endter-Wada, J., D. Blahna, R. Krannich, and M. Brunson. 1998. A framework for understanding social science contributions to ecosystem management. *Ecological Applications* 8(3): 891-904.

Engle, N.L., O.R. Johns, M.C. Lemos, and D.R. Nelson. 2011. Integrated and adaptive management of water resources: Tensions, legacies, and the next best thing. *Ecology and Society* 16(1): 19.

Floress, K., K. Akamani, K.E. Halvorsen, A.T. Kozich, and M. Davenport. 2015. The role of social science in successfully implementing watershed management strategies. *Journal of Contemporary Water Research & Education* 154(1): 85-105.

Folke, C. 2006. Resilience: The emergence of a perspective for social–ecological systems analyses. *Global Environmental Change* 16(3): 253-267.

Folke, C. 2007. Social–ecological systems and adaptive governance of the commons. *Ecological Research* 22(1): 14-15.

Folke, C., S. Carpenter, T. Elmqvist, L. Gunderson, C.S. Holling, and B. Walker. 2002. Resilience and sustainable development: Building adaptive capacity in a world of transformations. *Ambio* 31(5): 437-440.

Folke, C., S.R. Carpenter, B. Walker, M. Scheffer, T. Chapin, and J. Rockström. 2010. Resilience thinking: Integrating resilience, adaptability and transformability. *Ecology and Society* 15(4): 20.

Folke, C., T. Hahn, P. Olsson, and J. Norberg. 2005. Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30: 441-473.

Folke, C., A. Jansson, J. Rockström, P. Olsson, S.R. Carpenter, F.S. Chapin III, A.S. Crepin, G. Daily, K. Danell, J. Ebbesson, T. Elmqvist, V. Galaz, F. Moberg, M. Nilsson, H. Osterblom, E. Ostrom, A. Persson, G. Peterson, S. Polasky, W. Steffen, B. Walker, F. Westley. 2011. Reconnecting to the biosphere. *Ambio* 40(7): 719-738.

Folke, C., L. Pritchard, F. Berkes, J. Colding, and U. Svedin. 2007. The problem of fit between ecosystems and institutions: Ten years later. *Ecology and Society* 12(1): 30.

Fuller, J., W. Hermeston, M. Passey, T. Fallon, and K. Muyambi. 2012. Acceptability of participatory social network analysis for problem-solving in Australian Aboriginal health service partnerships. *BMC Health Services Research* 12: 152.

Gain, A.K., J.J. Rouillard, and D. Benson. 2013. Can integrated water resources management increase adaptive capacity to climate change adaptation? A critical review. *Journal of Water Resource and Protection* 5: 11-20.

Gelcich, S., T.P. Hughes, P. Olsson, C. Folke, O. Defeo, M. Fernández, S. Foale, L.H. Gunderson, C. Rodriguez-Sickert, M. Scheffer, R.S. Steneck, and J.C. Castilla. 2010. Navigating transformations in governance of Chilean marine coastal resources. *Proceedings of the National Academy of Sciences* 107(39): 16794-16799.

Global Water Partnership. 2000. *Integrated Water Resources Management*. Stockholm.

Green, O.O., B.A. Cosens, and A.S. Garmestani. 2013. Resilience in transboundary water governance: The Okavango River Basin. *Ecology and Society* 18(2): 23.

Gregory, R., L. Failing, and P. Higgins. 2006. Adaptive management and environmental decision making: A case study application to water use planning. *Ecological Economics* 58(2): 434-447.

Gunderson, L. 1999. Resilience, flexibility and adaptive management—Antidotes for spurious certitude? *Conservation Ecology* 3(1): 7.

Gunderson, L. and S.S. Light. 2006. Adaptive management and adaptive governance in the everglades ecosystem. *Policy Sciences* 39(4): 323-334.

Hardin, G. 1968. The tragedy of the commons. *Science* 162(3859): 1243-1248.

Healey, M.C. 1998. Paradigms, policies and prognostications about the management of watershed ecosystems. In: *River Ecology and Management: Lessons From the Pacific Coastal Ecoregion,* R.J. Naiman and R.E. Billy (Eds.). Springer, New York, NY, pp. 662-682.

Holling, C.S. (Ed.). 1978. *Adaptive Environmental Assessment and Management*, John Wiley and Sons, New York.

Holling, C.S. 2001. Understanding the complexity of economic, ecological, and social systems. *Ecosystems* 4(5): 390-405.

Holling, C.S. 2012. Response to "Panarchy and the law". *Ecology and Society* 17(4): 37.

Holling, C.S. and G.K. Meffe. 1996. Command and control and the pathology of natural resource management. *Conservation Biology* 10(2): 328-337.

Houde, N. 2007. The six faces of traditional ecological knowledge: Challenges and opportunities for Canadian co-management arrangements. *Ecology and Society* 12(2): 34.

Hughes, T.P., D.R. Bellwood, C. Folke, R.S. Steneck, and J. Wilson. 2005. New paradigms for supporting the resilience of marine ecosystems. *Trends in Ecology & Evolution* 20(7): 380-386.

Huitema, D., E. Mostert, W. Egas, S. Moellenkamp, C. Pahl-Wostl, and R. Yalcin. 2009. Adaptive water governance: Assessing the institutional prescriptions of adaptive (co-) management from a governance perspective and defining a research agenda. *Ecology and Society* 14(1): 26.

Jacobson, S.K. and M.D. Duff. 1998. Training idiot savants: The lack of human dimensions in conservation biology. *Conservation Biology* 12(2): 263-267.

Karkkainen, B.C. 2005. Transboundary ecosystem governance: Beyond sovereignty. In: *Public Participation in the Governance of International Freshwater Resources,* C. Bruch, L. Janskey, and M. Nkayama (Eds.). United Nations University Press, New York, NY, pp. 72-87.

Karpouzoglou, T., A. Dewulf, and J. Clark. 2016. Advancing adaptive governance of social-ecological systems through theoretical multiplicity. *Environmental Science & Policy* 57: 1-9.

Koontz, T.M., D. Gupta, P. Mudliar, and P. Ranjan. 2015. Adaptive institutions in social-ecological systems governance: A synthesis framework. *Environmental Science & Policy*. Available at http://dx.doi.org/10.1016/j.envsci.2015.01.003. Accessed June 10, 2016.

Lebel, L., R. Daniel, N. Badenoch, P. Garden, and M. Imamura. 2008. A multi-level perspective on conserving with communities: Experiences from upper tributary watersheds in montane mainland Southeast Asia. *International Journal of the Commons* 2(1): 127-154.

Lee, K.N. 1999. Appraising adaptive management. *Conservation Ecology* 3(2): 3.

Lélé, S. and R.B. Norgaard. 2005. Practicing interdisciplinarity. *BioScience* 55(11): 967-975.

Liu, J., T. Dietz, S.R. Carpenter, M. Alberti, C. Folke, E. Moran, A.N. Pell, P. Deadman, T. Kratz, J. Lubchenco, E. Ostrom, Z. Ouyang, W. Provencher, C.L. Redman, S.H. Schneider, and W.W. Taylor. 2007. Complexity of coupled human and natural systems. *Science* 317(5844): 1513-1516.

Madsen, W. and C. O'Mullan. 2016. Perceptions of community resilience after natural disaster in a rural Australian town. *Journal of Community Psychology* 44(3): 277-292.

Manfredo, M.J., T.L. Teel, M.C. Gavin, and D. Fulton. 2014. Considerations in representing human individuals in social-ecological models. In: *Understanding Society and Natural Resources: Forging New Strands of Integration Across the Social Sciences,* M.J. Manfredo, J.J. Vaske, A. Rechkemmer, and E.A. Duke (Eds.). Springer Open, New York, pp. 137-158.

Marshall, G.R. 2009. Polycentricity, reciprocity, and farmer adoption of conservation practices under community-based governance. *Ecological Economics* 68(5): 1507-1520.

Marshall, G.R. and D.M.S. Smith. 2010. Natural resources governance for the drylands of the Murray–Darling Basin. *The Rangeland Journal* 32(3): 267-282.

Mascia, M.B., J.P. Brosius, T.A. Dobson, B.C. Forbes, L. Horowitz, M.A. McKean, and N.J. Turner. 2003. Conservation and the social sciences. *Conservation Biology* 17(3): 649-650.

McLain, R.J. and R.G. Lee. 1996. Adaptive management: Promises and pitfalls. *Environmental Management* 20(4): 437-448.

Measham, T.G. 2013. How long does social learning take? Insights from a longitudinal case study. *Society and Natural Resources* 26(12): 1468-1477.

Medema, W., B.S. McIntosh, and P.J. Jeffrey. 2008. From premise to practice: A critical assessment of integrated water resources management and adaptive management approaches in the water sector. *Ecology and Society* 13(2): 29.

Molle, F. 2009. River-basin planning and management: The social life of a concept. *Geoforum* 40(3): 484-494.

Morris, M. and R.C. de Loe. 2016. Cooperative and adaptive transboundary water governance in Canada's Mackenzie River Basin: Status and prospects. *Ecology and Society* 21(1): 26.

Mostert, E. 2012. Water management on the Island of IJsselmonde 1000 to 1953: Polycentric governance, adaptation, and petrification. *Ecology and Society*, 17(3):12.

Muro, M. and P. Jeffrey. 2012. Time to talk? How the structure of dialog processes shapes stakeholder learning in participatory water resources management. *Ecology and Society* 17(1): 3.

Nelson, R., M. Howden, and M.S. Smith. 2008. Using adaptive governance to rethink the way science supports Australian drought policy. *Environmental Science & Policy* 11(7): 588-601.

Olsson, P. and C. Folke, C. 2001. Local ecological knowledge and institutional dynamics for ecosystem management: A study of Lake Racken watershed, Sweden. *Ecosystems* 4(2): 85-104.

Olsson, P., C. Folke, and F. Berkes. 2004. Adaptive comanagement for building resilience in social–ecological systems. *Environmental Management* 34(1): 75-90.

Olsson, P., C. Folke, and T.P. Hughes. 2008. Navigating the transition to ecosystem-based management of the Great Barrier Reef, Australia. *Proceedings of the National Academy of Sciences* 105(28): 9489-9494.

Olsson, P. and V. Galaz. 2009. Transitions to adaptive approaches to water management and governance in Sweden. In: *Water Policy Entrepreneurs: A Research Companion to Water Transitions Around the Globe,* D. Huitema and S. Meijerink (Eds.). Edward Elgar, Northampton, MA, pp. 304-324.

Olsson, P., V. Galaz, and W.J. Boonstra. 2014. Sustainability transformations: A resilience perspective. *Ecology and Society* 19(4): 1.

Olsson, P., L.H. Gunderson, S.R. Carpenter, P. Ryan, L. Lebel, C. Folke, and C.S. Holling. 2006. Shooting the rapids: Navigating transitions to adaptive governance of social-ecological systems. *Ecology and Society* 11(1): 18.

Österblom, H. and C. Folke. 2013. Emergence of global adaptive governance for stewardship of regional marine resources. *Ecology and Society* 18(2): 4.

Ostrom, E. 1999. Coping with tragedies of the commons. *Annual Review of Political Science* 2(1): 493-535.

Ostrom, E. 2010. Polycentric systems for coping with collective action and global environmental change. *Global Environmental Change* 20(4): 550-557.

Ostrom, E., J. Burger, C.B. Field, R.B. Norgaard, and D. Policansky. 1999. Revisiting the commons: Local lessons, global challenges. *Science* 284(5412): 278-282.

Pahl-Wostl, C. 2007a. The implications of complexity for integrated resources management. *Environmental Modelling & Software* 22(5): 561-569.

Pahl-Wostl, C. 2007b. Transitions towards adaptive management of water facing climate and global change. *Water Resources Management* 21(1): 49-62.

Pahl-Wostl, C., M. Palmer, and K. Richards. 2013. Enhancing water security for the benefits of humans and nature - The role of governance. *Current Opinion in Environmental Sustainability* 5(6): 676-684.

Pahl-Wostl, C., J. Sendzimir, and P. Jeffrey. 2009. Resources management in transition. *Ecology and Society* 14(1): 46.

Pahl-Wostl, C., J. Sendzimir, P. Jeffrey, J. Aerts, G. Berkamp, and L. Cross. 2007. Managing change toward adaptive water management through social learning. *Ecology and Society* 12(1): 30.

Plummer, R. 2009. The adaptive co-management process: An initial synthesis of representative models and influential variables. *Ecology and Society* 14(2): 24.

Plummer, R., D. de Grosbois, R. de Loë and J. Velaniškis. 2011. Probing the integration of land use and watershed planning in a shifting governance regime. *Water Resources Research* 47: 1-13.

Raadgever, G., E. Mostert, N. Kranz, E. Interwies, and T.G. Timmerman. 2008. Assessing management regimes in transboundary river basins: Do they support adaptive management? *Ecology and Society* 13(1):14.

Redman, C.L., J.M. Grove, and L.H. Kuby. 2004. Integrating social science into the Long-Term Ecological Research (LTER) Network: Social dimensions of ecological change and ecological dimensions of social change. *Ecosystems* 7: 161-171.

Reed, M., A.C. Evely, G. Cundill, I. Fazey, J. Glass, A. Laing, J. Newig, B. Parrish, C. Prell, C. Raymond, and L.C. Stringer. 2010. What is social learning? *Ecology and Society* 15(4).

Ringold, P.L., J. Alegria, R.L. Czaplewski, B.S. Mulder, T. Tolle, and K. Burnett. 1996. Adaptive monitoring design for ecosystem management. *Ecological Applications* 6(3): 745-747.

Schultz, L., C. Folke, H. Osterblom, and P. Olsson. 2015. Adaptive governance, ecosystem management, and natural capital. *PNAS* 112(24): 7369-7374.

Sendzimir, J., P. Magnuszewski, Z. Flachner, P. Balogh, G. Molnar, A. Sarvari, and Z. Nagy. 2008. Assessing the resilience of a river management regime: Informal learning in a shadow network in the Tisza River Basin. *Ecology and Society* 13(1): 11.

Sivapalan, M., M. Konar, V. Srinivasan, A. Chhatre, A. Wutich, C. Scott, J.L. Westcoat, and I. Rodriguez-Iturbe. 2014. Socio-hydrology: Use-inspired water sustainability science for the Anthropocene. *Earth's Future* 2(4): 225-230.

Sivapalan, M., H.H.G. Savenije, and G. Blochl. 2011. Socio-hydrology: A new science of people and water. *Hydrological Processes*. DOI: 10.1002/hyp.8426. Accessed June 10, 2016.

Smith, C.B. 2011. Adaptive management on the central Platte River–Science, engineering, and decision analysis to assist in the recovery of four species. *Journal of Environmental Management* 92(5): 1414-1419.

Stankey, G.H., B.T. Bormann, C. Ryan, B. Shindler, V. Sturtevant, R.N. Clark, and C. Philpot. 2003. Adaptive management and the Northwest Forest Plan: Rhetoric and reality. *Journal of Forestry* 101(1): 40-46.

Tuler, S. and T. Webler. 1999. Designing an analytic deliberation process for environmental health policy making in the U.S. nuclear weapons complex. *Risk* 10(1): 65-87.

Turner, B.L., R.E. Kasperson, P.A. Matson, J.J. McCarthy, R.W. Corell, L. Christensen, N. Eckley, J.X. Kasperson, A. Luers, M.L. Martello, C. Polsky, A. Pulsipher, and A. Schiller. 2003. A framework for vulnerability analysis in sustainability science. *PNAS* 100(14): 8074-8079.

Valman, M., H. Osterblom, and P. Olsson. 2015. Adaptive governance of the Baltic Sea – Lessons from elsewhere. *International Journal of the Commons* 9(1): 440-465.

Von der Porten, S. and R.C. de Loe. 2014. Water policy reform and indigenous governance. *Water Policy* 16: 222-243.

Walker, B. 2012. A commentary on "Resilience and water governance: Adaptive governance in the Columbia River Basin". *Ecology and Society* 1(2): 1.

Walters, C. 1986. *Adaptive Management of Renewable Resources*. McMillan, New York, NY.

Walters, C. 1997. Challenges in adaptive management of riparian and coastal ecosystems. *Conservation Ecology* 1(2): 1.

Walters, C.J. 2007. Is adaptive management helping to solve fisheries problems? *Ambio* 36(4): 304-307.

Walters, C.J. and C.S. Holling. 1990. Large-scale management experiments and learning by doing. *Ecology* 71(6): 2060-2068.

Wang, C.C. 1999. Photovoice: A participatory action research strategy applied to women's health. *Journal of Women's Health* 8(2): 185-191.

Weber, E.P. 2012. Getting to resilience in a climate-protected community: Early problem-solving choices, ideas, and governance philosophy. In: *Collaborative Resilience: Moving Through Crisis to Opportunity*, B.E. Goldstein (Ed.). The MIT Press, Cambridge, MA, pp. 177-206.

Webler, T. and S. Tuler. 1999. Integrating technical analysis with deliberation in regional watershed management planning: Applying the National Research Council approach. *Policy Studies Journal* 27(3): 530-543.

West, S.P. and L. Schultz. 2015. Learning for resilience in the European Court of Human Rights: Adjudication as an adaptive governance practice. *Ecology and Society* 20(1): 31.

White, L. 1967. The historical roots of our ecological crisis. *Science* 155(37671): 1203-1207.

Wyborn, C. and R.P. Bixler. 2013. Collaboration and nested environmental governance: Scale dependency, scale framing, and cross-scale interactions in collaborative conservation. *Journal of Environmental Management* 123: 58-67.

Yaffee, S.L. 1999. Three faces of ecosystem management. *Conservation Biology* 13(4): 713-725.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 19-33, August 2016

Institutional Constraints to Collaborative Ecosystem Management within a Wetlands Conservation Partnership

Jennifer A. Behnken, *John W. Groninger, and Kofi Akamani

Department of Forestry, Southern Illinois University, Carbondale, IL, USA
**Corresponding Author*

Abstract: Recent decades have seen an evolution in resource management paradigms from the multiple use and sustained yield toward ecosystem management. This ongoing paradigm shift has generated interest in collaborative decision-making as a more promising alternative to conventional top down institutions. While a collaborative approach to ecosystem management promises several benefits, including equity, efficiency, and sustainability, institutional constraints often impede the success of these initiatives. Using the Cache River Joint Venture Partnership in southern Illinois, USA as a case study, this study employed a qualitative research approach to understand how various institutional attributes influence prospects for conflict and cooperation in collaborative decision-making processes. We found that stakeholder motivation and capacity for cooperation with other partners in ecosystem management can be enabled or constrained by several institutional factors, including external policy and regulatory mandates, internal administrative and budget structures, institutional cultures, and approaches to science and decision-making criteria. We conclude with a suggestion that stakeholders in collaborative ecosystem management partnerships identify and address constraints for cooperative action during the early stages of the process.

Keywords: *adaptive management, adaptive governance, Cache River Joint Venture Partnership, collaboration, restoration, southern Illinois, sustainability, watershed*

The resource management paradigm is in a state of transition from the conventional sustained yield and multiple use management paradigm toward ecosystem management (Cortner and Moote 1999; Yaffee 1999; Chapin et al. 2010; Folke et al. 2011). The sustained yield and multiple use management paradigm was based on the view of humans as separate from nature and a belief in human mastery over nature (Cortner et al. 1998). This paradigm also relied on expert science and centralized institutions with the aim of enhancing efficiency and output maximization of economically valuable resources, including timber and water resources (Daniels and Walker 1996; Yaffee 1999). This over-emphasis on the predictable supply of commodity outputs has resulted in biodiversity crisis (Grumbine 1994; Chapin et al. 2010). Since the concept of sustainable development emerged in the 1980s, the paradigm of the ecosystem management has been gaining recognition as an alternative to the conventional paradigm (Cortner and Moote 1999). The turn toward ecosystem management in recent decades is a response to the biodiversity crises created by the narrow focus of conventional management approaches, as well as changing societal values, and new scientific insights from recent disciplines such as conservation biology and landscape ecology (Grumbine 1994; Cortner and Moote 1999).

Ecosystem management is a holistic approach to resource management that addresses the sustainability of human and ecological communities (Endter-Wada et al. 1998). In this regard, ecosystem management departs from the narrow focus of the conventional paradigm by emphasizing the pursuit of integrated management goals that cover social, ecological, and economic

dimensions (Keogh and Blahna 2005). As Stein and Gelburd (1998, 74) have succinctly stated, "The goal of the ecosystem approach is to restore and sustain the health, productivity, and biological diversity of ecosystems and the overall quality of life through a natural resource management approach that is fully integrated with social and economic goals." The ecosystem management paradigm considers humans an integral part of nature (Folke et al. 2011), and it recognizes the uncertainties that characterize the dynamic interaction between social and ecological systems (Yaffee 1996). In view of these uncertainties, ecosystem management highlights the need for interdisciplinary science and adaptive management processes (Cortner et al. 1998; Butler and Koontz 2005). Ecosystem management also emphasizes a bioregional approach that considers resource management issues across multiple political and administrative boundaries (Lee 1999). Examination of resource management goals across multiple spatial and temporal scales reveals significant conflicts and other organizational challenges that cannot be managed through conventional top-down institutional mechanisms (Daniels and Walker 1996; Bonnell and Koontz 2007). In this regard, collaborative decision-making is increasingly relied upon as a means of dealing with conflicts among the multiple stakeholders in ecosystem management (Imperial 1999; Wondolleck and Yaffee 2000; Keogh and Blahna 2005).

Besides the influence of the ecosystem management paradigm, the turn toward collaborative resource management is also fueled by the increasing devolution of decision-making authority and implementation responsibilities to state and non-state actors at lower administrative levels (Steel and Weber 2001). In this regard, co-management or collaboration can be seen as a form of governance (Carlsson and Berkes 2005), defined as "the whole of public as well as private interactions to solve societal problems and create societal opportunities" (Berkes 2010, 491). Collaborative decision-making entails the use of multiparty and multidisciplinary approaches to problem-solving (Yaffee 1996). Keogh and Blahna (2005) contend that collaboration goes beyond public involvement. Collaborative processes are characterized by inclusiveness, sharing of power and responsibilities, and joint decision-making among stakeholders, such as government representatives and resource users who interact as equals (Bonnell and Koontz 2007; Berkes 2010). The emergence of collaborative partnerships becomes the solution in natural systems requiring joint efforts to protect and enhance land and water resources at the landscape level (Wondolleck and Yaffee 2000). Collaboration may increase the impact of conservation and restoration practices beyond the capabilities of individual parties through the collective use of critical resources, including land, funding, personnel, or facilities (Burde et al. 1998; Carlsson and Berkes 2005). Additionally, collaborative partnerships improve the quality of resource management decisions through the integration of diverse perspectives and sources of knowledge among stakeholders (Margerum and Robinson 2015). Collaboration is also a promising mechanism for managing stakeholder conflicts, enhancing the efficiency of the decision-making process through reduced transaction costs, improving the legitimacy of decisions, and enhancing the sustainability of resources and livelihoods (Carlsson and Berkes 2005; Plummer and Armitage 2007; Cinner et al. 2012; Margerum and Robinson 2015).

While collaborative partnerships between governmental and non-governmental entities are essential for ecosystem management, several factors may inhibit their success. One legacy of the sustained yield and multiple use paradigm of resource management is the fragmentation of ownership and responsibilities for the management of ecosystems among land management organizations, such as the Forest Service, National Park Service, and the Bureau of Land Management (Yaffee 1997; Cortner and Moote 1999). Specifically, differences in mandates among the multiple land management organizations create conflicts and impede collaborative ecosystem management (Cortner et al. 1998; Haeuber 1998). Closely related to the constraints of differing legislative mandates, differences in budget structures and budget priorities among organizations are another constraint to collaboration (Stein and Gelburd 1998; Imperial 1999). Complexities inherent to watershed planning imply the need for flexibility within the planning process, but partnering

organizations may lack the pliancy in institutional policies and management operations needed to execute agreements or to accommodate unexpected changes in funding availability (Imperial 1999). This lack of flexibility may also stem from resistance to change by vested interests (Yaffee 1996) and the lack of reward for monitoring and experimentation (Butler and Koontz 2005). In view of these challenges, the literature does not yet adequately address institutional factors that influence cooperation among land management agencies, industries, and private landowners engaged in ecosystem management (Cortner et al. 1998; Imperial 1999).

The objective of this study is to understand how institutional characteristics impact land management decision-making and practices in collaborative ecosystem partnerships. Specifically, we explore the conditions underlying potential for conflicts and cooperation among institutional partners. We analyze these issues through a qualitative study of the Cache River Joint Venture Partnership (CRJVP), a land management partnership in the Cache River valley of southern Illinois, USA. The next section of the paper presents the context of the study area. Next, we present a description of methods for data collection and analysis. The subsequent section presents the results of the study, followed by a discussion of key findings. We then conclude with recommendations for institutions contemplating collaborative ecosystem partnerships in hydrologically complex watersheds.

Background

Forested wetlands in the mid-continental U.S. have been viewed as sources of timber, habitat for wildlife, impediments for row crop agriculture, assets for ecotourism development, and sinks for sediments and other pollutants. Extensive hydrologic alteration within these ecosystems has resulted in a variety of site conditions on landscapes held under multiple ownerships and managed for diverse objectives. Over the past century, state and federal natural resource agencies, non-profit organizations, and soil conservation programs for consumptive and non-consumptive recreation and resource management have emerged to address site specific management objectives on private and publicly held lands. More recently, coordination among landowning entities has created opportunities for management collaboration at multiple spatial scales. This cooperation is particularly important to address calls for restoring hydrologic connectivity, a key component for the renewed provision of some ecosystem services (Opperman et al. 2009).

The study was conducted within the Cache River watershed of southern Illinois (Figure 1). Renowned for its significant biological and cultural features including high value wetland habitats and unique landscape features, such as cypress-tupelo swamps, the Cache River watershed is recognized nationally and internationally with National Natural Landmarks and RAMSAR designations, respectively. Private land ownership with a focus on row crop agriculture is the primary land use across the watershed once dominated by forest cover and since transformed via land clearing and drainage (Durum et al. 2004). Concerns for hydrological and land use alterations prompted public land entities to actively explore opportunities for management interventions. The Cache River Joint Venture Partnership (CRJVP) was formed in 1991 with constituents including the Illinois Department of Natural Resources (IDNR), U.S. Fish and Wildlife Service (FWS), The Nature Conservancy (TNC), and Ducks Unlimited (DU). The USDA Natural Resource Conservation Service (NRCS) became a member of the CRJVP in 2008 to expand resource initiatives with conservation programs for private landowners within the watershed. The formation of the CRVJP was instrumental to addressing the scale and complexity of the efforts needed to protect and restore the Cache River watershed (Illinois Department of Natural Resources 2006). The primary goals of the CRJVP are: (1) forest and wetland habitat restoration; (2) reduction of sedimentation and streambank/bed erosion; and (3) managed reconnection of the Upper and Lower segments of the Cache River. The CRJVP currently controls or works within a 60,000 acre wetland corridor nearest the Cache River (Figure 1). While all members of the partnership work towards these broad goals, each partner approaches these goals through their own specific management objectives and actions.

Methods

This case study employed a qualitative research approach to gain an in-depth understanding of the institutional dimensions of the CRJVP from the perspectives of current, previous, and unofficial members of the partnership. Data for the study were generated using semi-structured interviews, observations, and document review (Creswell 2003). After gaining entry and establishing rapport with potential research participants, the chain referral method, or snowball method (Patton 1990) was used to purposively sample research participants (Baxter and Eyles 1997). Key informants were asked to identify other individuals who may provide a unique perspective of the CRJVP. Participants were chosen based on their membership status, knowledge, and experiences within the CRJVP. A total sample of twenty-five research participants were interviewed, including representatives of each of the member organizations of the CRJVP, as well as representatives of organizations working closely with the CRJVP, such as AmeriCorps and Friends of the Cache River Watershed. Most of these interviews were conducted in a one-on-one, face-to-face setting while a few were conducted via telephone. During the interviews, an interview guide was used to provide structure and consistency in questioning while permitting the flexibility for further probing to explore participants' unique experiences and perceptions (Patton 1990). The interviews explored participants' experiences in land and water resource management in the Cache River watershed as members of a particular organization and as partners in the collaborative initiatives of the CRJVP. Interviews averaged approximately one hour in length, ranging from thirty minutes

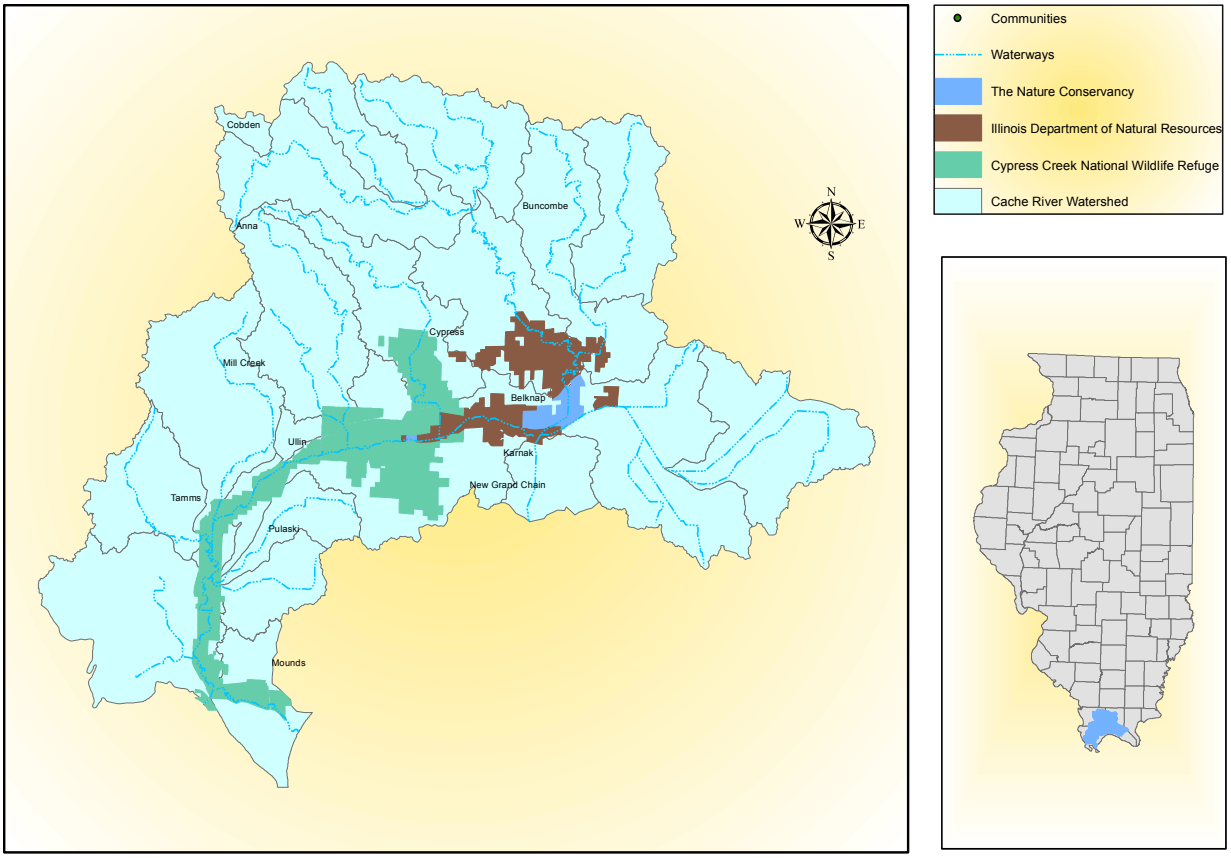


Figure 1. Study area. Public lands in the Cache River Watershed in southern Illinois (Source: Cache River Joint Venture Partnership).

to over two hours. All interviews were recorded using a digital voice recorder and transcribed post interview. The interviewer (JAB) took notes during the interview regarding initial observation of nuances, behaviors, emergent themes, and other commentary useful for collective assessment and future analysis. Reflexive reports were developed after each interview to document these initial reflections and impressions and to further identify thematic categories preceding coding (Shenton 2004).

In addition to interviews, participant observations at monthly meetings of the CRJVP were undertaken throughout the duration of the study, from March through October 2012. Additional observations were conducted at meetings where the CRJVP partners were involved, such as restoration committee meetings, the NRCS annual planning meeting, and the Friends of the Cache River Watershed monthly meetings. Observations were focused on discussion points of the meetings, current events, activities, concepts brought to the table for CRJVP discussion, and other issues requiring CRJVP attention. Data also were obtained from relevant documents including management plans, environmental studies, proposals, progress reports, articles, maps, institutional records, memoranda, and prior meeting minutes to gain in-depth knowledge on partnership history, goals and objectives, current practices, concerns and issues, and future outlooks.

Following the fieldwork, interview transcripts were imported into the Nvivo software to enhance data management. Data analysis and interpretation followed the guidelines of grounded theory principles (Strauss and Corbin 1998; Charmaz 2006). In grounded theory, the researcher seeks to derive a general explanation of a phenomenon of interest based on the views of the research participants (Creswell et al. 2007). Toward this end, the data were analyzed through coding to identify emerging themes from the data, as well as the relationships among these themes (Strauss and Corbin 1998). A number of measures were taken to enhance the rigor of the data analysis process, including the use of peer debriefing, and the triangulation of data from the interviews, document review, and participant observations (Creswell 2003).

Results

The following data analysis results are presented according to the major themes that emerged from the data: (1) policy mandates and regulatory ambiguities; (2) funding and budget mechanisms; (3) administrative structures and procedures; (4) institutional culture; and (5) science and decision-making criteria.

Policy mandates and regulatory ambiguities

The management of the Cache River watershed is divided among state and federal agencies, as well as the private entities that constitute the CRJVP. Each member of the partnership is bound by individual policy mandates that shape their authority and management priorities. The interviews indicated that these differences can lead to several challenges, including non-cooperation and limited implementation of the ecosystem management concept.

Differences in mandates among the CRJVP members seemed to explain the lack of interest of some partners in cooperating on some resource management projects. For instance, the IDNR is in a water structure agreement with an adjacent private landowner to control water levels. However, the structure has the potential to negatively impact other adjacent private lands. FWS is concerned that flooding private lands constitutes private property drainage which is inconsistent with the agency's mandate. As a result, FWS is unwilling to support other partners on projects that involve water level management as indicated by this interviewee:

"*There's a personal private property rights issue Fish and Wildlife Service has with the way IDNR is handling this private property issue and they're not supporting the way that IDNR is handling it. FWS has already said that if it [the issue] goes to court, that they're not going to be backing them and in many ways, they can't, because it does set a precedent that you're flooding somebody's land potentially that you know about and it's complicated and I don't see the partners coming together on these really big legal issues.*" (int. 22)

Key informants also mentioned that in instances where partners cooperate, unintended consequences may occur in partnership efforts as

a result of ambiguities in the multiple regulatory frameworks that may not be compatible among all parties. For instance, TNC enrolled their land in the Wetlands Reserve Program (WRP) administered by NRCS to enhance the protection of the land and to contribute to hydrologic reconnection efforts. WRP focuses on wetland restoration, protection, and enhancement, which is consistent with TNC's mission of enhancing the sustainability of land and water resource systems. However, additional regulations on flooding and changes in hydrology were enforced by the NRCS that could hinder the flexibility of TNC to undertake the management actions intended to promote reconnection.

"*Where we were proposing to divert the water was through a Wetland Reserve, WRP, and NRCS had some issues there. Can you really go into an easement and change the character of that easement, even if it's for reconnection?*" (int. 23)

While the problems discussed above are due to the multiple regulations by which members of the CRJVP must abide, conflicts have also arisen due to the lack of clear legal authorization of the landscape scale management actions required to achieve the goals of the CRJVP. In fact, some private landowners and the local Drainage District have obtained an injunction to halt IDNR water level management activities due to potential flooding impacts. As a result, IDNR cannot move forward with management until the lawsuits are resolved. Given the lack of authority of FWS to undertake management actions that impede drainage on private lands, the injunction against IDNR could delay or preclude the attainment of hydrologic reconnection goals.

Funding and budget mechanisms

Another theme that emerged from the interviews was the differences in funding sources and budget mechanisms among the CRJVP partners. These differences influence the priorities and capacity for each partner to contribute to the pursuit of common goals.

Interviews revealed that the CRJVP partners receive funding from private and public sources, and have differential levels of flexibility over the allocation of these funds. In the case of the FWS, the refuge manager has the flexibility to allocate budgets to specific on-site priorities. He also has the authority to transfer funds as needed from one project to the other.

"*It's kind of discretionary on [the] part of the refuge manager how much money they want to put towards biological stuff, visitor services, that type of thing. They're given categories of money for each of those things, like maintenance money or a general fund of money, but then it's kind of up to them on where they want that money to go... We are given an allotment, some money for each category, like maintenance, visitor services, but we can shuffle that money around. Nobody is stopping us from doing that. If the refuge manager wanted to say, take all the maintenance money and give it to biology, he could do that. He has the freedom and he's in charge of how we are going to spend our money basically.*" (int. 26)

The flexible budget mechanisms of the FWS differ radically from that of the IDNR which operates according to rigid budget structures that prohibit fund reallocation. Compared to IDNR, managers working for the privately-funded TNC have greater fiscal discretion. Besides managing their own properties, TNC even assists partners by providing additional funding for off-site restoration projects. However, projects that involve substantial funding require approval from the national level. Seeking funds at the national level is a lengthy process and can lead to delays in project implementation. These differences inhibit the implementation of CRJVP projects that require the pooling of resources among partners.

The interviews also established a relationship between funding sources and project priorities. Some CRJVP partners may be driven by funding availability to prioritize projects that may not always be consistent with the landscape scale goals being pursued by the CRJVP. The shortfalls of such project-driven management approaches are best illustrated in the case of the NRCS whose personnel concentrate their efforts primarily on private lands that are suitable for the various government-funded conservation projects administered by the agency. As a result, NRCS staff have few resources to actively participate in CRJVP projects that do not directly impact their private land owner constituents through the programs they administer.

Administrative structures and processes

Differences in the administrative structures of the members of the CRJVP were also highlighted by key informants. These differences have implications for the ability of partners to quickly respond to emerging resource management challenges. These differences are best illustrated in the contrast between the FWS and IDNR.

IDNR has a complex administrative structure that could potentially impede the flow of information, as well as decision-making at lower levels of the organization. The IDNR consists of several divisions, each of which has its own set of priorities focused on distinct resources, such as fisheries, forestry, natural heritage, and wildlife. District personnel work with the site superintendents and seek approval from them in order to carry out on-the-ground management actions. Regarding on-the-ground management, the staff members communicate between divisions to address site-specific needs. Communication also occurs between levels of hierarchy within individual divisions. For example, district forestry staff report to regional forestry supervisors and fisheries district staff report to their regional fisheries supervisors. At the apex of the chain of command is the overall director of IDNR, followed by the head of each of the divisions. This top-down approach to managing single resource categories may challenge the development of an enabling institutional framework for managing complex ecosystems.

The administrative structure of the FWS differs from that of IDNR. The refuge manager of the FWS possesses greater autonomy to make decisions on the ground as opposed to waiting for upper levels of administration to approve all management actions.

"*Refuge managers are god. They are given the control to make a decision on the ground, so the refuge managers do not have to call the regional office for every decision.*" (int. 23)

"*...let's say we've got a ditch that's plugged up somehow that's on us and a neighbor might feel it's a problem for them. The typical thing, we'd probably say yeah, we're gonna get to that. If [we] have neighbors that sometimes say, I'd like to go do that right now, we might give them a special use permit and say yeah, go ahead, you can do that.*" (int. 16)

In addition to differences in administrative structures of the CRJVP members, some key informants also mentioned the constraints associated with the procedural requirements of government agencies that distract from resource management activities.

"*It'd be nice if we were spending more time looking and thinking about ... resource management documents instead of more time on administrative stuff.*" (int. 16)

Institutional culture

Differences in the institutional cultures (i.e. the norms and traditions) of partner organizations of the CRJVP also emerged as an important source of disparities in the priorities and strategies of these organizations. These differences have implications for potential cooperation or non-cooperation on specific resource management initiatives.

The interviews revealed that the institutional members of the CRJVP focus on different scales in the analysis and management of resource management problems within the watershed, creating potential for conflicts. For instance, TNC defines its mission holistically and focuses on managing the entire watershed to recreate a sustainable, naturally flowing river system. As noted by one respondent, TNC aspires "...to actually protect, restore, and manage the significant natural character of the Cache River. So it involved more than just buying land..." (int. 18). Contrary to the landscape scale orientation of TNC, the traditional focus of NRCS is working with private landowners. The two most prevalent programs implemented in the Cache River watershed are the Environmental Quality Incentive Program (EQIP) and the Wetlands Reserve Program (WRP). Both programs provide financial incentives for landowners to implement best management practices that will benefit the landowners as well as support larger conservation efforts. These programs allow private landowners to retain their ownership rights while relinquishing some of their development rights.

"*In watersheds like the Cache, where we've had landowners that would have never sold their ground to Fish and Wildlife Service for part of the refuge, but yet as long as they can maintain ownership, we're willing to put it in the Wetland*

Reserve Program, and they maintain the ownership, but it's permanently enrolled in wetlands now and managed as a wetland restoration... The Wetland Reserve Program is tied nicely in that it's targeting some of those lands that never would have made it into federal ownership." (int. 12)

However, the unique relationship between NRCS and private landowners presents challenges for the pursuit of CRJVP goals. Private landowners are primarily concerned with sustaining agricultural production. While NRCS strives to meet the private landowners' needs while balancing best practices for natural resource conservation benefits, these two goals are not always compatible. In view of these conflicts between agricultural and conservation interests, representatives of the NRCS noted that their membership in the CRJVP sometimes threatens to erode the trust between the agency and private landowners.

"*Some of the conflicts the field office has are sometimes being pulled towards the interests of agriculture versus interests that the state and federal agencies have. So there's a delicate balance that the field offices try to maintain, especially when it comes to things like agricultural drainage or policy of how lands will be procured that are going to be federal lands, but there's a delicate balance of trust that we have between groups like the Drainage Districts, let's say. Our agency versus the Fish and Wildlife Service or IDNR, who they see as maybe a primary enemy of them in terms of drainage and stuff... we have to be a little careful there, when you're working with people… to do that in a way where it doesn't look like we're trying to take what some other agency wants to do and cram it down their throat. That's the hardest part and probably the toughest challenge. It's caused friction in the past.*" (int. 12)

"*Some of the landowners who don't agree with what the CRJVP's trying to do I actually saw it as a negative, because they didn't always want to work with us because we were in cahoots with the CRJVP. Down there, it was actually more of a disadvantage being a member of the [Cache River Joint Venture] Partnership.*" (int. 30)

Besides the conflicting scales of operation, many interviewees remarked that differences exist in the CRJVP members' fundamental assumptions about what constitutes a desirable ecosystem and what management interventions are needed to achieve that goal. Given the same piece of land, two partners may possess differing perspectives regarding the method of management and preferred management outcomes.

"*If [IDNR] buys a piece of land, [IDNR] looks at it, look at the public land survey notes, look at the adjacent habitat, [and] determine to the best of our ability what it would have looked like prior to human disturbance, and that's what we put back. Fish and Wildlife Service might look at the same piece of the land and if it's not high quality, they might say yeah, we want to put forest on here, but we have an opportunity to put three hundred acres of moist soil units, intensively managed unnatural system, that would provide benefits for waterfowl, which are, you know, part of their mandates, so they're going to do that.*" (int. 8)

These philosophical differences are best illustrated in the management of cypress-tupelo swamps, a natural community type that occurs on both state and federal lands in the watershed. One issue of contention is the desired health and appearance of cypress-tupelo trees in the watershed. Both IDNR and FWS recognize that current swamp water levels are producing stress on the trees. IDNR is managing the swamp for open, deep water habitat with the intent of imitating historical conditions (a mission for IDNR's Division of Natural Heritage and Cache River State Natural Area). For IDNR, a degree of tree stress is acceptable and a natural characteristic of pre-settlement conditions. On the other hand, FWS views tree stress as a problem, recognizing that static water affects tree health, regeneration potential, and primary productivity. They propose management to mimic drier conditions whereby the swamp is subject to seasonal variation, allowing dry periods between flood pulses instead of permanent, deeper water levels.

"*We're dealing with something like that in the Cache with the effects of water levels on cypress trees and what's the best way to maintain the cypress swamp and what state do we want to maintain a swamp in. Do you want to restore what was there historically or maybe there's a better use for that*

swamp now, something that's a little different than what was there a hundred or two hundred years ago, but because the landscape has changed or because things have disappeared other places and maybe pushing that in a little different direction would make more sense, and their arguments often times are on several sides of that issue. It can be difficult." (int. 27)

Several key informants noted that differences in partner strategies and goals in the management of the complex ecosystems in the watershed can lead to adverse transboundary impacts. For instance, IDNR's land is positioned upstream of the refuge and higher in the watershed, meaning that IDNR management decisions, especially water management, influence practices on the refuge.

"*You can't throw a rock in the Upper Cache [River] without something happening fifty miles downstream in the Lower Cache [River].*" (int. 5)

"*There's stuff [water structures and manipulating water levels] on the IDNR side that impacts refuge land, so yeah, that gets a little more challenging because those things are under one partner's control and they affect more than that one partner, so I'd say that does make it a little more challenging.*" (int. 16)

However, other respondents saw the differences in organizational cultures from a positive perspective, emphasizing the potential for more pragmatic solutions in spite of the philosophical differences.

"*There is disagreement about how to manage the water, but disagreement is okay as long as you can maintain the conversation.*" (int. 20)

"*When your common goals aren't fully aligned, that can be an issue, but sometimes it can also be a positive thing too, depending on how you look at it. I always think it's good. I like to be not only questioned by partners, but by constituents, and opponents [to] make sure we're always doing the right thing.*" (int. 17)

Similarly, several key informants expressed optimism that collective action towards the pursuit of common interests among CRJVP members could be triggered by the window of opportunity created by perceived social or ecological crisis.

"*Unfortunately, a lot of times, collaboration doesn't happen until [there is] some emergency. There's gotta be a real need, not just, well, someday we're going to do this, and then we keep doing what we do. A lot of times, real collaboration comes out of an immediate need, an emergency. It doesn't always have to be a natural disaster, but so many times, we rise to be our best after a natural disaster. That's when all the agency labels are put aside. Somebody sets up this command incident system ... and you are part of the team.*" (int. 13)

Science and decision-making criteria

Another theme emerging from key informant interviews was how differences in preferred scientific methods and decision-making styles influenced stakeholder positions and approaches to various CRJVP goals. The interviews revealed that partner organizations in the CRJVP differ in their positions on what constitutes adequate scientific evidence for informing program implementation. In some instances, agencies have differed in their interpretation of when baseline research should partially give way to actual implementation. Some respondents expressed frustration about the delays caused by failure to translate the science into program implementation.

"*We're kind of at the point where we're saturated with so much science on the ground that we really need to move with getting stuff done on the ground, ... I think there are things that I think we can move forward on now. We have these management plans and we have the science and we should be utilizing that to do more on the ground.*" (int. 22)

On the other hand, other respondents highlighted the inadequacy of current understanding about the ecosystem and the need for more rigorous scientific research before management interventions.

"*There's a lot of grey area out there, and even taking lessons learned in one spot or one site, and trying to transfer them to another site, all of a sudden, it's gray. It's not exactly the same situation, the critters are a little different, the landscape is a little different, the key players or the stakeholders are a little bit different, so there's a lot of grey out there and everyone isn't entitled to their own science, but everyone is entitled to their own opinion, and a lot of these things, when decisions*

are being based, and when you don't have the best science to base stuff and a lot is just professional opinion, then that can get kind of difficult, and it's just hard to get to an answer because you can't prove that one opinion is right or wrong or better than the other, so that can be difficult." (int. 27)

The goal of reconnecting the lower and upper sections of the Cache River is one area where the research versus implementation conundrum is best illustrated. Over the past thirty years, CRJVP partners have contributed resources towards modeling reconnection locations, studied biological and ecological effects of reconnection, analyzed potential impacts on private lands, addressed legal proceedings with adjacent private landowners, and examined flooding potential and long-term effects of hydrologic connectivity. IDNR and TNC matched funds to hire a restoration coordinator to oversee the process of gathering information from various stakeholders, including agencies, local community members, and researchers conducting studies to map reconnection effects. The Illinois State Water Survey also updates inventories and provides the latest on-the-ground observations. However, the reconnection still has not occurred. As a result, TNC has been losing interest in supporting a goal that it perceives may never be achieved.

"*We can't continue to pour resources and money into science when we're not implementing things and we're not using it and then actually doing stuff on the ground… Restoring the hydrology is huge and that is really one of the main reasons why The Nature Conservancy is here, so it's hard for us to really be involved when that is all at a standstill.*" (int. 22)

Slow progress towards the goal of reconnection, compounded by financial challenges, has caused TNC to shift focus towards the larger population centers where most donors live and to pursue management actions showing evident and quantifiable outcomes. As a result, TNC has been backing away from active participation in CRJVP and taking the position of a silent partner.

Despite the aforementioned challenges, several key informants were of the view that the perceived lack of progress may be due to a narrow definition of success, suggesting the need for the development of methods and comprehensive criteria for evaluating the impacts of the CRJVP.

"*We've been so focused on reconnection, even though we've been doing a lot of other things, it can kind of look like, well, you've just been focused on reconnection for twenty years and it hasn't happened. So it kind of looks like a failure in a way, or not much success, whereas I would argue, well, we've still been planting trees, we've still been buying land, we've still been doing outreach, we've been doing other things. It's just that we really don't talk about them very much. And reconnection may never happen. That just might be the reality… I always think that there's a danger on, if you're focused just on this great big holy grail that could take a long time, it's not a way to really encourage people being involved, because a lot of people just won't have the patience or whatever. So I think you need to think about what are the other things we're also doing that will be successful to move us forward.*" (int. 16)

Discussion

In recent decades, collaborative approaches to ecosystem management have received attention among resource management science and policy researchers in the U.S. and elsewhere. Yet the institutional dimensions of ecosystem management remain less understood. In this paper, we have examined how the institutional attributes of the CRJVP members influence prospects for cooperation or non-cooperation toward realization of landscape scale conservation goals. The results reveal that characteristic institutional differences, such as policy mandates, administrative structures and budget mechanisms, institutional cultures, and approaches to science and decision-making influence the motivation and capacity for collaboration among the CRJVP members. These key findings from the study are discussed below in the context of the relevant literature.

Policy mandates and regulatory ambiguities

Collaborative decision-making provides a framework for harmonizing the efforts of different types of institutions, including states, markets, and communities toward the realization of common goals at the bioregional level. However,

such processes are likely to be constrained by conflicting policy guidelines that shape the priorities and mandates of individual participants (Butler and Koontz 2005), as well as the lack of clear legal guidelines authorizing such regional level initiatives, often characterized by multiple jurisdictions and land ownership types (Cortner and Moote 1999). These issues can hamper partners' motivation and capacity for cooperation.

Analysis revealed areas of consensus and disagreement within the policy frameworks guiding the activities of each CRJVP partner. All members of the partnership agree on general goals regarding the protection, conservation, and restoration of land and water resources in the watershed. However, important differences exist across management actions each partner is mandated to implement, leading to non-cooperation on specific projects or initiatives. For instance, the FWS is unwilling to participate in some of IDNR water level management activities because the potential flooding of private lands violates the FWS mandate. Additionally, a lawsuit by the local Drainage District and private landowners seeking to halt the IDNR actions also highlights the challenges associated with the lack of clarity regarding the legal basis for these ecosystem management activities, especially when organizational policies impact private property.

Administrative and budget structures

In addition to the effect of the external regulatory environment discussed earlier, collaborative ecosystem management processes can be constrained by differences in the internal organizational structures and budget mechanisms of participating organizations (Imperial 1999). Most agencies have traditionally operated with a top-down decision-making approach that is inconsistent with the need for adaptable institutions in ecosystem management processes (Cortner and Moote 1999; Bonnell and Koontz 2007). Similarly, rigid budget structures targeting a narrow set of resource management priorities can constitute an impediment to the mobilization of resources for pursuing long term and holistic ecosystem management goals (Cortner et al. 1998; Stein and Gelburd 1998).

Consistent with these issues, analysis of the CRJVP revealed that partners differed in their organizational structures for budgeting and decision-making. Some partners, such as the FWS, granted autonomy for decision-making and implementation at lower levels of the organization. This autonomy included the flexibility to allocate funds and transfer them to appropriate projects as needed, enhancing the responsiveness of the FWS to emerging resource management challenges. In contrast, the IDNR operated according to a more rigid, top-down structure, constraining opportunities for flexible decision-making and funding allocation at lower levels of the organization. Funding availability also appeared to shape the project priorities of partner organizations, thus potentially diverting their focus from CRJVP goals. For instance, NRCS is strongly program-driven and requires that personnel concentrate their efforts on only those private tracts that fit within guidelines of programs currently funded. A weakened regional economy and dwindling capital resources constitute a double-edged sword further threatening the efficacy of the CRJVP. While the literature on collaboration suggests that limited budgets may encourage increased reliance on partnerships to meet long-term goals (Wondolleck and Yaffee 2000), the view expressed by key informants is that institutions tend to prioritize internal needs over common partnership goals when capacity levels diminish due to shrinking staffing and budgets. These issues impede the overall resourcefulness and adaptability of the CRJVP.

Institutional culture

One of the major constraints to the implementation of collaborative ecosystem management is the path-dependency effect caused by the norms that shaped past management practices of each involved organization (Yaffee 1996; Cortner et al. 1998). These differences in institutional cultures manifest themselves in the meanings that different groups associate with ecosystem management, ranging from anthropocentric to eco-centric interpretations (Endter-Wada et al. 1998; Yaffee 1999). In line with this diversity of institutional cultures, our analysis of the CRJVP revealed differences in the resource management philosophies and preferred scales of management intervention by the partners. This

presents various challenges as well as opportunities for cooperation.

The preferred scales of management intervention for CRJVP partners range from the level of the farm to the entire watershed. Conflicts sometimes erupt among the different interests being pursued at the various scales within the watershed. For instance, TNC defines their mission holistically with a focus on managing the entire watershed to create a sustainable, naturally flowing river system. TNC policies concerning hydrological processes and other issues are consistent with those governing the broader ecosystem management concerns of the CRJVP. On the other hand, the NRCS primarily focuses at the farm level to provide services to private landowners through conservation programs under their administration. NRCS staff has access to private landowners' properties through contractual conservation easements. This arrangement allows private landowners to retain land ownership rights while also receiving the benefits of conservation programs. However, the primarily agricultural focus of private land owners within the Cache River watershed poses a particular challenge to the agencies whose focus is on managing publicly owned lands near the Cache River channel and wetlands. These conflicts between agricultural and conservation interests threaten to compromise the trust between the NRCS and private landowners.

Members of the CRJVP also differ in their perspectives toward land management and these differences shape their variable goals and solutions to common problems in the watershed. For instance, both the IDNR and the FWS agree that current water levels in the cypress-tupelo swamps of the Cache River watershed have adverse effects on the health of the trees. However, the two agencies differ on the desired future condition of the ecosystem, as well as the strategies for achieving those goals. The IDNR appears to be guided by an ecocentric or biocentric philosophy that posits that the maintenance of ecological integrity is the primary goal of ecosystem management, and that human interventions in ecosystems are detrimental (Endter-Wada et al. 1998). As such, IDNR management proposals focus on returning the ecosystem to pre-settlement conditions that are assumed to be natural. On the other hand, the FWS appears to be guided by an anthropocentric philosophy that emphasizes the usefulness of ecosystems to meeting human needs, as well as the resilience of ecosystems to human use (Endter-Wada et al. 1998). As such, the agency's proposals aim at maintaining the ecological conditions needed to enhance the productivity of the ecosystem rather than returning it to some natural pre-settlement condition. The issue is compounded by the fact that upstream-downstream relationships within the watershed mean that actions taken by one agency have transboundary impacts on others.

Science and decision-making criteria

Effective ecosystem management requires holistic and integrated science-based decision making criteria (Cortner and Moote 1999). Much of the ecological research that informed conventional resource management was conducted at fine scales inadequate for understanding complex ecosystem processes at larger scales (Stein and Gelburd 1998). Given the shortcomings associated with the dominance of ecological experts in conventional resource management, multidisciplinary approaches that integrate knowledge from the social and ecological sciences are essential for meeting the knowledge requirements of the ecosystem management paradigm (Endter-Wada et al. 1998; Keogh and Blahna 2005). Adaptive management is also considered an important strategy for dealing with the knowledge inadequacies in ecosystem management, as it emphasizes the need for learning to deal with uncertainties in the resource management process through monitoring and experimentation (Allen et al. 2011). However, new knowledge may not always be integrated into subsequent decision-making processes as required (Allen and Gunderson 2011), and the methods and indicators for evaluating success may not be well-developed. These challenges at the science-policy interface were encountered in our analysis of the CRJVP.

The interviews revealed that CRJVP members differed on the question of what constitutes adequate science for informing management actions. The goal of hydrologic connectivity of the Lower and Upper Cache Rivers highlighted this tension among partners. Since the formation of the partnership in the early 1990s, significant research has been conducted by TNC, IDNR, and

other partners to gain a deeper understanding of the issue. However, little of the science has been translated into management actions. While some respondents expressed frustration over the lack of progress in project implementation, others favored a more cautious approach so as to clarify knowledge gaps and system uncertainties. Key informants also differed in terms of their criteria for judging the success of the CRJVP. While some saw the partnership as a failure due to the lack of progress toward realizing the goal of hydrologic connectivity, others suggested the need for broader evaluation criteria that included the other projects being implemented by the partnership agencies. In the absence of such criteria, the perceived lack of progress, combined with declining resources, has resulted in a loss of motivation by some partners, notably TNC, to actively contribute to the CRJVP efforts.

Conclusion

The emergence of the ecosystem management paradigm over the last few decades has heightened awareness and interest regarding the benefits of collaborative decision-making. Unlike conventional top-down institutional practices, collaboration offers opportunities for engaging diverse stakeholders in consensus-based decision-making processes. The effective implementation of broad-based collaborative processes could enhance the sustainability of ecosystems and human communities while contributing to principles of good governance, such as equity, legitimacy, and participation. While collaborative partnerships build upon synergistic relationships to pursue common interests, they often fall short of expectations by failing to foresee inherent limitations (Cortner et al. 1998; Imperial 1999).

In this paper, we employed a qualitative research approach in exploring how various institutional characteristics influence prospects for conflict and cooperation among members of the CRJVP in southern Illinois. Our results revealed that institutional attributes, such as differences in policy mandates and regulations, administrative and budget structures, institutional cultures, as well as approaches to science and decision-making interact to influence the motivation and capacity for actors to cooperate. Based on these results, we suggest that the successful management of the Cache River watershed calls for an adaptive governance approach to managing conflicts and dealing with uncertainties (Dietz et al. 2003; Folke et al. 2005; Akamani et al. 2016). Adaptive governance moves beyond public involvement in collaborative processes (Akamani and Wilson 2011) to explicitly address the need for dealing with uncertainties through adaptive management, as well as harnessing a diversity of knowledge systems and institutions across multiple scales to promote ecosystem management (Folke et al. 2005; Gunderson and Light 2006). In the Cache River watershed, such an adaptive governance approach will call for an expanded membership of the CRJVP to include local representatives, the explicit integration of social science and non-scientific knowledge, prioritization of adaptive management, and investing in the capacity to manage conflicts in stakeholder values and interests. We conclude that such flexible multi-level partnerships stand a better chance of achieving success if potential challenges are identified prior to entering agreements. If all parties openly discuss potentially conflicting land management policies and their own institutional limitations as early in the emerging partnership process as possible, greater resiliency may be attained and more realistic goals achieved.

Acknowledgements

This research was funded by the McIntire-Stennis Cooperative Forestry Research Program. The authors gratefully acknowledge E. Seekamp and J. Zaczek who provided helpful suggestions throughout the study and reviewed an earlier version of this manuscript.

Author Bio and Contact Information

Jennifer Behnken is a community forester for the Missouri Department of Conservation based in Cape Girardeau. She is involved in public and private forest management along with assisting communities and municipalities in southeast Missouri with their forestry needs. Her employment background includes environmental education and outreach in the Cache River watershed. She also worked as an urban and community forestry coordinator for Southern Illinois University, assisting communities throughout Illinois in urban forest management. She holds a B.S. and M.S. in

Forestry from Southern Illinois University. She can be reached at jenniferbehnken@gmail.com.

John Groninger (corresponding author) is a professor in the Department of Forestry at Southern Illinois University, a position he has held since 1997. His teaching, research, and outreach interests include silviculture, agroforestry, urban forestry, and watershed rehabilitation. Along with his students, Dr. Groninger has been conducting research in the Cache River watershed throughout his tenure at SIU, addressing bottomland hardwood forest management issues. He can be reached at groninge@siu.edu and at 1205 Lincoln Drive, Room 184, Carbondale, IL 62901.

Kofi Akamani is an Assistant Professor of Forest Recreation and Conservation Social Science in the Department of Forestry at Southern Illinois University. He received his Ph.D. in Natural Resources from the University of Idaho. His research involves the integration of concepts from social-ecological systems research and rural sociology among other fields with the aim of gaining a theoretical understanding of human-environment interactions across multiple scales and informing policies that promote human well-being and ecosystem health across the rural urban continuum. He is particularly interested in understanding and enhancing the resilience of forest-dependent communities and the adaptive governance of social-ecological systems. He can be reached at k.akamani@siu.edu.

References

Akamani, K., E.J. Holzmueller, and J.W. Groninger. 2016. Managing wicked environmental problems as complex social-ecological systems: The promise of adaptive governance. In: *Landscape Dynamics, Soils and Hydrological Processes in Varied Climates*, A.M. Melesse and W. Abtew (Eds.). Springer International Publishing, New York, pp. 741-762.

Akamani, K. and P.I. Wilson. 2011. Toward the adaptive governance of transboundary water resources. *Conservation Letters* 4(6): 409-416.

Allen, C.R., J.J. Fontaine, K.L. Pope, and A.S. Garmestani. 2011. Adaptive management for a turbulent future. *Journal of Environmental Management* 92(5): 1339-1345.

Allen, C.R. and L.H. Gunderson. 2011. Pathology and failure in the design and implementation of adaptive management. *Journal of Environmental Management* 92(5): 1379-1384.

Baxter, J. and J. Eyles. 1997. Evaluating qualitative research in social geography: Establishing 'rigor' in interview analysis. *Transactions of the Institute of British Geographers* 22(4): 505-525.

Berkes, F. 2010. Devolution of environment and resources governance: Trends and future. *Environmental Conservation* 37(4): 489-500.

Bonnell, J.E. and T.M. Koontz. 2007. Stumbling forward: The organizational challenges of building and sustaining collaborative watershed management. *Society & Natural Resources* 20(2): 153-167.

Burde, J., S. Kraft, and R. Becker. 1998. Partnerships in wetland restoration. *Parks & Recreation* 33(2): 34-47.

Butler, K.F. and T.M. Koontz. 2005. Theory into practice: Implementing ecosystem management objectives in the USDA Forest Service. *Environmental Management* 35(2): 138-150.

Carlsson, L. and F. Berkes. 2005. Co-management: Concepts and methodological implications. *Journal of Environmental Management* 75: 65-76.

Chapin, F.S., S.R. Carpenter, G.P. Kofinas, C. Folke, N. Abel, W.C. Clark, P. Olsson, D.M.S. Smith, B. Walker, O.R. Young, F. Berkes, R. Biggs, J.M. Grove, R.L. Naylor, E. Pinkerton, W. Steffen, and F.J. Swanson. 2010. Ecosystem stewardship: Sustainability strategies for a rapidly changing planet. *Trends in Ecology & Evolution* 25(4): 241-249.

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

Cinner, J.E., T.R. McClanahan, M.A. MacNeil, N.A. Graham, T.M. Daw, A. Mukminin, D.A. Feary, A.L. Rabearisoa, A. Wamukota, N. Jiddawi, S.J. Campbell, A.H. Baird, F.A. Januchowski-Hartely, S. Hamed, R. Lahari, T. Morove, and J. Kuange. 2012. Comanagement of coral reef social-ecological systems. *Proceedings of the National Academy of Sciences* 109(14): 5219-5222.

Cortner, H. and M.A. Moote. 1999. *The Politics of Ecosystem Management.* Island Press, Washington, D.C.

Cortner, H.J., M.G. Wallace, S. Burke, and M.A. Moote. 1998. Institutions matter: The need to address the institutional challenges of ecosystem management. *Landscape and Urban Planning* 40(1): 159-166.

Creswell, J.W. 2003. *Research Design: Qualitative, Quantitative, and Mixed Methods Approaches* (2nd Ed.). Sage Publications, Thousand Oaks, CA.

Creswell, J.W., W.E. Hanson, and V.L.P. Clark. 2007. Qualitative research designs: Selection and

implementation. *The Counseling Psychologist* 35(2): 236-264.

Daniels, S.E. and G.B. Walker. 1996. Collaborative learning: Improving public deliberation in ecosystem-based management. *Environmental Impact Assessment Review* 16: 71-102.

Dietz, T., E. Ostrom, and P.C. Stern. 2003. The struggle to govern the commons. *Science* 302(5652): 1907-1912.

Durum, L.A., J. Bathgate, and C. Ray. 2004. A local example of land-use change: Southern Illinois - 1807, 1938, and 1993. *The Professional Geographer* 56: 127-140.

Endter-Wada, J., D. Blahna, R. Krannich, and M. Brunson. 1998. A framework for understanding social science contributions to ecosystem management. *Ecological Applications* 8(3): 891-904.

Folke, C., T. Hahn, P. Olsson, and J. Norberg. 2005. Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30: 441-473.

Folke, C., A. Jansson, J. Rockström, P. Olsson, S.R. Carpenter, S.F. Chapin III, A.S. Crepin, G. Daily, K. Danell, J. Ebbesson, T. Elmqvist, V. Galaz, F. Moberg, M. Nilsson, H. Osterblom, E. Ostrom, A. Persson, G. Peterson, S. Polasky, W. Steffen, B. Walker, and F. Westley. 2011. Reconnecting to the biosphere. *Ambio* 40(7): 719-738.

Grumbine, R.E. 1994. What is ecosystem management? *Conservation Biology* 8(1): 27-38.

Gunderson, L. and S.S. Light. 2006. Adaptive management and adaptive governance in the everglades ecosystem. *Policy Sciences* 39(4): 323-334.

Haeuber, R. 1998. Ecosystem management and environmental policy in the United States: Open window or closed door? *Landscape and Urban Planning* 40: 221-233.

Imperial, M.T. 1999. Institutional analysis and ecosystem-based management: The institutional analysis and development framework. *Environmental Management* 24(4): 449-465.

Illinois Department of Natural Resources, 2006. The Cache River Wetlands Joint Venture: A watershed scale restoration project. Available at http://dnr.state.il.us/lands/landmgt/parks/r5/CCR/ccr_JointVentureAccomplishDoc.pdf. Accessed February 22, 2012.

Keough, H.L. and D. Blahna. 2005. Achieving integrative, collaborative ecosystem management. *Conservation Biology* 20(5): 1373-1382.

Lee, K.N. 2001. Appraising adaptive management. *Ecology and Society* 3(2): 3.

Margerum, R.D. and C.J. Robinson. 2015. Collaborative partnerships and the challenges for sustainable water management. *Current Opinion in Environmental Sustainability* 12: 53-58.

Olsson, P. and C. Folke. 2001. Local ecological knowledge and institutional dynamics for ecosystem management: A study of Lake Racken watershed, Sweden. *Ecosystems* 4(2): 85-104.

Opperman, J.J., G.E. Galloway, J. Fargione, J.F. Mount, B.D. Richter, and S. Secchi. 2009. Sustainable floodplains through large-scale reconnection to rivers. *Science* 326(5959): 1487-1488.

Patton, M.Q. 1990. *Qualitative Evaluation and Research Methods*. Sage Publications, Thousand Oaks, CA.

Plummer, R. and D. Armitage. 2007. A resilience-based framework for evaluating adaptive co-management: Linking ecology, economics and society in a complex world. *Ecological Economics* 61(1): 62-74.

Shenton, A.K. 2004. Strategies for ensuring trustworthiness in qualitative research projects. *Education for Information* 22: 63-75.

Steel, B.S. and E. Weber. 2001. Ecosystem management, decentralization, and public opinion. *Global Environmental Change* 11(2): 119-131.

Stein, S.M. and D. Gelburd. 1998. Healthy ecosystems and sustainable economies: The federal interagency ecosystem management initiative. *Landscape and Urban Planning* 40(1): 73-80.

Strauss, A. and J. Corbin. 1998. *Basics of Qualitative Research: Procedures and Techniques for Developing Grounded Theory*. Sage Publications, Thousand Oaks, CA.

Wondolleck, J.M. and S.L. Yaffee. 2000. *Making Collaboration Work: Lessons from Innovation in Natural Resource Management*. Island Press, Washington, D.C.

Yaffee, S.L. 1996. Ecosystem management in practice: The importance of human institutions. *Ecological Applications* 6(3): 724-727.

Yaffee, S.L. 1997. Why environmental policy nightmares recur. *Conservation Biology* 11(2): 328-337.

Yaffee, S.L. 1999. Three faces of ecosystem management. *Conservation Biology* 13(4): 713-725.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 34-45, August 2016

The Effects of Policy Implementation Uncertainty on Groundwater Extraction

Dacheng Bian[1], *Ryan B. Williams[1], Aaron Benson[1,2], and Eduardo Segarra[1]

[1]*Department of Agricultural and Applied Economics, Texas Tech University, Lubbock, TX*
[2]*Zions Bancorporation, Salt Lake City, UT*
**Corresponding Author*

Abstract: Concerns about the high rate of depletion of the Ogallala Aquifer in the Southern High Plains of Texas (Texas High Plains) in recent years have led to the enactment of policies designed to slow down water extraction and increase the usable life of the aquifer. However, policy implementation has not been uniform across the aquifer, leaving some farmers in portions of the aquifer with no effective groundwater extraction restrictions only a short distance away from areas where farmers face regulatory limits. This study investigates the effects of policy implementation uncertainty on the extraction of groundwater. An econometric model is estimated for harvested acreage of the three major crops produced in the five representative counties of the Texas High Plains. Corn acreage (an irrigation-intensive crop) is used as a proxy for groundwater use. After controlling for price and climate effects, it was concluded that there is strong evidence that groundwater use policy uncertainty is likely to increase groundwater extraction in the Texas High Plains region. The social desire to achieve perpetual sustainability of groundwater resources in the study region, or at the very least to prolong the life of the Ogallala Aquifer, has resulted in discussions about alternative groundwater management policies. Producers, in their concern about the implementation of policies to slow down groundwater extraction, increase their use of water in the short-run in order to maximize profits before groundwater use restriction policies are enforced.

Keywords: *policy uncertainty, groundwater extraction, irrigation, agricultural productivity*

Over the past fifty years, demand for water resources in American agriculture has increased significantly. Irrigated agriculture currently uses between 80% and 90% of all consumptive water activity taking place in the United States (U.S.) (USDA ERS 2015). According to the 2008 Farm and Ranch Irrigation Survey, 74 million acre-feet (24 trillion gallons) of water are used annually for agricultural irrigation in the western states, and 48% of the water used is withdrawn from regional aquifers. These figures have been on the rise over the past few years (Schaible and Aillery 2012). The growing demand for water resources in agriculture has placed increased pressure on existing water sources and has induced an increased interest in water management, water conservation related issues, and the future sustainability of irrigated agriculture. Policies aimed at reducing water use and increased water resources conservation efforts have been adopted, and various technologies enhancing water use efficiency that directly contribute to improved sustainability of the water resources have been developed.

Agricultural irrigation has played a significant role in the economic growth and development of the Great Plains Region of the U.S. The Ogallala Aquifer (Figure 1) is the major source of irrigation water in this area, which includes 174,000 square miles. It lies beneath portions of eight states of the U.S.: Colorado, New Mexico, Wyoming, Texas, Kansas, Oklahoma, South Dakota, and Nebraska (Alley et al. 1999). Every year, almost 65% of the overall water used for irrigation in the U.S. is extracted from the Ogallala Aquifer. Furthermore, approximately 82% of the 2.3 million people who live in the Great Plains region acquire their drinking water from the Ogallala

Aquifer (Maupin and Barber 2005).

Intensive agricultural irrigation threatens the usable life of the Ogallala Aquifer (Dennehy et al. 2002). Daily groundwater use for irrigation from the Ogallala Aquifer has increased from 480 million cubic feet of groundwater in the early 1950s to 2,150 million cubic feet in 1980 (Alley et al. 1999). The decline of the water table across the region ranges from one-half foot to several feet per year. The Ogallala Aquifer is an unconfined aquifer, but when the rate of recharge is compared to the rate of extraction, the former is negligible (Birkenfeld 2003). The regional economy of the Great Plains depends on agricultural irrigation, but many people consider that the decline of groundwater of the Ogallala Aquifer is likely to play a potentially dramatic and significant negative role in the future (Terrell et al. 2002).

The southernmost portion of the aquifer – that which underlies the Texas High Plains (THP) – has experienced faster declines in groundwater because of the relatively low rate of recharge compared to the northern portion of the aquifer. In the southern portion of the aquifer, most people believe that recharge rates range from one-half inch to several inches per surface acre annually (High Plains Underground Water District No.1 2004). The ratio of extraction to recharge ranges from approximately 50:1 to as low as 1:1 (These estimates were extrapolated from changes in saturated thickness of the aquifer. Saturated thickness data was acquired from the Texas Tech University Center for Geospatial Technology: http://www.gis.ttu.edu/center/Ogallala/OgallalaData.html). Another study made a water use projection for the Texas Water Development Board using the Groundwater Availability Model for the Southern High Plains of Texas. They estimated the annual amount of water used for irrigation in the southern portion of the aquifer to be approximately 3,800,000 acre-feet, which

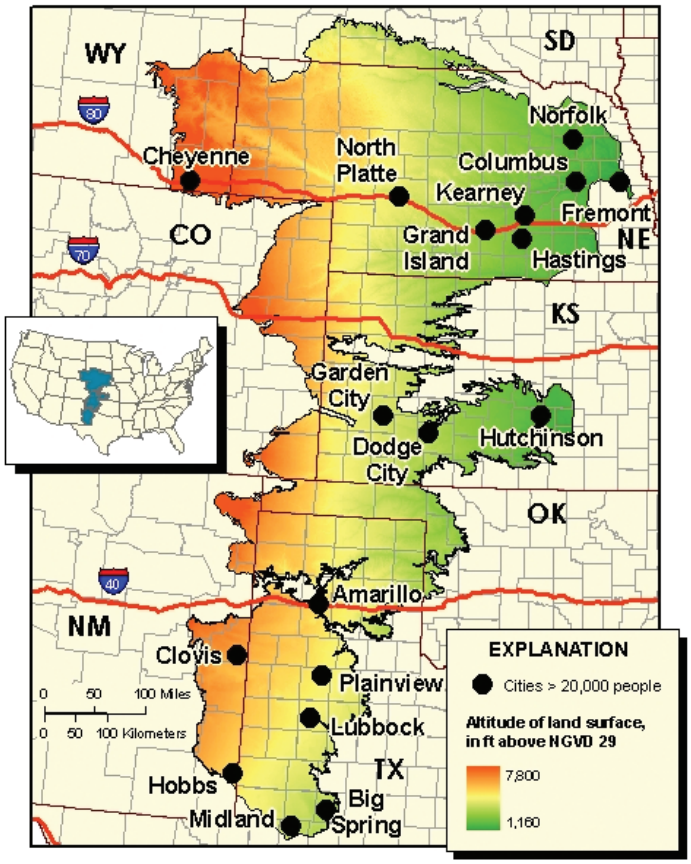


Figure 1. Location of the Ogallala Aquifer. Source: U.S. Geological Survey (USGS) http://ne.water.usgs.gov/ogw/hpwlms/.

is used to irrigate approximately 3,500,000 acres (Amosson et al. 2003). The main agricultural commodities produced in the southern portion of the Great Plains include wheat, grain sorghum, peanuts, corn, and cotton. This area accounts for a significant proportion of Texas agricultural crop production, including 26% of the production of grain sorghum, 46% of the production of wheat, 59% of the production of cotton, 40% of the production of peanut, and 10% of the production of corn (NASS 2006).

The future of agricultural production in the Southern High Plains of Texas, and the rural economies supported by it, depend in part on the future availability of groundwater. In order to slow down groundwater extraction and increase the usable life of the Ogallala Aquifer, several policies have been designed and implemented. However, policies have not been uniform across the aquifer, and in some counties local groundwater district's rules have not been enforced. To this end, public policy has been established which clarifies natural resource property rights and, in some cases, limits the private landowners' rights. In portions of the policy entitled "Rules of the High Plains Underground Water Conservation District No. 1" limitations of groundwater use for each year after 2012 are addressed. Specifically, producers are limited to apply up to 18 acre-inches per year in years 2014-2015 and up to 15 acre-inches beginning in 2016. However, these limits have yet to be enforced and these particular limits are not binding for most producers.

This study evaluates how policy implementation uncertainty (which means the policies are existing, expected to become increasingly binding, but not currently enforced) can impact groundwater extraction in the Texas High Plain area. Groundwater use policy implementation has not been uniform across the aquifer in the Texas High Plains, leaving some farmers in portions of the aquifer with no effective groundwater extraction policy only a short distance away from areas where farmers face regulatory limits on groundwater extraction. Farmers make decisions regarding what to produce to maximize their own profits. The market prices of the five crops and their relative irrigation requirements are the important factors that affect farmers' decisions. Moreover, the availability of groundwater in the Southern Ogallala Aquifer directly impacts the agricultural sector of the Texas High Plains. The optimal crop choice for each producer is constrained by this physical availability, and potentially further constrained by groundwater policy.

The specific objective of this study is to evaluate if uncertain or impending policies regarding irrigation water restrictions may actually cause farmers to withdraw more groundwater than in the absence of those expectations.

Literature Review

Irrigated farming activities in the Texas High Plains have led to a significant decline in the groundwater resources of the Ogallala Aquifer over the past 60 years. Decreased saturated thickness and increased depth to groundwater, combined with rising energy prices, has significantly increased farmers' pumping costs and decreased profits in recent years.

Surface water resources are nonexistent for the purpose of irrigated crop production in the Texas High Plains, therefore the Ogallala Aquifer is vital for irrigated agriculture to continue in the region. However, the capacity of the Ogallala Aquifer is limited, so, in order to preserve the economic life of the aquifer and maintain the viability of the regional economy which significantly depends on agriculture, water conservation policies need to be designed and implemented. The U.S. federal government encouraged farmers to develop irrigated agriculture as a way to increase revenues in the past, but the availability of the groundwater resources in the west of the U.S. has significantly declined in recent years. Therefore, government entities at various levels of aggregation have started to design policies with the purpose of conserving water (Wheeler 2008). For example, on September 1, 2005 the 79th Texas Legislature passed House Bill 1763, which requires, among other issues, that groundwater management areas around the state establish upper limits for groundwater extraction locally (Mace et al. 2006). It is anticipated that in the long-term, these factors will play an important role in the economic viability of agricultural irrigation in the

Texas High Plains (Taylor et al. 2007).

Agricultural producers in Texas, who utilize groundwater as the source of water for their irrigated operations, follow what is commonly known as the "right of capture" law (Terrell 1998). This law grants full property rights to landowners to use the groundwater extracted from beneath their land to their benefit (Kaiser and Skillern 2001). Falling groundwater tables – resulting in increased pumping lifts – combined with high energy prices in the late 1970s gradually caused reduction in groundwater yields, overall pumping, and consequently total area irrigated. In 2002, there were 3.5 million Texas High Plains' crop acres irrigated; this figure was about 55% lower than the number of acres irrigated in the late 1960s (Texas Senate Bill 2, Austin, TX, 2002).

In the late 1990s and early 2000s, the Texas legislation explicitly recognized the increasing scarcity of groundwater supplies, particularly Senate Bill 1 (Texas Senate Bill 1, Austin, TX, 1997) and Senate Bill 2 (Texas Senate Bill 2, Austin, TX, 2002). Senate Bill 1 (SB 1) modified several portions of Texas water law and modified the structure of the state water management modus operandi. It required the Texas Water Development Board (TWDB) to develop a comprehensive statewide water use plan that incorporated locally developed regional water plans (Article 1, Section 1.01). Senate Bill 2 (SB 2) established a Texas Water Advisory Committee, as well as a guide to improve surface water and groundwater management planning at the local, regional, and state levels (Texas Joint Committee on Water Resources 2002). SB 2 also increased the power of groundwater management districts over area within their jurisdiction, allowing them to carry out production charges to limit groundwater exploitation. This law explicitly stated that these costs cannot be more than one U.S. dollar per acre-foot of water extracted.

High Plains Underground Water Conservation District #1 Rules (HPWD1R) indicate that a "50/50 Management Goal" which means that 50% of the saturated thickness of the Ogallala Aquifer will need to be in the Ogallala Aquifer 50 years later would be desirable as a goal. The first planning interval of the 50/50 Management Goal started on January 1, 2010 and will end on January 1, 2060. Moreover, since January 1, 2012 all persons who own or operate a well or well system that withdraws groundwater from the Ogallala Aquifer are required to limit the total amount of water extracted from the well or well system. However, the High Plains Underground Water Conservation District #1 (HPUWCD1) extended moratoriums for the limit through 2014. Specifically, during their November 12, 2013 regular meeting, the HPUWCD1 Board of Directors voted 5-0 to extend current water extraction limit moratoriums in the district's rules through calendar year 2014 (Musico 2013).

Conceptual Framework

Following Beattie et al. (1985), this study assumes that a representative farmer in a given county will receive profits by producing different crops. In this study, the choice is limited to three crops – corn, cotton, and sorghum – because peanut and wheat production is not prevalent in the sub-area of the Texas High Plains that was the focus of our research. This can be shown by defining a profit function as:

$$\prod_j = TR_j - TC_j = \sum p_i * Q_{ij} - \sum r_k * x_{kj} - FC_j \quad (1)$$

where, $\prod_j$ represents profit in county j, TR_j represents total revenue in county j, TC_j represents total cost of production in county j, p_i represents price of crop i, Q_{ij} represents production of crop i in county j, r_k represents the price of input k, x_{kj} represents the quantity of input k used in county j, and FC_j is the fixed cost of production in county j. A supply function can be defined as:

$$Q_{ij} = A_{ij} * Y_i \quad (2)$$

where, A_{ij} is the number of acres planted of crop i in county j, and Y_i is the yield of crop i. Substitution of equation (2) into (1), results in an indirect profit function defined as:

$$\prod_j = \sum p_i * A_{ij} * Y_i - \sum r_k * x_{kj} - FC_j \quad (3)$$

It is assumed that farmers make their decisions regarding what to plant depending on several factors. These include time trend, price of crops, rainfall, and availability of groundwater. That is,

$$A_{ij} = f(t, p_i, r_k, R, G) \quad (4)$$

where, t represents time trend, p_i represents price

of crop i, R represents rainfall, and G represents availability of groundwater. Substituting (4) into (3) gives:

$$\prod_j = \sum p_i * Y_i * f(t, p_i, r_k, R, G) - \sum r_k * x_{kj} - FC_j \quad (5)$$

or more compactly,

$$\prod_j = \prod_j (t, p_i, r_k, R, G). \quad (6)$$

Given that the availability of groundwater for irrigation depends on the stock of water in the Ogallala Aquifer in the county and the existence of groundwater use policies, if it is assumed that the stock of water in the aquifer is sufficient enough for irrigation purposes then only the existence of groundwater use policies, or restrictions on groundwater withdrawals, influences the availability of groundwater use for irrigation. So, the indirect profit function can be rewritten as:

$$\prod_j = \prod_j (t, p_i, r_k, R, pol) \quad (7)$$

where, *pol* represents the existence of policies impacting groundwater use in the county. Beattie et al. (1985) also indicate that, following Hotelling's lemma, the partial derivative of the indirect profit function with respect to crop prices is equal to the supply equation:

$$\frac{\partial \prod_j}{\partial p_i} = Q_{ij.} \quad (8)$$

From equation (2) and (4), if Y_i is constant, then the supply function can be defined as:

$$Q_{ij} = f(t, p_i, r_k, R, pol) \quad (9)$$

where, $Q_{ij} = Y_i * A_{ij}$. Assuming that the cross price effects are equal across crops, and given that in this particular case the three crops analyzed will be cotton, corn, and sorghum, the following restrictions were added:

$$\frac{\partial Q_{corn}}{\partial p_{cotton}} = \frac{\partial Q_{cotton}}{\partial p_{corn}} \quad (10)$$

$$\frac{\partial Q_{corn}}{\partial p_{sorghum}} = \frac{\partial Q_{sorghum}}{\partial p_{corn}} \quad (11)$$

$$\frac{\partial Q_{sorghum}}{\partial p_{cotton}} = \frac{\partial Q_{cotton}}{\partial p_{sorghum}} \quad (12)$$

That is, the effect of cotton price on corn production is equal to the effect of corn price on cotton production; the effect of sorghum price on corn production is equal to the effect of corn price on sorghum production; and the effect of cotton price on sorghum production is equal to the effect of sorghum price on cotton production.

Methods and Procedures

As highlighted earlier, the purpose of this study is to estimate an econometric model to analyze how the uncertainty associated with the implementation of groundwater use polices can affect farmers' agricultural production decisions in the short run. Specifically, this study seeks to determine if the uncertain implementation or impending policy enforcement regarding irrigation water restrictions may actually induce farmers to withdraw more groundwater than in the absence of those expectations. This study assumes that farmers want to maximize their profits over time, so they have to make their decisions regarding input use with future input availability in mind. It follows that, if farmers anticipate a restriction in input use (groundwater in this case) in the future, they may intensify the use of that input in the current period. The yield response to applied water is positive and decreasing (∂yield / ∂irrigation > 0, ∂^2yield / ∂water^2 < 0) over the range of profit-maximizing water use, so farmers' input use decisions are revealed by the production of the crops. In other words; the level of output, the intensity of input use, and the crops produced reflect the inter-temporal profit-maximizing decisions farmers have made. The econometric model estimated here contains crop acreage as a function of a time trend variable (year), a yearly adjusted crop price variable, a yearly rainfall variable, and the existence of the policy regarding water use limitation variable (defined as a dummy variable). Corn, cotton, and sorghum have been the main crops produced in the research area in the past few decades. Corn is the crop that requires the most water for growth when compared with the other two crops. It is assumed that if increased amounts of water are used for irrigation this would lead to an increase in crop production. Conversely, if crop production changes are experienced, so this would directly reflect water use changes for irrigation, ceteris paribus.

Data were collected for 20 counties that are located within both the High Plains Underground Water Conservation District #1 and the Northern High Plains Underground Water Conservation District, both of which are located within the area comprising the Ogallala Aquifer in the Texas High Plains. As shown in Figure 2, the light blue area is the Northern High Plains Groundwater Conservation District, the orange area is the Panhandle Groundwater Conservation District, the green area is the Hemphill County Underground Water Conservation District, and the purple area is the High Plains Underground Water Conservation District #1. The majority of the data used was obtained from the National Agricultural Statistics Service (NASS 2006). However, data availability required the study to be restricted to only five counties within the High Plains Underground Water Conservation District #1. These five countries included Deaf Smith, Parmer, Castro, Hale, and Floyd (shown in Figure 1 and Figure 2). Through direct observation of the data gathered, these five counties produce corn, cotton, and sorghum. The corn acreage within these five counties comprises the majority of the corn acreage grown in the entire Texas High Plains area. Table 1 presents the average acres of each crop planted, by county, over the study period of 1991 through 2013.

The current High Plains Underground Water Conservation District #1's groundwater use restrictions have never been enforced, but irrigators have the expectation that enforcement of this policy is quite likely in the future. It is hypothesized that producers who expect to face future production losses and lower revenues due to the enforcement of water limiting use policies will use as much

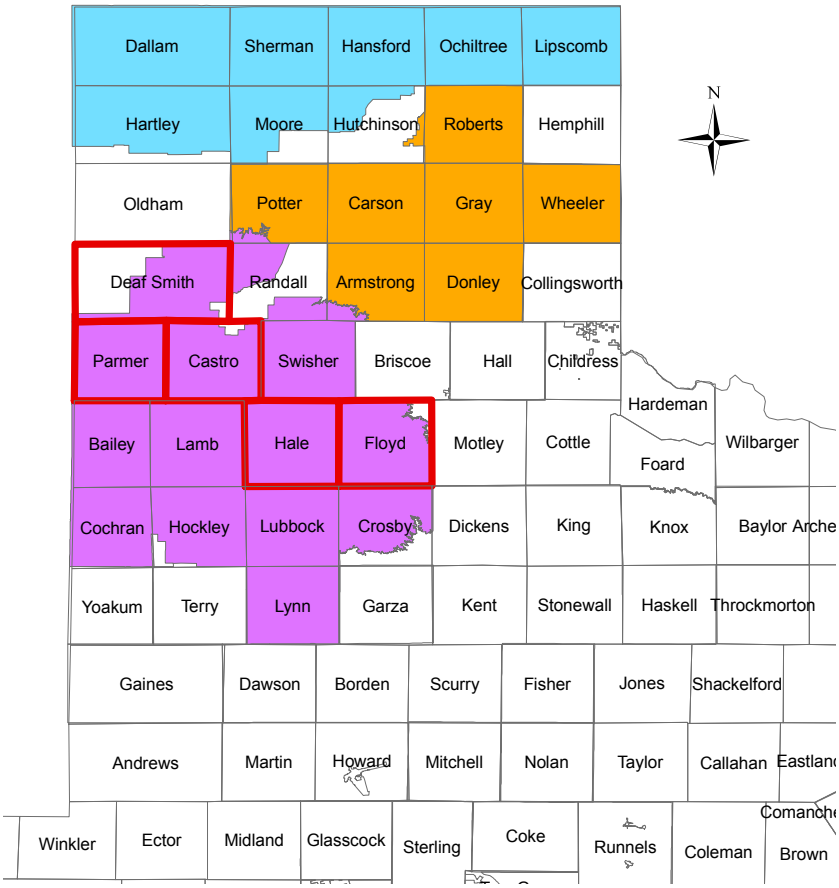


Figure 2. Location of Texas counties analyzed (outlined on map). Analyzed counties (Deaf Smith, Parmer, Castro, Hale, and Floyd) are within the High Plains Underground Water Conservation District #1. Source: Texas Water Development Board http://www.twdb.texas.gov/mapping/gisdata.asp.

Table 1. Average acreage of crops planted for each county.

County	Corn Acres Planted	Cotton Acres Planted	Irrigated Cotton Acres Planted	Non-irrigated Cotton Acres Planted	Sorghum Acres Planted	Irrigated Sorghum Acres Planted	Non-irrigated Sorghum Acres Planted
Castro	102,234	62,956	60,820	2,136	27,686	17,278	10,407
Deaf Smith	37,991	25,960	18,426	7,534	77,913	31,880	46,033
Parmer	75,652	66,704	62,747	3,956	44,108	23,340	20,767
Hale	49,743	243,960	213,260	30,700	45,900	33,761	12,138
Floyd	14,052	184,617	126,495	58,121	41,818	24,910	16,908
Swisher	19,247	82,078	63,891	18,187	38,861	20,042	18,819
Total	298,919	666,275	545,639	120,634	276,286	151,211	125,072

water as they can now to increase current profits. In order to test this hypothesis, this study uses a representative crop production regression model for the five counties (modeling the five counties as a profit-maximizing producer), and this model is estimated using seemingly unrelated regression (SUR) methods to analyze how the existence of the policy, crop prices, and climate affect crop production. Given that corn, cotton, and sorghum are the three main crops produced in the five counties analyzed, this study uses acres harvested for each crop as its dependent variable reflecting the level of crop production. Additional crops, as well as additional inputs to the production process, could be included in the analysis using an index approach, but such analysis is beyond the scope of this study. Three separate linear supply equations (which represent the first derivatives of an unspecified normalized quadratic profit function) for each crop in the five counties were estimated.

As highlighted above, SUR procedures were followed in the estimation of the regression model. The SUR system for the five counties includes three individual relationships that are linked by the fact that their disturbance terms are correlated (price of corn, cotton, and sorghum). The correlations among the disturbance terms could come from several sources, including crop prices. Thus, in this case, there is a need to include constraints linking restrictions on the parameters across the different equations for given counties. On the other hand, equations explaining some phenomenon in the counties selected provide a natural application as these various factors are likely to be subject to spillovers from farm level or county level shocks.

There are two main motivations for the use of SUR in this study. First, this study proposes to gain efficiency in estimation by combining information on different equations. The second motivation is to impose restrictions that involve parameters in different equations. SUR provides an appropriate model and can clearly demonstrate the relationship of each observation in this study.

The functions specified for each of the five counties included in the estimation are:

$$\textit{Corn Acres Harvested} = f(\textit{year, rainfall, policy, corn adjusted price, cotton adjusted price, sorghum adjusted price}) \quad (13)$$

$$\textit{Cotton Acres Harvested} = f(\textit{year, rainfall, policy, corn adjusted price, cotton adjusted price, sorghum adjusted price}) \quad (14)$$

$$\textit{Sorghum Acres Harvested} = f(\textit{year, rainfall, policy, corn adjusted price, cotton adjusted price, sorghum adjusted price}) \quad (15)$$

The adjusted commodity prices are calculated by dividing the specific commodity price received in one year divided by the prices received index for the prior year. The prices received index is an index of all prices received for crops and livestock, with the base being 1990 - 1992 = 100.

In order to protect the stock of groundwater in the Ogallala Aquifer, policies have been crafted to limit its use. In this case, for the High Plains Underground Water Conservation District #1, the existence of the groundwater use limitation policy restriction was specified as a dummy variable: 0

reflecting no policy for groundwater use is in place, and a 1 reflecting the existence of the policy (years 2010 through 2013), but unenforced, limiting groundwater use. The model to be estimated will support the hypothesis in the affirmative if the estimated parameter associated with the policy variable in the corn supply equation is positive and significant; where a positive and significant policy variable would indicate that corn acreage would be expected to increase due to the introduction of the unenforced policy. This would support the hypothesis as stated because corn is the most water-intensive crop among all crops considered, and an increase in corn acreage would be directly associated with an increase in total groundwater use/extracted from the aquifer.

Results

The result of the SUR analysis for Hale County is presented in Table 2. Data for the analysis encompasses the years of 1991 through 2013. The policy dummy variable takes a value of 1 for the years 2010 through 2013. The estimated parameter on the policy variable in the corn supply equation is positive and statistically significant. This result supports the hypothesis that when the unenforced policy is in place corn acreage and, consequently, groundwater usage increase. Similar results are found for each of the counties in the study area. The parameter estimates for the *policy* variable from each of the county level crop production models, along with their associated p-values, are presented in Table 3. In addition, the table provides the average acres of corn harvested for the period 1991 through 2009, prior to the implementation of the groundwater policy.

The results presented in Table 3 are both statistically significant and significant in their magnitudes. The change in the acres of corn production, by county, represent a meaningful increase relative to the average corn production prior to the policy implementation. The optimal levels of groundwater application for the irrigation of corn, cotton, and sorghum are assumed to be 21.80 inches, 9.86 inches, and 8.73 inches per acre, respectively (Turner et al. 2011). Therefore, if producers switch acres from cotton or sorghum to corn, the resulting change in water withdrawals would be expected to be approximately ten inches per acre. A ten acre-inch increase in irrigation for each of those additional acres of corn production contributes significantly to the annual drawdown of the Ogallala Aquifer.

Table 4 provides the results of a SUR model using the aggregate data from the five county study

Table 2. Regression parameters for crop production models, Hale County, Texas (1991 – 2013).

	Constant	Year	Rainfall	Policy	Adjusted Corn Price Received	Adjusted Cotton Price Received	Adjusted Sorghum Price Received
Corn Acres Harvested	7,805,467 (0.000)*	-3,922 (0.000)	1,130 (0.051)	20,416 (0.035)	89,596 (0.000)	-70,429 (0.022)	-31,864 (0.037)
Cotton Acres Harvested	-8,691,431 (0.030)	4,529 (0.024)	1,047 (0.575)	-31,483 (0.331)	-70,429 (0.022)	230,319 (0.003)	-30,935 (0.197)
Sorghum Acres Harvested	378,279 (0.857)	-1,850 (0.859)	-1,072 (0.334)	-10,601 (0.593)	-31,864 (0.037)	-30,935 (0.197)	38,558 (0.003)

* Probability value associated with parameter.

Restriction:

(1) [corn acres harvested] adjusted sorghum price received - [sorghum acres harvested] adjusted corn price received = 0

(2) [corn acres harvested] adjusted cotton price received - [cotton acres harvested] adjusted corn price received = 0

(3) [cotton acres harvested] adjusted sorghum price received - [sorghum acres harvested] adjusted cotton price received = 0

Table 3. Policy parameter estimate for corn acres harvested in county-level SUR regressions with associated p-values and historic corn acres harvested.

County	Policy Parameter Estimate	P-value	Average Corn Acres Harvested 1991-2009	Average Corn Acres Harvested 2010-2013
Castro	15,666	0.140	86,568	97,275
Deaf Smith	14,063	0.017	29,321	29,675
Parmer	21,460	0.069	62,578	50,950
Hale	20,416	0.035	45,910	38,275
Floyd	7,736	0.031	11,426	11,988

area, and we provide a matrix of error correlations for the model in Table 5. Here, the policy variable is not statistically significant in the corn supply equation. However, it is statistically significant in the sorghum supply equation where the parameter estimate is negative. Sorghum has the lowest crop water requirement of the three in the model, so acreage leaving sorghum production to either corn or cotton reflects an increase in groundwater usage. Heterogeneity across acres in the study area influences to which of the two crops an individual producer will switch. The sorghum variable is the only one that is statistically significant because, in the aggregate, some acreage moves to corn and some moves to cotton (not all producers have access to sufficient water to make the transition to corn). The specific transitions are largely a function of geographic location, so when averaged across the study area we find that the estimated changes in corn acres and cotton acres are not statistically significant.

These results again support the hypothesis stated above, that the uncertainty associated with the implementation of groundwater use policy indeed would affect farmers' agricultural production decisions. Specifically, uncertain or impending adoption of policies restricting groundwater use for irrigation is found to cause farmers to withdraw more groundwater than in the absence of those potential restrictions.

By using the optimal irrigation rates from Turner et al. (2011), and adjusting average groundwater use for cotton and sorghum production to account for both irrigated and non-irrigated cotton and sorghum acreages used in this study, we are able to estimate the groundwater extraction changes resulting from the policy implementation. These calculations are made as follows:

For corn:

$$\textit{Groundwater Use Change for Corn} = app_{cor} * \textit{policy of corn} \quad (16)$$

where, app_{cor} represents groundwater application for the irrigation of corn.

For cotton:

$$\textit{Adjusted Average Groundwater Use for Cotton} = \textit{irrigated cotton acres planted} * app_{cot} / \textit{cotton acres planted} \quad (17)$$

$$\textit{Groundwater Use Change for Cotton} = (17) * \textit{policy of cotton} \quad (18)$$

where, app_{cot} represents groundwater application for the irrigation of cotton.

For sorghum:

$$\textit{Adjusted Average Groundwater Use for Sorghum} = \textit{irrigated sorghum acres planted} * app_{sor} / \textit{sorghum acres planted} \quad (19)$$

$$\textit{Groundwater Use Change for Sorghum} = (19) * \textit{policy of sorghum} \quad (20)$$

where, app_{sor} represents groundwater application for the irrigation of sorghum.

By performing the calculations highlighted above, the following result is obtained. The predicted increase in groundwater use for irrigation across the three crops over the entire area is a total of 1,041,440 acre-inches. Given that the actual amount of groundwater use without the policy in place is 11,991,990 acre-inches, the existence of the policy – without enforcement – results in an estimated increase of 8.68% on groundwater use in the five counties.

Table 4. Regression parameters for crop production models for the research area.

	Constant	Year	Rainfall	Policy	Adjusted Corn Price Received	Adjusted Cotton Price Received	Adjusted Sorghum Price Received
Corn Acres Harvested	2.57e+07 (0.000)*	-12,915 (0.000)	6,855 (0.006)	51,084 (0.264)	195,874 (0.001)	-85,099 (0.153)	-81,254 (0.019)
Cotton Acres Harvested	-1.89e+07 (0.053)	9,940 (0.042)	1,367 (0.772)	67,746 (0.421)	-85,099 (0.153)	248,457 (0.034)	-52,870 (0.173)
Sorghum Acres Harvested	6,732,761 (0.214)	-3,355 (0.216)	781 (0.799)	-129,246 (0.016)	-81,254 (0.019)	-52,870 (0.173)	83,266 (0.001)

* Probability value associated with parameter.

Restriction:

(1) [corn acres harvested] adjusted sorghum price received - [sorghum acres harvested] adjusted corn price received = 0

(2) [corn acres harvested] adjusted cotton price received - [cotton acres harvested] adjusted corn price received = 0

(3) [cotton acres harvested] adjusted sorghum price received - [sorghum acres harvested] adjusted cotton price received = 0

Table 5. Correlation matrix of residuals.

	Corn Acres Harvested	Cotton Acres Harvested	Sorghum Acres Harvested
Corn Acres Harvested	1.00	--	--
Cotton Acres Harvested	-0.108	1.00	--
Sorghum Acres Harvested	-0.1985	-0.6708	1.00

Breusch-Pagan test of independence: Pr = 0.0092.

Conclusion

This study determines if uncertain or impending policy enforcement regarding irrigation water restrictions may actually induce farmers to withdraw more groundwater than in the absence of those expectations. This study (1) constructs a crop production yield econometric model to determine the relationship among selected factors, and analyzes how several independent variables affect the dependent variables included (crop acreages), and (2) tests if and how existing policies of groundwater use in the Ogallala Aquifer affect irrigation decisions and groundwater extraction levels.

The results of this study show that when the groundwater restriction policy is in place, farmers are likely to produce more corn than the other two major crops considered. This implies that the number of corn acres and cotton acres are likely to increase under the presence of the water use policy restriction, and grain sorghum acres would be expected to decrease. Groundwater use would be expected to increase as a result because corn requires more water than cotton for growth, and cotton requires more water than sorghum for growth. Based on these results, uncertainty in groundwater use restriction policy or imminent enforcement of such policy was found to indeed affect farmers' agricultural production decisions. That is, uncertain or impending policy implementation enforcement regarding irrigation groundwater restrictions is found to actually induce agricultural producers to withdraw more groundwater than in the absence of those restrictions. The enactment of specific policies, when not enforced, may lead to unintended consequences. In this particular case, a policy intended to curb groundwater withdrawals

from the Ogallala Aquifer is actually associated with the acceleration of groundwater withdrawals.

Specific conclusions for the Texas High Plains from this study include: (1) uncertainty regarding the enforcement of groundwater use limits can lead (and indeed *has* led in recent years) to exactly the opposite result, increased groundwater use, initially intended by the policy in the first place, and (2) decision makers must be aware that when evaluating the potential impacts of groundwater use policies, the evaluation and estimation of the impacts using aggregate (region-wide) versus disaggregate (county-wide) estimates can lead to overestimation or underestimation of total groundwater use.

The main limitation of this study is that the results obtained here are deemed to be applicable only in the "short run" because no alternative crops were considered in the analysis. Future research could explore the impact on crop choice induced by the short run increase in water usage. Additionally, this study focused on the entire research area (the five counties) and the general directional response. This does not provide policy makers with an understanding of why there were differential responses, in terms of magnitude, across the counties. Potential future work would identify the county level characteristics, or perhaps the individual producer characteristics, that are correlated with increased water use in the face of an unenforced groundwater extraction quota.

Author Bio and Contact Information

Dacheng Bian was born in Lanzhou, China in 1985. He earned his B.A. in Land Resources Management from Hubei University of Science and Technology, Xianning, Hubei, in 2008, a M.S. in Agricultural and Applied Economics from Texas Tech University, Lubbock, TX, in 2012, and his Ph.D. in Agricultural and Applied Economics from Texas Tech University, Lubbock, TX, in 2015. He is currently a Postdoctoral Research Associate in the Department of Agricultural and Applied Economics at Texas Tech University. His research area focuses on natural resource economics, ecosystem evaluation, and policy analysis.

Ryan Blake Williams (corresponding author) was born in Eureka, CA in 1976. He earned his B.A. in Economics from Emory University, Atlanta, GA, in 1999, a M.E. in Economics from North Carolina State University, Raleigh, NC, in 2005, and his Ph.D. in Economics from Texas Tech University, Lubbock, TX, in 2009. His current appointments are Assistant Professor in the Department of Agricultural and Applied Economics at Texas Tech University and Assistant Professor at Texas A&M AgriLife Research and Extension Center at Lubbock. He also serves as Director of the Texas Tech University Food and Resource Experimental Economics Laboratory (FREEL). He may be contacted at ryan.b.williams@ttu.edu and at Department of Agricultural and Applied Economics, Texas Tech University, MS 2132, Lubbock, TX, 79409.

Aaron Benson was born in Salt Lake City, UT in 1979. He earned a B.S. in Mathematics from Brigham Young University in 2003 and a Ph.D. in Economics from Washington State University in 2007. He spent 7 years as a faculty member of the Texas Tech University Department of Agricultural and Applied Economics, and is currently a Corporate Risk Officer at Zions Bancorporation, in Salt Lake City, UT.

Eduardo Segarra was born in Monterrey, Mexico in 1958. He earned his B.A. in Economics from the Universidad Autonoma de Nuevo Leon in 1979, a M.S. in Agricultural Economics from the University of Missouri-Columbia in 1982, and a Ph.D. in Agricultural and Applied Economics from Virginia Tech in 1986. He joined the faculty of Texas Tech University, Department of Agricultural and Applied Economics, in 1987, where he serves as Professor. He served as Chairman of the department from 2005 to 2013.

References

Alley, W.M., T.E. Reilly, and O.L. Franke. 1999. Sustainability of ground-water resources. *U.S. Geological Survey Circular 1186.* Available at http://pubs.usgs.gov/circ/circ1186/. Accessed June 21, 2016.

Amosson, S., T. Marek, L. New, F. Bretz, and L. Almas. 2003. Estimated irrigation demand for the southern Ogallala GAM. Available at https://www.twdb.texas.gov/publications/reports/contracted_reports/doc/2001483379_AppendixB.pdf. Accessed June 21, 2016.

Beattie, B.T., C.T. Taylor, and M.J. Watts. 2009. *The Economics of Production*, 2nd ed. Krieger Publishing Company, Malabar, FL.

Birkenfeld, D. 2003. Reclaiming water as a commons. *Southwest Chronicle* 4: 1-4.

Dennehy, K.F., D.W. Litke, and P.B. McMahon. 2002. The High Plains Aquifer, USA: Groundwater development and sustainability. *Geological*

Society, *London, Special Publications* 193: 99-119.

High Plains Underground Water District No.1. 2004. The Ogallala Aquifer. Available at http://www.hpwd.org/. Accessed June 21, 2016.

Kaiser, R.A. and F.F. Skillern. 2001. Deep trouble: Options for managing the hidden threats of aquifer depletion in Texas. *Texas Tech Law Review* 32: 249-304.

Mace, R.E., R. Petrossian, R. Bradley, and W.F. III Mullican. 2006. A streetcar named desired conditions: The new groundwater availability for Texas. In State Bar of Texas 7th Annual the Changing Face of Water Rights in Texas, 18-19 May, San Antonio, TX.

Maupin, M.A. and N.L. Barber. 2005. Estimated withdrawals from principal aquifers in the United States, 2000. *U.S. Geological Survey Circular 1279*.

Musico, J. 2013. Water district board extends moratorium for penalties on certain water use. *Lubbock Avalanche-Journal Lubbock Online,* Lubbock, TX.

National Agricultural Statistics Service. 2006. Quick Stats: Agricultural Statistics Data Base. Available at http://www.nass.usda.gov/Quick_Stats. Accessed June 21, 2016.

Schaible, G.D. and M.P. Aillery. 2012. Water conservation in irrigated agriculture: Trends and challenges in the face of emerging demands. EIB-99, U.S. Department of Agriculture, Economic Research Service.

Taylor, R.H., L.K. Almas, and W.A. Colette. 2007. Economic analysis of water conservation policies in the Texas Panhandle. Selected paper prepared for Southern Agricultural Economics Association Annual Meeting, Mobile, AL, 4-7 February.

Terrell, B.L., P.N. Johnson, and E. Segarra. 2002. Ogallala Aquifer depletion: Economic impact on the Texas High Plains. *Water Policy* 4: 33-46.

Terrell, B. 1998. Economic impacts of the depletion of the Ogallala Aquifer: An application to the Texas High Plains. Unpublished MS thesis, Texas Tech University, Lubbock, Texas.

Texas Senate Bill 1, Austin, TX, 1997. Available at http://www.twdb.texas.gov/waterplanning/swp/1997/. Accessed February 4, 2009.

Texas Senate Bill 2, Austin, TX, 2002. Available at http://www.twdb.texas.gov/waterplanning/swp/2002/index.asp. Accessed February 4, 2009.

The Texas Joint Committee on Water Resources, 2002. Texas Joint Committee on Water Resources report to the 78th Legislature. Available at http://www.senate.state.tx.us/75r/Senate/commit/c895/Downloads/water.pdf. Accessed June 21, 2016.

Turner, C.G., K. McAfee, S. Pandey, and A. Sunley. 2011. Irrigation metering and water use estimates: A comparative analysis, 1999–2007. *Texas Water Development Board*, Report 378, July.

USDA ERS. Irrigation Water Use. *Ers.usda.gov*. Last modified Thursday, October 08, 2015. Available at http://www.ers.usda.gov/topics/farm-practices-management/irrigation-water-use.aspx. Accessed June 21, 2016.

Wheeler, E.A. 2008. Water conservation reserve program alternatives for the southern Ogallala Aquifer. Dissertation, Texas Tech University, Lubbock, Texas.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 46-61, August 2016

Climate Change, Agriculture and Water Resources in the Southwestern United States

*Emile Elias[1], Al Rango[1], Ryann Smith[2],
Connie Maxwell[1], Caiti Steele[2], and Kris Havstad[1]

[1]USDA Southwest Regional Climate Hub, USDA-ARS Jornada Experimental Range, Las Cruces, NM
[2]New Mexico State University, Las Cruces, NM
**Corresponding Author*

Abstract: In February 2014 the USDA established regional climate hubs across the United States to assist farmers, ranchers and foresters in adapting to the effects of climate change. The Southwest (SW) region encompasses six states which provide highly diverse agricultural crops including cotton, stone fruit and grapes. Here we report on the hydrologic context within which SW working landowners operate, with focus on regional water resources and likely impacts of climate change. Water is a critical component of agricultural vulnerability in the SW, where high agricultural production can occur with sufficient irrigation. Since 1978, crop yield declines were reported on 11-21% of total irrigated acres, mostly due to surface water shortage. Southwestern agriculture relies on groundwater, using it to supply at least one-third of the agricultural water demand across the region since 1955. Regional groundwater use varies over time, with a decline in the agricultural groundwater fraction in Arizona, but an increase in the fraction in Nevada and Utah. Observed and predicted changes in the southwestern hydrologic cycle can impact regional agriculture. Observed records show an increase in the fraction of precipitation falling as rain, which is expected to continue with future warming, and decrease the natural high-mountain storage reservoir provided by snowpack. Warming causes the snowmelt to peak earlier in the season than observed in historical records, and can reduce water available to crops during the summer months without additional water storage. Observed records indicate streamflow has shifted earlier in the year, most notably in snow dominated watersheds. A continuation of this trend may challenge regional agriculture by further limiting water supplies.

Keywords: *climate change, agriculture, water resources, Southwest Climate Hub*

In February 2014 the U.S. Department of Agriculture (USDA) created seven regional hubs and three subsidiary hubs across the U.S. for climate change adaptation and mitigation (Figure 1). The overall goal of the regional hubs is to maintain and strengthen agricultural production and conservation of natural resources under increasing climate variability and environmental change.

The USDA Southwest Regional Climate Hub (SW Hub) works in six states: New Mexico, Arizona, Utah, Nevada, California and Hawaii (and the U.S. affiliated Pacific Islands west of Hawaii) (http://www.climatehubs.oce.usda.gov/southwest). There are major deserts in five of the six SW Hub states and irrigation water is essential in maintaining agricultural production and local economies of the region. Consequently, the SW Hub focuses on research activities related to limited water supplies, drought, climate change and agriculture in the arid SW. This article provides the operational context within which the SW Hub functions with regard to natural and man-made water storage, irrigation, and drought as well as the likely implications for water management in a changed climate. Southwestern agriculture is defined by availability of water for irrigation, which will generally decrease in the southwestern U.S. over time (Dettinger, Udall, and Georgakakos 2015).

Irrigation in the Southwest

When there is sufficient water for irrigation, the region can support plentiful agricultural production. California leads the U.S. in agricultural productivity

(USDA–ERS 2015) and depends heavily on irrigation in the Central Valley. California accounts for 66-70% of the total irrigated area in the region, with 7.5 to 9 M irrigated acres (Figure 2). While the total irrigated area in other states in the SW region is less than that of California, irrigation is vital to crop production in all states of the region. Since 1978, total irrigated acres have fluctuated somewhat in each state, with an apparent slight decline in Arizona, Hawaii and Nevada. Irrigated area in Utah has remained fairly constant since 1978.

Snowmelt produces a large portion of the water used for irrigation in the SW. In certain snowmelt mountain basins of the SW, snow is the source of up to 75% of the seasonal water supply (Serreze et al. 1999; Hamlet et al. 2005; Rango 2006). The snow cover in New Mexico, Arizona, Utah, Nevada, and California usually persists for up to six months or more, which acts as a natural reservoir releasing the snowmelt water gradually during the early part of the growing season (Serreze et al. 1999). Storage of water runoff from snow in artificial reservoirs then takes over in supplying irrigation water for the latter part of the growing season. Traditionally water is stored in surface reservoirs during times of excess. During drought conditions, reservoir supplies can be depleted, leading to low water levels associated with reservoir draw-down. As surface water supplies in streams and reservoirs decline, farmers rely upon supplemental groundwater supplies. Pumping of water from wells can be costly and result in declining water tables, further incurring pumping costs (Konikow 2013; Taylor et al. 2013). Such activities are essentially mining the groundwater, or removing groundwater in excess of the recharge rate of the aquifer.

Diminished Yields from Irrigation Interruption

The farm and ranch irrigation survey (FRIS), conducted every five years, collects information on irrigation across the U.S. (NASS 2015). The FRIS reports crops with diminished yields resulting from irrigation interruption by cause. Between 1978 and 2012, 11-21% of the total irrigated acreage in the SW region had crop yield declines due to irrigation interruption. The total diminished regional acreage ranged from 1.4 M acres (1982) to 2.4 M acres (2007). The potential causes included in the survey are shortage of surface water, shortage of groundwater, irrigation equipment failure, energy shortage, poor water quality, loss of water rights and other causes. Since 1978, the most reported cause of irrigation interruption and diminished crop yields in the SW is shortage of surface water (Figure 3). Over 70% of respondents indicated surface water shortage as the cause of irrigation interruption in the most recent surveys (2002, 2007, and 2012). Groundwater shortage was indicated for 3-30% of acreage with interrupted irrigation, with the highest percentage in the most recent surveys (2007, 2012). Shortage of surface water and groundwater appear to comprise a larger portion of the yield reduction cause in more recent years (1992, 2002, 2007 and 2012). Irrigation equipment failure caused crop yield reduction 9 to 42% of the time. Energy shortage was reported at 22% in 1978, but declined as a cause to 6% in 2012. Poor water quality, loss of water rights and water costs were less frequently reported causes of irrigation interruption and diminished crop yields.

These reductions in crop yield based upon unexpected irrigation interruptions reflect the current climate within which SW farmers operate. The hotter and drier conditions predicted in a changing climate represent a ‘new normal’ that could further exacerbate crop yield declines due to likely additional irrigation interruptions (Diffenbaugh, Swain, and Touma 2015). Irrigation serves as an important adaptive strategy to cope with higher temperatures (Haim, Shechter, and Berliner 2008), however shifts in relative profitability of irrigated production and declining water supplies may drive declining irrigated acreage in parts of the Southwest, mostly due to surface water constraints of a changing climate (Marshall et al. 2015).

Discontinued Irrigation in the Southwest

FRIS also reports discontinued irrigation statistics. Historically, the reason for discontinued irrigation is not consistently lack of water. For example, in 2003 most farmers cited sufficient soil moisture as the reason for discontinued irrigation, whereas shortage of surface and groundwater was the major reason in 2013. In 2013, 7,805 farms (470,404 acres) throughout the SW reported that they had discontinued irrigation since the previous census year; among the reasons for



Figure 1. United States Department of Agriculture Southwest Climate Hub Region.

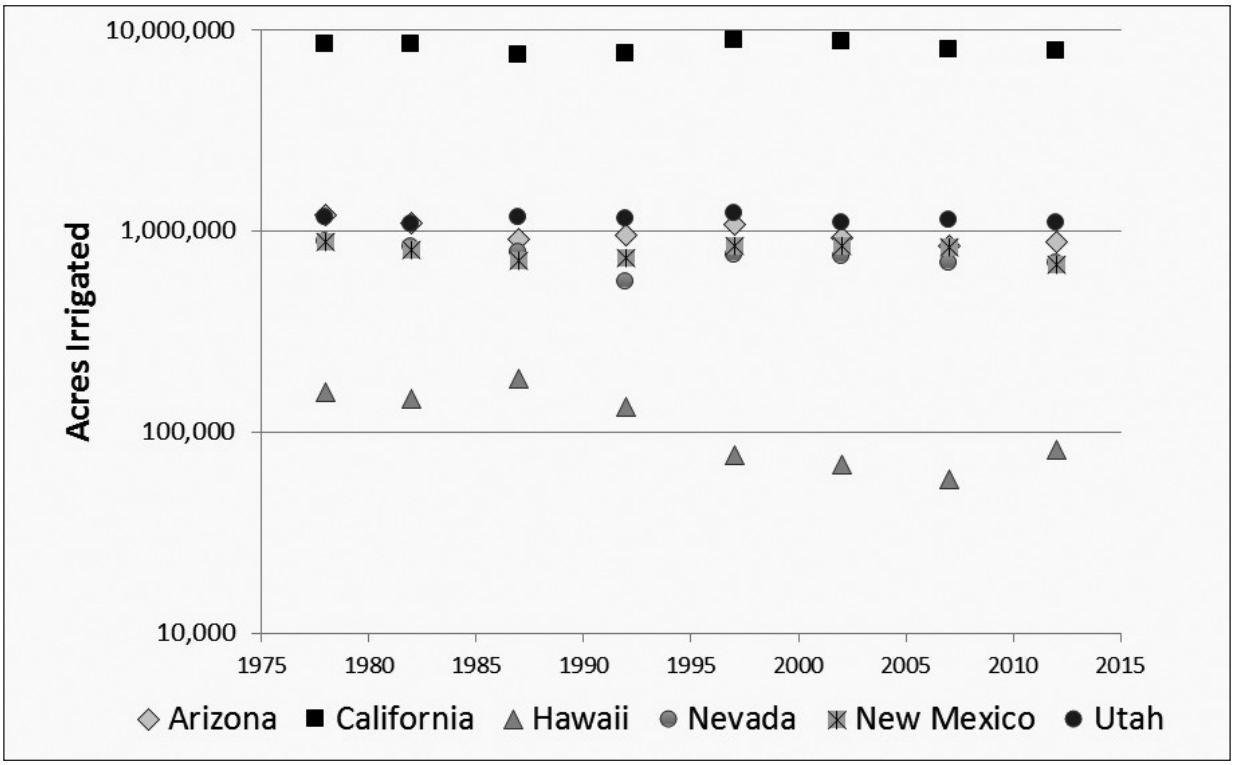


Figure 2. Acres irrigated by state in the Southwest Climate Hub region (NASS 2015).

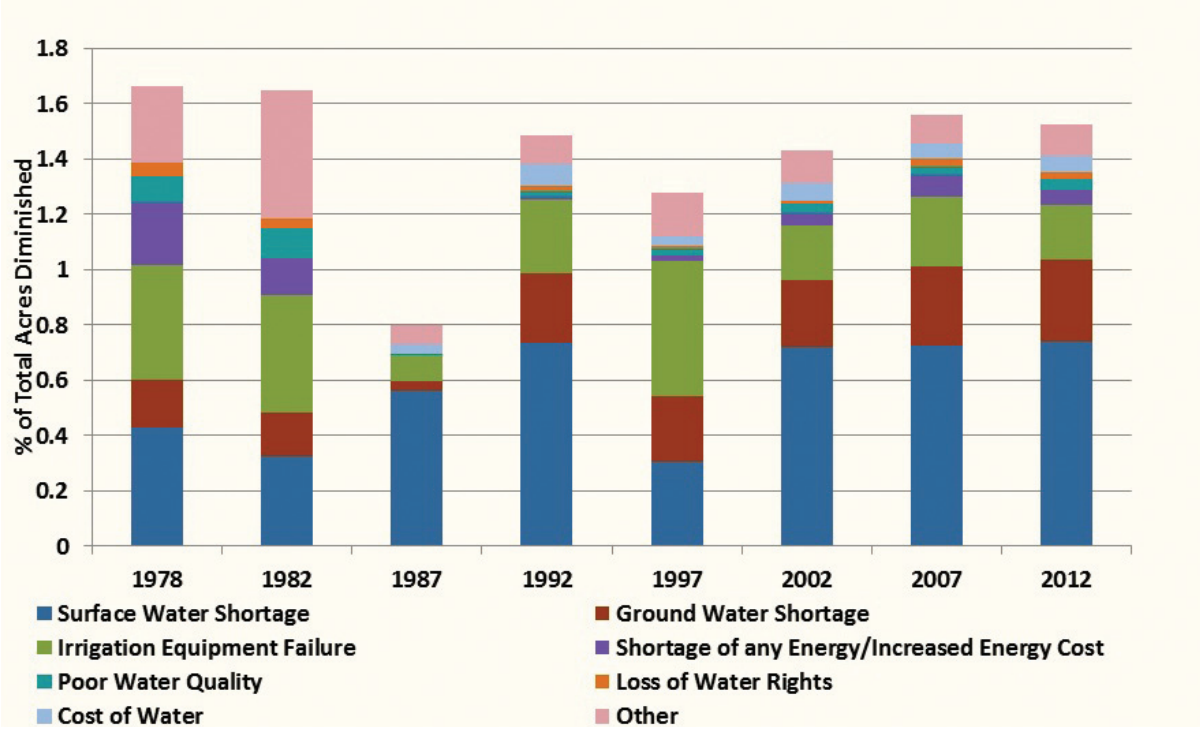


Figure 3. Diminished crop yields due to irrigation interruption by cause in the southwestern Hub states, 1978-2012 (NASS 2015). Note: total percentage can be larger than 100% because respondents could select multiple causes of irrigation interruption.

discontinuing irrigation were shortages of surface and groundwater and sufficient soil moisture. Sufficient soil moisture accounted for 8.4% of the discontinued irrigation acreage in 2013 (519 farms; 39,922 acres). Shortage of surface water accounted for 37% of the discontinued irrigation (1,704 farms; 173,674 acres) and groundwater shortage accounted for 22% of the discontinued irrigation in 2013 (698 farms; 102,486 acres). Interestingly, in 2013, 9% of farms and 4% of acreage (708 farms; 19,783 acres) reported the discontinued irrigation as permanent.

Southwestern Reliance on Groundwater

Water use data by state indicate changes in total and agricultural water use over a five year interval since 1955 (U.S. Geological Survey 2016). Between 1955 and 2010, groundwater has become a larger portion of the total agricultural water use in parts of the SW, primarily NV and UT, and a smaller portion of the total agricultural water use in other parts of the region, primarily AZ. Since 1955, total agricultural water use in the region ranged between 30,000 MGD and 54,000 MGD, with a general decline since 1980 (Table 1). The portion of total water use that is directed to agriculture for the SW region has declined from 92% in 1955 to 78% in 2005 and 2010. The overall fraction of agricultural water use supplied by groundwater in the region was between 44-46% in the early record (1955-1980), with a slight decline to 38-40% in the latter record (1985-2010) (Figure 4). Thus, southwestern agriculture consistently relies on groundwater to supply more than one-third of the agricultural water demand. As of 2010, AZ and CA groundwater fraction was slightly less than 40%, whereas NV and NM were slightly more than 40%. Utah represents a regional anomaly, with only 20% of the total agricultural water supplied by groundwater. In terms of total amount of groundwater use for agriculture, southwestern states are ranked CA (8,000 to 18,000 MGD) > AZ (1,900 to 5,000 MGD) > NM (900 to 1,600 MGD) > NV and UT.

Consistent with having the largest agricultural production in the region, California agriculture used between 60-69% of the total agricultural

Table 1. Agricultural water use for the contiguous Southwest Climate Hub Region and Southwestern states, 1955 to 2010 (U.S. Geological Survey 2016).

MGD		1955	1960	1965	1970	1975	1980	1985	1990	1995	2000	2005	2010
Southwest	Groundwater	16,696	13,001	17,057	21,986	23,603	24,451	15,294	15,774	15,538	17,112	13,671	13,259
	Total	38,637	29,560	39,243	48,777	51,704	54,232	46,288	43,164	43,374	45,796	38,504	36,617
	Fraction	0.43	0.44	0.43	0.45	0.46	0.45	0.33	0.37	0.36	0.37	0.36	0.36
Arizona	Groundwater	4,700	3,000	3,900	3,800	4,200	3,710	2,520	2,081	2,163	2,748	2,280	1,757
	Total	6,900	4,700	5,900	6,200	7,000	7,112	5,577	5,391	5,704	5,404	4,824	4,644
	Fraction	0.68	0.64	0.66	0.61	0.60	0.52	0.45	0.39	0.38	0.51	0.47	0.38
California	Groundwater	10,089	8,561	11,034	16,038	17,042	18,036	10,420	10,897	11,024	11,982	8,894	8,945
	Total	23,111	17,986	25,082	33,091	35,101	37,087	30,844	28,287	29,353	31,444	25,162	24,250
	Fraction	0.44	0.48	0.44	0.48	0.49	0.49	0.34	0.39	0.38	0.38	0.35	0.37
Nevada	Groundwater	213	274	462	382	534	534	755	872	642	567	693	669
	Total	1,918	1,777	1,967	2,985	3,142	3,143	3,373	2,828	1,650	2,107	1,522	1,629
	Fraction	0.11	0.15	0.23	0.13	0.17	0.17	0.22	0.31	0.39	0.27	0.46	0.41
New Mexico	Groundwater	1,416	921	1,233	1,312	1,313	1,610	1,183	1,389	1,308	1,229	1,319	1,289
	Total	2,520	1,845	2,765	2,844	2,944	3,620	2,871	3,033	3,026	2,862	2,891	2,756
	Fraction	0.56	0.50	0.45	0.46	0.45	0.44	0.41	0.46	0.43	0.43	0.46	0.47
Utah	Groundwater	278	245	428	454	514	561	415	535	401	586	485	599
	Total	4,188	3,252	3,529	3,657	3,517	3,270	3,623	3,625	3,641	3,979	4,105	3,338
	Fraction	0.07	0.08	0.12	0.12	0.15	0.17	0.11	0.15	0.11	0.15	0.12	0.18

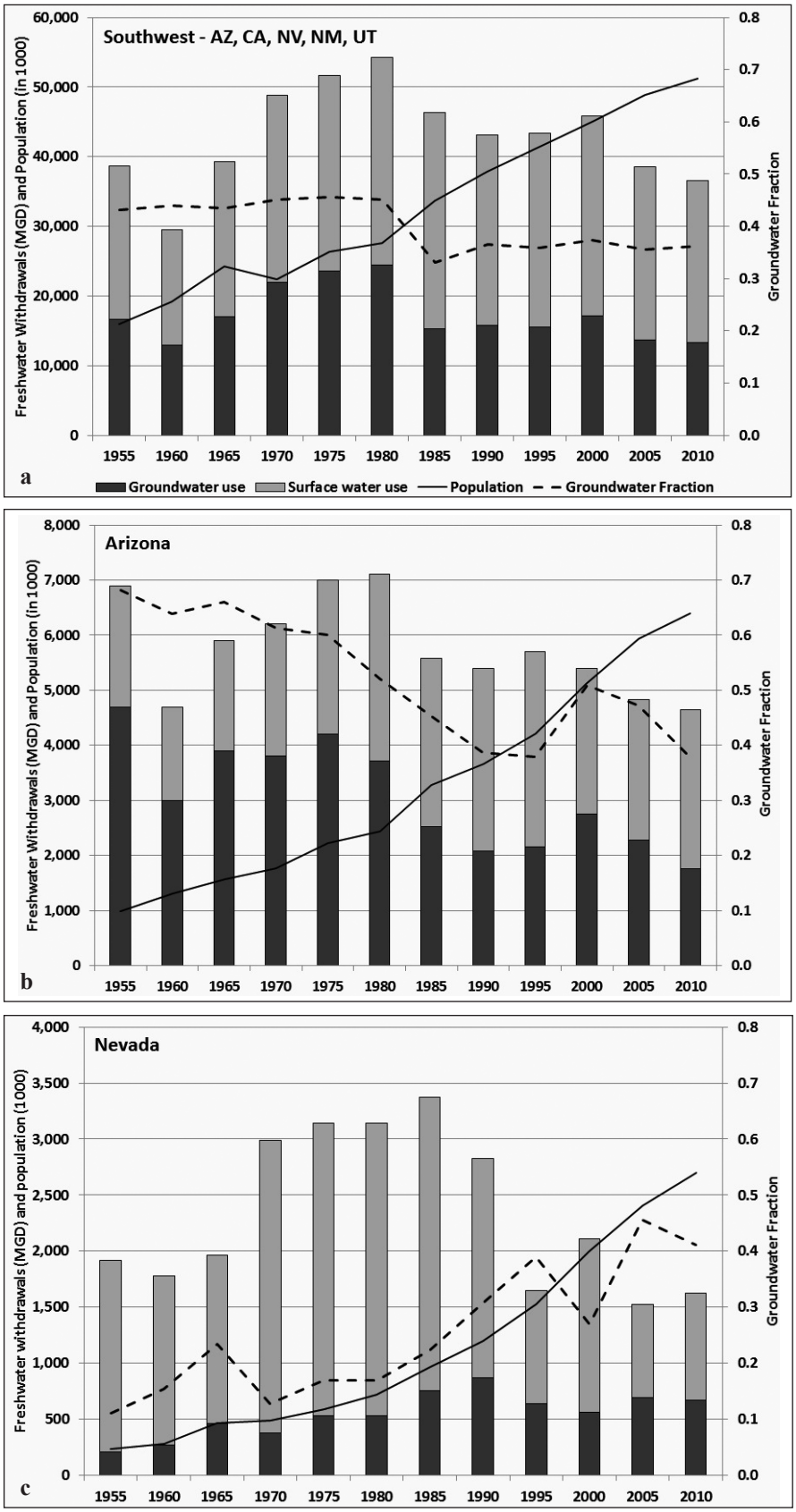


Figure 4. Total agricultural freshwater withdrawals, regional population change, and fraction of agricultural water use supplied by groundwater for a) southwestern states (AZ, CA, NM, NV, and UT), b) Arizona and, c) Nevada, 1955 to 2010 (U.S. Geological Survey 2016).

water in the region between 1955 and 2010, with around half the agricultural water of California supplied by groundwater. In 1955, Arizona agriculture used the largest percentage of groundwater to total agricultural water use or agricultural groundwater fraction (AGF) of any state/year, with 68% of the total agricultural water supplied by groundwater (Figure 4). Since 1955, there has been a gradual decline in groundwater use fraction in AZ for agriculture, with the 2010 AGF of 38%. A small increase in AGF in 2000 to 50% corresponds with the drought conditions of that year, with a statewide Palmer Drought Severity Index value of -2.53 (NOAA 2015). Arizona's Groundwater Management Act may be largely responsible for the decline in AGF in that State (Colby and Jacobs 2007).

Arizona's Groundwater Management Act was a result of the Central Arizona Project (CAP). Although the CAP, which diverts Colorado River water to central Arizona, was conceived in the 1940s, legal action, including decisions made by the Supreme Court on Colorado River water allocation hindered CAP construction until the Colorado River Basin Project Act of 1968. It took more than a decade to secure funds and political will for CAP construction and ultimately required Arizona to demonstrate a commitment to curtail groundwater overdrafting in the state, leading to the adoption of the Groundwater Management Act of 1980. The Act established active management areas where groundwater dependent development was discouraged, and irrigation nonexpansion areas. The CAP began supplying water in the early 1990's. The total amount of groundwater used in Arizona agriculture has generally declined since 1980 (Figure 4).

In September 2014, California approved the Sustainable Groundwater Management Act which empowers local agencies to adopt groundwater management plans and requires overdrafted basins to have a management plan by 2020 (State of California 2016). While California groundwater use for agriculture hovered around 10,000 MGD between 1985 and 2010, groundwater resources across agricultural regions of California have been more stressed during the 2012-2015 drought than at any other time in history in regard to depth to groundwater, groundwater demand and pumping costs increasing (Harter 2015). The Act aims at long-term preservation of groundwater resources for multiple uses, including agriculture.

In contrast to the decline in AGF in Arizona, the AGF in Nevada increased from 11% in 1995 to 41% in 2010, with seven counties in southern and central NV reporting an AGF >80% in 2010. AGF also increased in Utah from 6% in 1955 to 18% in 2010, due to increased groundwater use in five counties of western UT. NM AGF ranged from 41-56%, indicating that roughly half of agricultural water use in NM has been supplied by groundwater since 1955. There are parts of the region that appear to have become less reliant on groundwater over time (AZ) whereas other parts of the region (NV and UT) seem to have become more reliant on groundwater to support agricultural production.

Greater depth to water increases pumping costs associated with groundwater use. Depth to water is consistently greater in AZ than other states, but has generally declined since 1984, possibly influenced by the observed shift away from agricultural groundwater use in that state. NV, which has been using more groundwater to support agriculture, also has an apparent increase in depth to water since 1974, possibly resulting from groundwater mining (Figure 5). The agricultural groundwater fraction within each state depicts areas of high groundwater reliance for production (Figure 6).

The projection of increased groundwater demand reinforces the need for novel methods to measure changes in total terrestrial water storage over large areas of the southwest. NASA's Gravity Recovery and Climate Experiment (GRACE) data have been used to detect changes in total water of a relatively large area (NASA 2015). GRACE consists of twin polar-orbiting satellites flying at a distance of ~220 km. Regional changes in terrestrial water storage affect the gravitational force exerted on the twin satellites causing minute changes in the speed of the satellites and the distance between them. GRACE data have been used in conjunction with other datasets to develop drought indicators (Houborg et al. 2012) and to estimate changes in regional and national water mass (Famiglietti and Rodell 2013). GRACE data were used to estimate groundwater loss in the Colorado Basin during the drought of 2004-2013 and researchers found a large freshwater loss in the region (Castle

et al. 2014). Another analysis relying on GRACE reported that total water storage decreased during the 2010-2013 drought, with most of the decline from surface reservoir and soil moisture storage in the upper basin and additional reductions in groundwater storage in the lower basin (Scanlon et al. 2015). As climate changes, traditional and novel ways of measuring available water may help ameliorate projected water scarcity.

Climate Change and Water Resources in the Southwestern United States

Climate change will affect the hydrologic cycle through various pathways including shifts in precipitation, potential evapotranspiration (ET) and snowmelt runoff timing and amount (Dettinger, Udall, and Georgakakos 2015). Thus, southwestern agriculture may suffer diminished water supplies in much of the region.

Changes in Precipitation

Annual average precipitation has decreased in parts of the Southwest (Walsh et al. 2014) and is projected to continue to decrease (Cayan et al. 2013). On a global scale, precipitation models are in complete agreement with decreased precipitation in the subtropics and increased precipitation at higher latitudes (Walsh et al. 2014). Since the U.S. is located in the transition zone between the two regions, projected changes in national precipitation range from small increases to small decreases leading to uncertain changes in overall future annual precipitation (Walsh et al. 2014). The annual average precipitation changes to expect in the SW U.S. are often quoted as being between plus or minus 10% and within the range of natural variability (Walsh et al. 2014). In general, the southern parts of the U.S. are expected to receive less precipitation in the future.

Snow is an important resource for several reasons. First, the snowpack is a natural reservoir that delays snowmelt until later in the season when irrigation water can efficiently be used to grow crops. But, temperatures will continue to rise throughout the 21st Century (Walsh et al. 2014). More often snow will be converted to rain as it falls in a basin. Even if the snow remains in the solid form as it reaches the ground, the high basin temperatures will quickly melt the snowpack, and the residence time for snow in the basin will be reduced.

The amount of snow that melts in a basin will occur earlier in time except in the highest elevation zones where below freezing temperatures persist. Low elevation snow course measurement stations show a strong decreasing trend in snow water equivalent (SWE) whereas there is little trend in stations higher than 2,500 meters above sea level (Regonda et al. 2005). Observed 1960–2002 snow course measurements show general decreasing trends in April 1 SWE across the western U.S. (Mote 2006). Warming trends will affect snowpack, even at higher elevations. Warming causes the snowmelt to peak earlier in the season making it unavailable to crops when needed later in the year. This facet of the changing snow hydrology is already leading to considerations of growing different crops, development of new water storage structures, or even leasing of water rights in order to continue efficiently using water when available.

The changes in solid to liquid precipitation can already be observed in current precipitation records. Observed records show an increase in the fraction of precipitation falling as rain, with considerable interannual variability causing changes at the local scale (Fritze, Stewart, and Pebesma 2011; Mankin and Diffenbaugh 2015). Some basins of the SW, particularly the Sierra Nevada mountains of California and parts of the Sangre de Cristo mountains of New Mexico, have shifted from mostly snowmelt dominated in the mid-1900s to mostly rain dominated in the early 21st century (Fritze, Stewart, and Pebesma 2011). When using observed data, one must be concerned with the snow catch deficit that is caused by wind. Most gauges do not have wind shields, and unfortunately the snow records are not corrected for this deficit issue, although correction factors are readily available (Sevruk 1985, 1989). Future research incorporating snow catch deficit correction is recommended. Beyond anticipated changes in precipitation amount, phase and regional snowpack, increased temperatures will impact other parts of the hydrologic cycle.

Changes in Evapotranspiration

Evapotranspiration (ET) is one of the largest components of the water cycle in the west,

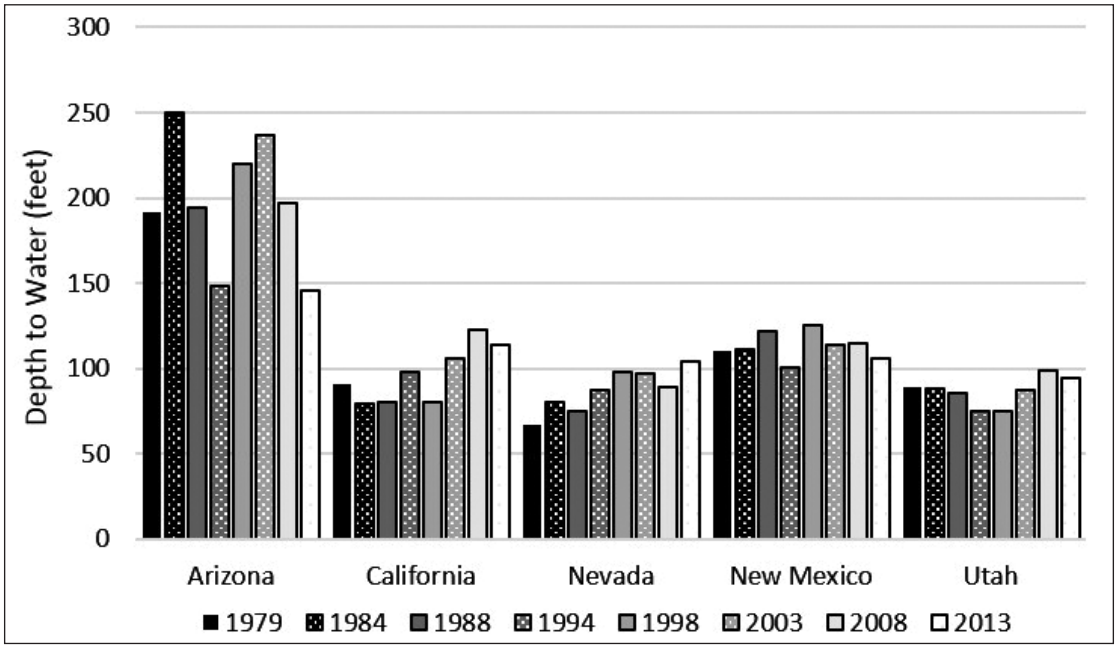


Figure 5. Depth to water (in feet) from irrigation wells used on farms by state, 1979 to 2013 (NASS 2015).

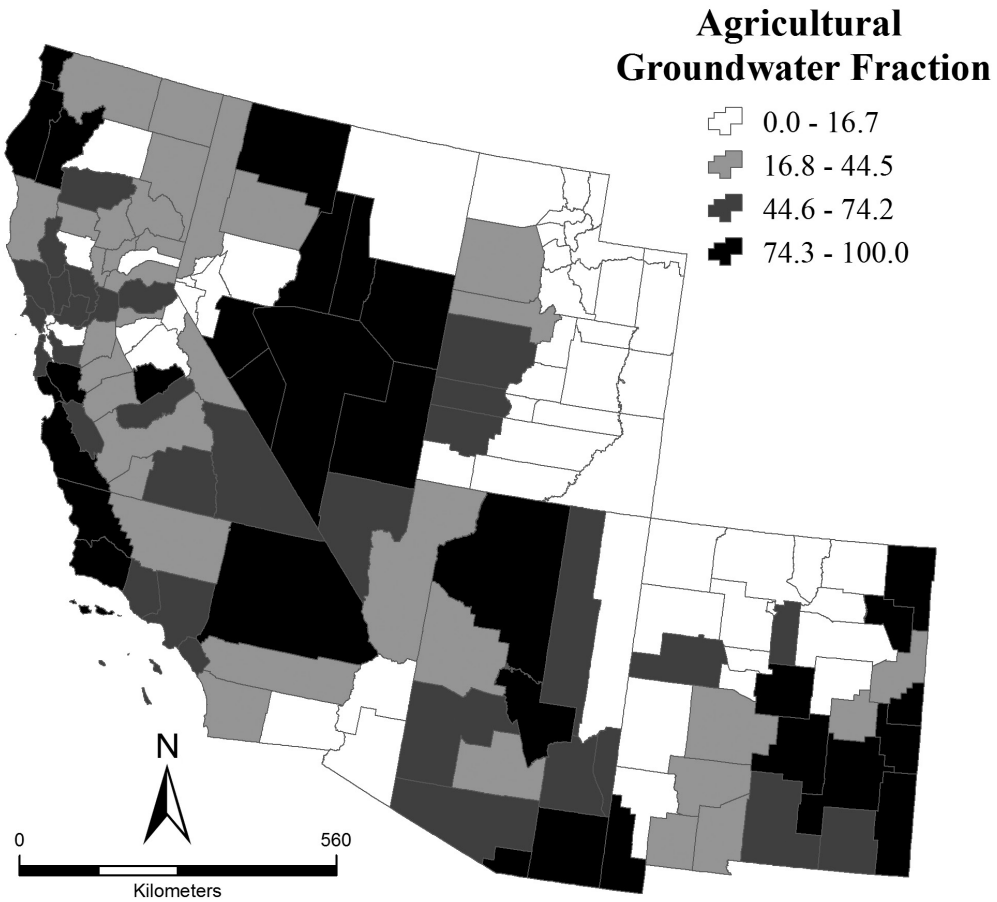


Figure 6. Agricultural groundwater fraction for the southwestern states in 2010. Fraction is groundwater used for irrigation/total water used for irrigation (U.S. Geological Survey 2016).

second to precipitation (Dettinger, Udall, and Georgakakos 2015). Global land ET estimates (derived from FLUXNET, meteorological observation, and remote sensing) suggest that the rate of ET increased from the early 1980s to the late 1990s (Jung et al. 2010). This trend of rising ET rates disappeared after the last major El Niño event in 1998, after which point global land ET rates began to decline. These trends are consistent with expected hydrologic cycle changes due to temperatures, radiative forcing, and evaporative demand (Jung et al. 2010).

Dettinger, Udall, and Georgakakos (2015) state that although projected actual ET rates vary by region, potential ET is expected to increase with warming across the Southwestern U.S. Potential ET is the maximum amount of water that would be evapotranspired if enough water were available, whereas actual ET is the amount of water actually evapotranspired and is limited by available water content. ET rates and totals are likely to be affected in the SW by changes in land use, vegetation, soil moisture, and changes in the lengths of growing and snow-covered seasons. Goulden and Bales (2014) found that warming projected for 2100 could increase ET in the Kings River basin in California's Sierra Nevada by 28%. Increased temperature, which can shift snowmelt timing and soil moisture, is also predicted to increase ET rates (Christensen and Lettenmaier 2007; Ficklin, Luedeling, and Zhang 2010; Hay, Markstrom, and Ward-Garrison 2011; Hoerling et al. 2012).

Changes in Runoff

In the early 1990s, researchers observed that the fraction of annual runoff occurring between April and July had been declining (Wahl 1991; Aguado et al. 1992), possibly related to a trend in decreasing snowmelt runoff (Roos 1991). The trend was most evident in mid-altitude basins where winter temperatures are near freezing and small temperature increases can result in less snow formation and earlier melt (Dettinger and Cayan 1995). The observed earlier snowmelt in California was related to trends toward warmer winters since the 1940s (Dettinger and Cayan 1995). These trends were evaluated over a broader area of the western U.S. using 302 stream gauges. Researchers observed earlier spring pulse onset and center of mass for annual flow for snowmelt dominated basins due, in part, to higher winter and spring temperatures (Stewart, Cayan, and Dettinger 2005). While temperature changes were partly controlled by decade scale Pacific climate mode (PDO), a springtime warming trend that spans the PDO phases was identified. Thus, a portion of the streamflow timing trend appears to be related to temperature increases consistent with observed increases in global temperature. More recent analyses of shifts in streamflow timing from1948 to 2008 indicate that streamflow has continued to shift earlier in the year, most notably in snow dominated watersheds (Fritze, Stewart, and Pebesma 2011). Researchers have observed that shading (Lundquist and Flint 2006) and dust on snow (Painter et al. 2007) have an important role in peak streamflow timing and snow cover depletion. Earlier snowmelt has also been observed using imagery from the Landsat program (Hall et al. 2015).

Observed changes in SWE, runoff timing and snowpack area are reflected in climate change simulations in the western U.S. Widespread warming caused decreasing trends in April 1 SWE across much of the Western United States (Hamlet et al. 2005) . Simulations of the impacts of temperature and precipitation changes using the Snowmelt Runoff Model (SRM) indicate advancement in runoff timing and reduced summer flow during the growing season of some crops (Rango and van Katwijk 1990; van Katwijk and Rango 1991; van Katwijk, Rango and Childress 1993; Rango and Martinec 1994, 1999, 2000; Rango, Martinec, and Roberts 2008; Elias et al. 2015; Elias et al. 2016). Measurement of impacts of climate change on snow covered area (SCA), streamflow timing and runoff volume of twenty-four sub-basins of the Upper Rio Grande, using four future climate conditions, reflect climate change simulation results in other western locations. Results indicate change in total annual volume with disparity between increased volume in wetter simulations (+7%) and decreased volume (-18%) in drier simulations. Peak flow is 14-24 days earlier in the future climates. Among the twenty-four sub-basins there was considerable range in mean melt season SCA, total volume change and runoff timing advancement, indicating that climate change is best evaluated at the sub-basin scale. The

timing of water supply will shift to earlier in the year and water management flexibility for current water users may decrease (Elias et al. 2015). These observed and simulated impacts to the hydrologic cycle are cumulative and serve to decrease surface water availability and likely increase reliance on groundwater resources to support agricultural production.

Changes in Drought

While some studies argue that a link between climate and drought cannot be made, there is recent scientific evidence clearly linking southwestern drought to the increased temperatures of climate change (Mann and Gleick 2015). The discrepancy in studies linking climate change and drought is due in part to the multiple definitions of drought, where studies considering only precipitation changes find little link to climate change, but metrics incorporating both incoming and outgoing moisture show increased drought risk in a future warmer climate. The SW climate might have already transitioned to a drier climate state leading to higher drought risk (Prein et al. 2016). Recent droughts in the upper Colorado River Basin have been amplified by warmer temperatures (Woodhouse et al. 2016). The recent California drought beginning in 2012 has similarly been linked to anthropogenic warming (Cook, Ault, and Smerdon 2015; Diffenbaugh, Swain, and Touma 2015; Mann and Gleick 2015). The California drought spanning 2012-2014 was the most severe drought in the past 1,200 years and was driven by reduced precipitation and record high temperatures (Griffin and Anchukaitis 2014). Griffin and Anchukaitis conducted their analysis prior to 2015. The 2015 water year in California was the hottest and driest on record, thereby exacerbating the drought severity reported in 2014. The hydrologic drought in the Southwest U.S. has been linked to observed and predicted increased aridity in the region, where the recent drought, the Dust Bowl and the 1950s droughts are projected to become the new climatology of the Southwest (Seager et al. 2007). Should drought conditions become the new 'normal' in the face of climate change, groundwater reserves would likely decline (Taylor et al. 2013). A study of the impacts of climate change on groundwater recharge indicates average declines of 10-20% in total aquifer recharge across southwestern aquifers, but with a wide range of uncertainty (Meixner et al. 2016).

Summary

Irrigation of cropland is critical in five of the six states of the SW region for continued high agricultural productivity across the region. The irrigation water supply comes primarily from snowmelt runoff, and this water supply is subject to the effects of climate change. If sufficient irrigation water supplies exist, agricultural production can be high in the SW. Since 1978, total irrigated acres have fluctuated somewhat in each state, with a slight decline in Arizona, Hawaii and Nevada. Between 1978 and 2012, 11-21% of the total irrigated acreage in the SW region had crop yield declines due to irrigation interruption. The most reported cause of irrigation interruption and diminished crop yields was surface water shortage. Shortage of surface water and groundwater appear to comprise a larger portion of the crop yield reduction in more recent years (2002, 2007, and 2012).

The portion of total water use that is directed to agriculture for the SW region has declined from 92% in 1955 to 77% in 2005 and 2010. At the state level, the overall fraction of agricultural water use supplied by groundwater was between 33-46% since 1955, with a slight decline in the latter half of the record (1985-2010). Southwestern agriculture relies consistently on groundwater, using it to supply more than one-third of the agricultural water demand across the region, with certain counties using groundwater to satisfy more than 80% of the agricultural water demand. In 2010, 40-50% of agricultural water use was supplied by groundwater in all SW states except UT.

In 1955, Arizona agriculture used the largest agricultural groundwater fraction (AGF) of any state/year, with 68% of the total agricultural water supplied by groundwater. Since that time, there has been a gradual decline in groundwater use fraction in AZ for agriculture, with the 2010 AGF of 38%. This decline may be related to the enactment of the Arizona Groundwater Management Code in 1980. In contrast, the AGF in Nevada and Utah increased since 1955. Depth to water is consistently largest in AZ, but has been recovering since a peak in 1984,

possibly influenced by the observed shift towards less agricultural groundwater use in that state. NV, which has been using more groundwater to support agriculture, has an observed increase in depth to water since 1974.

Observed and predicted changes in the southwestern hydrologic cycle will impact regional agriculture. Observed records show an increase in the fraction of precipitation falling as rain, which is expected to continue with future warming. Potential ET is expected to increase with warming across the Southwestern U.S., with projections in parts of the region of an increase in ET of 28% by 2100. Annual average precipitation has decreased in parts of the SW and is projected to continue to decrease. Snowpack functions as a natural reservoir, but warming causes the snowmelt to peak earlier in the season and reduces water available to crops during the summer months. Observed records indicate streamflow has continued to shift earlier in the year, most notably in snow dominated watersheds. This facet of the changing snow hydrology is already leading to considerations by agricultural producers of changing to different crops, development of new water storage structures, or leasing water rights in order to continue efficiently using water when available.

The observed shifts in streamflow timing will likely continue into the future with the increasing temperatures of a changing climate, first at lower elevations and then progressively into higher elevations as temperatures increase. The storage reserves of high mountain snowpack will decline. Should irrigated agriculture continue at a similar footprint, then reliance on groundwater to support production will likely increase, with concurrent economic and environmental consequences. Climate change affects water resources of the SW to a degree that is probably unmatched anywhere in the U.S. SW agriculture, which relies heavily on available water for production, is particularly vulnerable to the reduced water availability stemming from current and predicted climate changes.

Acknowledgements

We thank Danielle Shannon with the Northern Forests Climate Hub for providing the Regional Climate Hub Map (Figure 1). Thanks to Sharon Megdal, director of the Arizona Water Resources Research Institute, for sharing insight into groundwater trends in Arizona.

Author Bio and Contact Information

Emile Elias is deputy director of the USDA Southwest Climate Hub and research hydrologist at the USDA-ARS. Her current research focuses on the impacts of climate change on water quantity and quality in the Southwestern U.S. Additional recent research includes vulnerability assessment of Southwestern crops to a changing climate. She may be contacted at emile.elias@ars.usda.gov, phone: 575-646-5190, fax: 575-646-5889, or physical address: USDA Southwest Climate Hub, Jornada Experimental Range, 2995 Knox Street, Las Cruces, NM, 88003.

Albert Rango is a research hydrologist at the USDA-ARS-Jornada Experimental Range in Las Cruces. He directs the Unmanned Aerial Vehicle research program that has operated in New Mexico, Idaho, Arizona, and Alaska. He was also recently selected as the director of the USDA Southwest Climate Hub that covers NM, AZ, UT, NV, CA, and HI.

Ryann Smith is a junior at New Mexico State University pursuing a Bachelor of Science in Agriculture degree with a major in Wildlife Science. She works as a research assistant at the USDA Southwest Climate Hub.

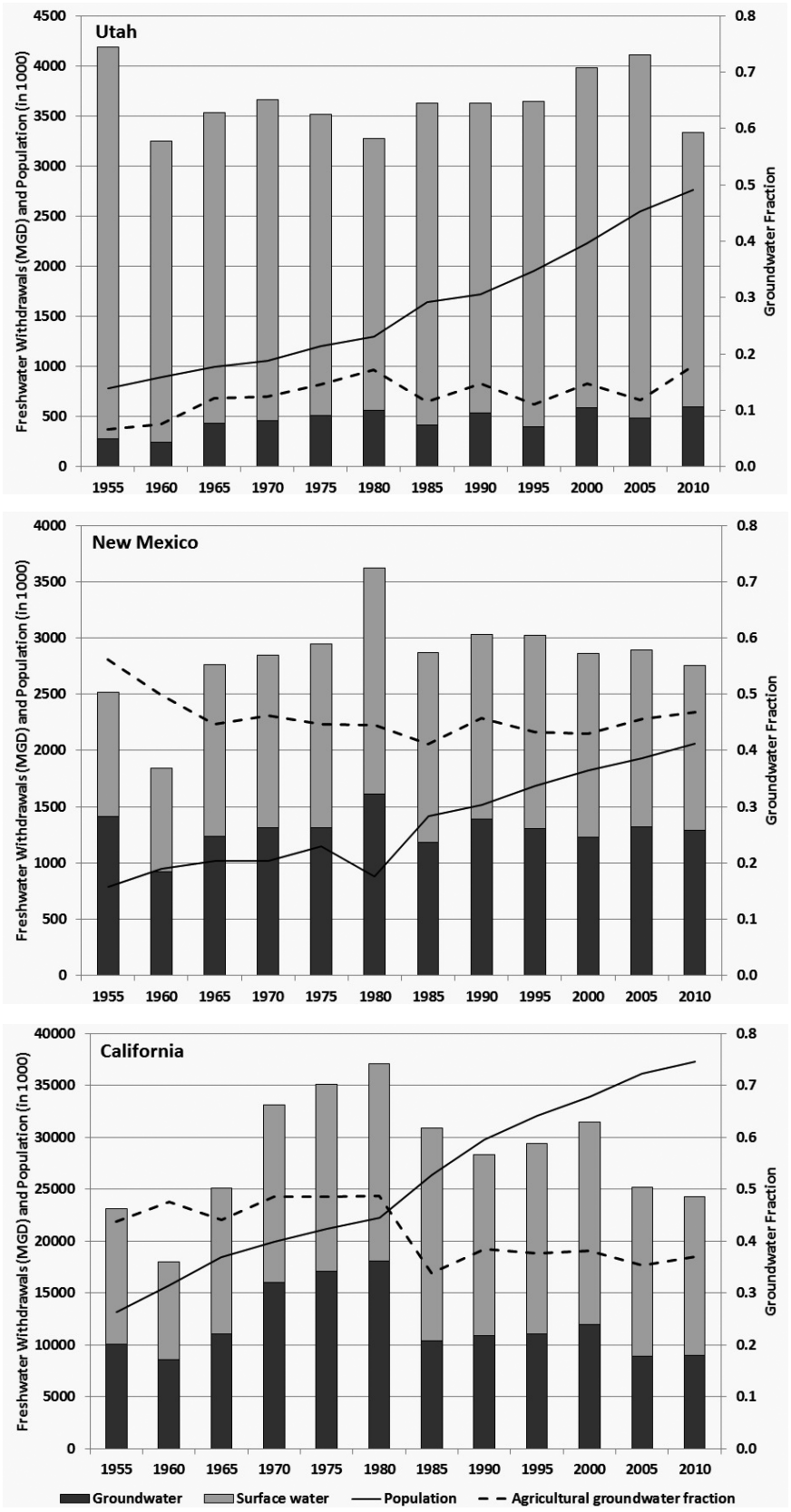
Connie Maxwell is a Biological Science Technician at the USDA-ARS Jornada Experimental Range specializing in plant ecophysiology, and she manages flight operations for the Unmanned Aerial Vehicle research program. Her current research focuses on Southwestern groundwater use in a historical context.

Caiti Steele is deputy director of the USDA Southwest Climate Hub. Her primary research interests are terrestrial remote sensing (vegetation and snow); current research focuses on how climate change affects the contribution of snowmelt to surface water availability. Additional research includes vulnerability assessment of agricultural and forest resources in Hawaii and U.S. affiliated Pacific Islands.

Kris Havstad was Research Leader and Supervisory Range Scientist at the USDA-ARS Jornada Experimental Range from 1988-2016. His primary research interests include rangeland ecology and management.

Appendix

Agricultural water use, groundwater fraction and population change in California, New Mexico and Utah, 1955 to present.



References

Aguado, E.D., D. Cayan, L.G. Riddle, and M. Roos. 1992. Climatic fluctuation and the timing of west coast streamflow. *Journal of Climate* 5: 1468-1483.

Castle, S.L., B.F. Thomas, J.T. Reager, M. Rodell, S.C. Swenson, and J.S. Famiglietti. 2014. Groundwater depletion during drought threatens future water security of the Colorado River Basin. *Geophysical Research Letters* 41(16): 5904-5911. DOI: 10.1002/2014GL061055. Accessed June 28, 2016.

Cayan, D., M. Tyree, K. Kunkel, C. Castro, A. Gershunov, J. Barsugli, A.J. Ray, J. Overpeck, M. Anderson, J. Russell, B. Rajagopalan, I. Rangwala, and P. Duffy. 2013. Future climate: Projected average. In: *Assessment of Climate Change in the Southwest United States: A Report Prepared for the National Climate Assessment*, G. Garfin, A. Jardine, R. Merideth, M. Black, and S. LeRoy, (Eds.). Southwest Climate Alliance, Washington, D.C., pp. 101-125.

Christensen, N.S. and D.P. Lettenmaier. 2007. A multimodel ensemble approach to assessment of climate change impacts on the hydrology and water resources of the Colorado River Basin. *Hydrology and Earth System Sciences* 11(4): 1417-1434.

Colby, B.G. and K.L. Jacobs. 2007. Arizona Water Policy: Management Innovations in an Urbanizing, Arid Region. Resources for the Future, Washington, D.C.

Cook, B.I., T.R. Ault, and J.E. Smerdon. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. *Scientific Advances* 1(1). Available at http://advances.sciencemag.org/content/1/1/e1400082. Accessed June 28, 2016.

Dettinger, M. and D. Cayan. 1995. Large-scale atmospheric forcing of recent trends towards early snowmelt in California. *Journal of Climate* 8: 606-623.

Dettinger, M., B. Udall, and A. Georgakakos. 2015. Western water and climate change. *Ecological Applications* 25(8): 2069-2093. Available at http://onlinelibrary.wiley.com/doi/10.1890/15-0938.1/abstract. Accessed June 28, 2016.

Diffenbaugh, N.S., D.L. Swain, and D. Touma. 2015. Anthropogenic warming has increased drought risk in California. Proceedings of the National Academy of Sciences 112(13): 3931-3936. DOI: 10.1073/pnas.1422385112. Available at http://www.pnas.org/content/112/13/3931.abstract. Accessed June 28, 2016.

Elias, E.H., A. Rango, C.M. Steele, J.F. Mejia, and R. Smith. 2015. Assessing climate change impacts on water availability of snowmelt-dominated basins of the Upper Rio Grande Basin. *Journal of Hydrology: Regional Studies* 3(0): 525-546. Available at http://dx.doi.org/10.1016/j.ejrh.2015.04.004. Accessed June 28, 2016.

Famiglietti, J.S. and M. Rodell. 2013. Environmental science. Water in the balance. *Science* 340 (6138):1300-1301. DOI: 10.1126/science.1236460. Available at http://science.sciencemag.org/content/340/6138/1300. Accessed June 28, 2016.

Ficklin, D.L., E. Luedeling, and M.H. Zhang. 2010. Sensitivity of groundwater recharge under irrigated agriculture to changes in climate, CO2 concentrations and canopy structure. *Agricultural Water Management* 97(7): 1039-1050. DOI: 10.1016/j.agwat.2010.02.009. Accessed June 28, 2016.

Fritze, H., I.T. Stewart, and E. Pebesma. 2011. Shifts in western North American snowmelt runoff regimes for the recent warm decades. *Journal of Hydrometeorology* 12: 989-1006. DOI: 10.1175/2011JHM1360.1. Accessed June 28, 2016.

Goulden, M.L. and R.C. Bales. 2014. Mountain runoff vulnerability to increased evapotranspiration with vegetation expansion. Proceedings of the National Academy of Sciences 111(39): 14071-14075. DOI: 10.1073/pnas.1319316111. Accessed June 28, 2016.

Griffin, D. and K.J. Anchukaitis. 2014. How unusual is the 2012–2014 California drought? *Geophysical Research Letters* 41(24): 9017-9023. DOI: 10.1002/2014GL062433. Accessed June 28, 2016.

Haim, D., M. Shechter, and P. Berliner. 2008. Assessing the impact of climate change on representative field crops in Israeli agriculture: A case study of wheat and cotton. *Climatic Change* 86(3): 425-440. DOI: 10.1007/s10584-007-9304-x. Accessed June 28, 2016.

Hall, D.K., C.J. Crawford, N.E. DiGirolamo, G.A. Riggs, and J.L. Foster. 2015. Detection of earlier snowmelt in the Wind River Range, Wyoming, using Landsat imagery, 1972-2013. *Remote Sensing of Environment* 162: 45-54. DOI: 10.1016/j.rse.2015.01.032. Accessed June 28, 2016.

Hamlet, A.F., P.W. Mote, M. Clark, and D. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. *Journal of Climate* 18: 4545-4559.

Harter, T. 2015. California's agricultural regions gear up to actively manage groundwater use and

protection. *California Agriculture* 69(3): 193-201. DOI: 10.3733/ca.E.v069n03p193. Accessed June 28, 2016.

Hay, L.E., S.L. Markstrom, and C. Ward-Garrison. 2011. Watershed-scale response to climate change through the twenty-first century for selected basins across the United States. *Earth Interactions* 15(17): 1-37. DOI: 10.1175/2010EI370.1. Accessed June 28, 2016.

Hoerling, M.P., J.K. Eischeid, X.-W. Quan, H.F. Diaz, R.S. Webb, R.M. Dole, and D.R. Easterling. 2012. Is a transition to semipermanent drought conditions imminent in the U.S. Great Plains? *Journal of Climate* 25(24): 8380-8386. DOI: 10.1175/JCLI-D-12-00449.1. Accessed June 28, 2016.

Houborg, R., M. Rodell, B. Li, R. Reichle, and B.F. Zaitchik. 2012. Drought indicators based on model-assimilated Gravity Recovery and Climate Experiment (GRACE) terrestrial water storage observations. *Water Resources Research* 48(7). DOI: 10.1029/ 2011WR011291. Accessed June 28, 2016.

Jung, M., M. Reichstein, P. Ciais, S.I. Seneviratne, J. Sheffield, M.L. Goulden, G. Bonan, A. Cescatti, J. Chen, R. de Jeu, A.J. Dolman, W. Eugster, D. Gerten, D. Gianelle, N. Gobron, J. Heinke, J. Kimball, B.E. Law, L. Montagnani, Q. Mu, B. Mueller, K. Oleson, D. Papale, A.D. Richardson, O. Roupsard, S. Running, E. Tomelleri, N. Viovy, U. Weber, C. Williams, E. Wood, S. Zaehle, and K. Zhang. 2010. Recent decline in the global land evapotranspiration trend due to limited moisture supply. *Nature* 467(7318): 951-954. DOI: 10.1038/nature09396. Accessed June 28, 2016.

Konikow, L.F. 2013. Groundwater Depletion in the United States (1900-2008). In: Scientific Investigations Report 2013-5079: USGS. Available at http://pubs.usgs.gov/sir/2013/5079/. Accessed June 28, 2016.

Lundquist, J.D. and A.L. Flint. 2006. Onset of snowmelt and streamflow in 2004 in the western United States: How shading may affect spring streamflow timing in a warmer world. *Journal of Hydrometeorology* 7(6): 1199-1217. DOI: 10.1175/JHM539.1. Accessed June 28, 2016.

Mankin, J.S. and N.S. Diffenbaugh. 2015. Influence of temperature and precipitation variability on near-term snow trends. *Climate Dynamics* 45(3-4): 1099-1116. DOI: 10.1007/s00382-014-2357-4. Accessed June 28, 2016.

Mann, M.E. and P.H. Gleick. 2015. Climate change and California drought in the 21st century. Proceedings of the National Academy of Sciences 112(13): 3858-3859. DOI: 10.1073/pnas.1503667112. Accessed June 28, 2016.

Marshall, E., M. Aillery, S. Malcolm, and R. Williams. 2015. Agricultural production under climate change: The potential impacts of shifting regional water balances in the United States. *American Journal of Agricultural Economics* 97(2): 568-588. DOI: 10.1093/ajae/aau122. Accessed June 28, 2016.

Meixner, T., A.H. Manning, D.A. Stonestrom, D.M. Allen, H. Ajami, K.W. Blasch, A.E. Brookfield, C.L. Castro, J.F. Clark, D.J. Gochis, A.L. Flint, K.L. Neff, R. Niraula, M. Rodell, B.R. Scanlon, K. Singha, and M.A. Walvoord. 2016. Implications of projected climate change for groundwater recharge in the western United States. *Journal of Hydrology* 534: 124-138. DOI: 10.1016/j.jhydrol.2015.12.027. Accessed June 28, 2016.

Mote, P. W. 2006. Climate-driven variability and trends in mountain snowpack in western North America. *Journal of Climate* 19(23): 6209-6220. DOI:10.1175/JCLI3971.1. Accessed July 15, 2016.

NASA. 2015. Gravity Revocery and Climate Experiment. Available at http://science.nasa.gov/missions/grace/. Accessed June 28, 2016.

NASS (National Agricultural Statistics Service). 2015. Farm and Ranch Irrigation Survey. USDA Census of Agriculture, Last Modified 11/13/2014. Available at http://www.agcensus.usda.gov/Publications/Irrigation_Survey/. Accessed June 28, 2016.

NOAA. 2015. Palmer Drought Severity Index Data. Available at http://www.ncdc.noaa.gov/temp-and-precip/drought/historical-palmers/. Accessed June 28, 2016.

Painter, T.H., A.P. Barrett, C.C. Landry, J.C. Neff, M.P. Cassidy, C.R. Lawrence, K.E. McBride, and G.L. Farmer. 2007. Impact of disturbed desert soils on duration of mountain snow cover. *Geophysical Research Letters* 34(12): L12502. DOI: 10.1029/2007GL030284. Accessed June 28, 2016.

Prein, A.F., G.J. Holland, R.M. Rasmussen, M.P. Clark, and M.R. Tye. 2016. Running dry: The U.S. Southwest's drift into a drier climate state. *Geophysical Research Letters*. DOI: 10.1002/2015GL066727. Accessed June 28, 2016.

Rango, A. 2006. The real water supply for the Rio Grande Basin. *New Mexico Journal of Science* 44: 99-118.

Regonda, S.K., B. Rajagopalan, M. Clark, and J. Pitlick. 2005. Seasonal cycle shifts in hydroclimatology

over the western United States. *Journal of Climate* 18: 372-384.

Roos, M. 1991. A trend of decreasing snowmelt runoff in northern California. In: 59th Annual Western Snow Conference Proceedings, Juneau, AK, pp. 29-36.

Scanlon, B.R., Z. Zhang, R.C. Reedy, D.R. Pool, H. Save, D. Long, J. Chen, D.M. Wolock, B.D. Conway, and D. Winester. 2015. Hydrologic implications of GRACE satellite data in the Colorado River Basin. *Water Resources Research* 51(12): 9891-9903. DOI: 10.1002/2015WR018090. Accessed June 28, 2016.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H.P. Huang, N. Harnik, A. Leetmaa, N.C. Lau, C. Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316(5828): 1181-1184. DOI: 10.1126/science.1139601. Accessed June 28, 2016.

Serreze, M.C., M. Clark, R.L. Armstrong, D.A. McGinnis, and R. Pulwarty. 1999. Characteristics of the western United States snowpack from snowpack telemetry (SNOTEL) data. *Water Resources Research* 35(7): 2145-2160.

Sevruk, B. 1985. Correction of Precipitation Measurements: Summary Report. In: Proceedings of the International Workshop on the Correction of Precipitation Measurements, Zurich, Switzerland, 1-3 April 1985, WMO/TD No. 104, Geneva.

Sevruk, B. 1989. Reliability of Precipitation Measurement. In: Proceedings of the WMO/IAHS/ETH Workshop on Precipitation Measurement, St. Moritz, Switzerland.

State of California. 2016. California Groundwater: Sustainable Groundwater Management Act. Available at http://www.water.ca.gov/cagroundwater/legislation.cfm. Accessed June 28, 2016.

Stewart, I.T., D. Cayan, and M. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. *Journal of Climate* 18: 1136-1155.

Taylor, R.G., B. Scanlon, P. Doll, M. Rodell, R. van Beek, Y. Wada, L. Longuevergne, M. Leblanc, J.S. Famiglietti, M. Edmunds, L. Konikow, T.R. Green, J.Y. Chen, M. Taniguchi, M.F.P. Bierkens, A. MacDonald, Y. Fan, R.M. Maxwell, Y. Yechieli, J.J. Gurdak, D.M. Allen, M. Shamsudduha, K. Hiscock, P.J.F. Yeh, I. Holman, and H. Treidel. 2013. Ground water and climate change. *Nature Climate Change* 3(4): 322-329. DOI: 10.1038/nclimate1744. Accessed June 28, 2016.

U.S. Geological Survey. 2016. USGS Water Use Data for the Nation. Available at http://waterdata.usgs.gov/nwis/wu. Accessed June 28, 2016.

USDA - ERS. 2015. Agricultural Productivity in the U.S. Available at http://www.ers.usda.gov/data-products/agricultural-productivity-in-the-us.aspx#28247. Accessed June 28, 2016.

Wahl, K.L. 1991. Is April to July Runoff Really Decreasing in the Western United States? In: Proceedings of the 59th Annual Western Snow Conference, Juneau, AK.

Walsh, J., D. Wuebbles, K. Hayhoe, J. Kossin, K. Kunkel, G. Stephens, P. Thorne, R. Vose, M. Wehner, J. Willis, D. Anderson, S. Doney, R. Feely, P. Hennon, V. Kharin, T. Knutson, F. Landerer, T. Lenton, J. Kennedy, and R. Somerville. 2014. Ch. 2: Our changing climate. In: *Climate Change Impacts in the United States: The Third National Climate Assessment*, J.M. Melillo, T.C. Richmond, and G.W. Yohr (Eds.). U.S. Global Change Research Program, pp. 19-67.

Woodhouse, C.A., G.T. Pederson, K. Morino, S.A. McAfee, and G.J. McCabe. 2016. Increasing influence of air temperature on upper Colorado River streamflow. *Geophysical Research Letters* 43(5): 2174-2181. DOI: 10.1002/2015GL067613. Accessed June 28, 2016.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 62-77, August 2016

Irrigation Technology Choice as Adaptation to Climate Change in the Western United States

*George Frisvold and Ting Bai

Department of Agricultural & Resource Economics, University of Arizona, Tucson, AZ
**Corresponding Author*

Abstract: Several commentators have suggested adoption of improved irrigation will be important for agricultural adaptation to climate change. This study used data from a special tabulation of the United States Department of Agriculture (USDA) Farm and Ranch Irrigation Survey to examine how climate and other factors influence the choice of sprinkler versus gravity-flow irrigation across 17 western states. Because studies of irrigation technology adoption often concentrate on small geographic areas with the same climate, few have estimated effects of climate on irrigation technology choice. Results suggest sprinkler irrigation is a more promising climate adaptation in areas that are relatively cooler and where extreme precipitation events increase. Sprinkler adoption is less likely in warmer climates and under drier climate change scenarios. An implication is that for hot, arid parts of the West, other climate adaptation strategies may be more important to pursue. Consistent with several previous studies, sprinkler adoption was greater among farms that were larger, had higher water costs, and relied more on groundwater. Sprinkler adoption was also greater in states with a higher erosion index that captured effects of rainfall, field slope, and soil water-holding capacity, which also have been found to be important adoption predictors in a number of previous studies.

Keywords: *irrigation, climate change, technology adoption, farm size, sprinkler, gravity-flow*

Improved irrigation efficiency has been cited as an important way to adapt to climate change (Burton 2000; Smit and Skinner 2002; Kurukulasuriya and Rosenthal 2003; Cavagnaro et al. 2006; Jackson et al. 2009; Joyce et al. 2009). Compared to gravity irrigation, drip or sprinkler irrigation can achieve better control over the timing and level of water applied to crops. This can better match water applications to plant requirements. Improved irrigation timing can help protect crops from weather extremes such as drought stress or frost. Yet, drip or sprinkler irrigation can require large capital costs. Irrigators often cite financial constraints as major barriers to investing in improved irrigation efficiency (Frisvold and Deva 2012).

Irrigation Technology Choice as Climate Adaptation

Researchers have thoroughly studied factors affecting adoption of improved irrigation technology. Yet, relatively few studies have included climate variables in empirical analyses. Studies of irrigation technology adoption have often concentrated on small geographic areas, such as a single irrigation district. The geographic scope of such analyses can be too narrow to measure effects of climate. Long-term climate averages change little over time and localized studies may have insufficient variation in climate to allow for econometric analysis. Put simply, over too narrow a geographic range, potential climate variables do not vary. Narrow geographic scope can also lead to multi-collinearity between climate variables, limiting the number of variables that can be assessed (Fleischer et al. 2008).

This study adds to a small list of studies that have focused on the role of climate in adoption of irrigation technology. Negri et al. (2005) examined the effect of climate on the decision whether or not to irrigate. Negri and Brooks (1990) considered effects of climate on choice of irrigation technology among irrigators. Mendelsohn and Dinar (2003) examined climate effects on the share of acres that were

irrigated with gravity, sprinkler, or drip systems across U.S. counties. Caswell et al. (2001) also examined climate effects on the choice of whether to irrigate, along with the choice of irrigation method for individual producers. Frisvold and Deva (2013) examined how climate affects the choice between sprinkler and gravity-flow technologies using farm-size stratified, state-level data. These studies found climate variables had important impacts on irrigation adoption decisions. Dinar et al. (1992) found that temperature and rainfall variables were not significant, but the study was limited to California's San Joaquin Valley and coefficients of variation for climate variables were quite low. More recently, Moreno and Sunding (2005) and Schoengold et al. (2006) found the number of frost-free days was important for explaining joint crop-irrigation technology choices. Olen et al. (2016), using farm-level data for three West Coast states, found that growers use sprinklers to mitigate risk of crop damage from extreme weather.

Here, we consider how climate, farm size, water costs, labor costs, and soil characteristics affect irrigator choice between gravity flow and sprinkler irrigation in 17 western states. To make sure there is enough spatial variation in climate variables, we use data from special cross-tabulations of the Farm and Ranch Irrigation Survey (FRIS) developed by USDA's Economic Research Service (USDA ERS 2016). While the USDA conducts the national FRIS every five years, published tables report state-level aggregate data, focusing on 2 x 2 relationships. This data configuration is not amenable to multivariate analysis. ERS's Special Tabulation, however, reports data for the 1998 and 2008 FRIS, stratifying observations by four farm sales classes for the 17 westernmost contiguous U.S. states. This provides 136 state-farm size pairs with sufficient geographic scope to conduct a regional analysis of irrigation technology adoption.

The study begins with a discussion of the importance of irrigation to western agriculture. Next, we compare the salient features of sprinkler and gravity flow irrigation and discuss trends in the diffusion of sprinkler technology. We then review the literature on sprinkler irrigation adoption, which has examined the role of soil characteristics (especially those affecting water-holding capacity of soils), water and labor costs, farm size, and climate as important factors influencing adoption of modern irrigation technologies. The next section introduces a regression specification and data sources. We then analyze the results of the regression and conclude by discussing the implications of our main findings.

Western Irrigated Agriculture

Irrigation is enormously important to western agriculture. In the 17 westernmost contiguous states, about 75% of the value of crops grown comes from the 25% of the cropland that is irrigated (Gollehon and Quinby 2000). Improved irrigation technology has been seen as a necessary, if not sufficient, part of relieving pressure on water supplies and quality in the West. Improved technologies such as sprinkler or drip irrigation allow producers better control over water applications and increase the share of applied water that is taken up by plants. Such improved technologies have been seen as means to improve farm yields and incomes (Caswell and Zilberman 1986; Aillery and Gollehon 2003), allow producers to better adapt to drought or climate change (Mendelsohn and Dinar 2003; Schuck et al. 2005), and reduce water pollution from soil erosion and chemical leaching (Caswell et al. 1990, 2001; Dressing 2003).

Irrigation Technology

Traditional irrigation systems use gravity to distribute water. Water is conveyed to the field using open ditches or pipes, and then released along the upper end of the field. Furrows control water movement and channel the flow down or across the field. Gravity systems are best suited to soils with higher moisture-holding capacities and relatively flat fields to prevent excessive water runoff (Negri and Brooks 1990; Negri et al. 2005).

Negri and Brooks (1990, 214) noted, "Sprinkler irrigation technologies save water relative to gravity-flow systems by distributing water evenly on the field, reducing percolation below the root zone, and eliminating field runoff." Sprinkler systems can be used on steeper slopes unsuited to gravity systems and on soils with lower water-holding capacity. Sprinkler irrigation can have much higher application efficiencies than traditional gravity irrigation (Sloggett 1985). While gravity

systems have application efficiencies usually ranging from 40-65%, efficiencies from sprinkler systems typically range from 75-85% (Aillery and Gollehon 2003). Beyond critical temperature thresholds, advantages of sprinkler systems may be negated, however, by large evaporation losses (Olen et al. 2016). Evaporation losses from heat can be exacerbated by windy conditions (Negri and Hanchar 1989).

In addition, on-farm application efficiency is not the same as system-wide efficiency. Although more applied water may be taken up by the crop in sprinkler systems, this means that there is less water available as return flows or to aquifers through groundwater recharge. Thus, at a system level, more "efficient" systems can actually increase consumptive use of water (Ward and Pulido-Velazquez 2008). Greater off-farm flows may have benefits of, "diluting agricultural chemicals or septic tank leachate in shallow groundwater, providing groundwater recharge to shallow wells, and providing delayed return flow to the stream thus maintaining in-stream flow after peak runoff periods" (Fernald and Guldan 2006, 79). Yet, in other cases, sprinkler systems can lessen production of contaminated drainage water (Caswell 1991).

Sprinkler systems have higher capital costs than gravity systems. This may act as a barrier to adoption for farms below some critical size. Sprinkler systems tend to be energy-using and labor-saving relative to gravity systems. Thus, their relative profitability will depend on labor and energy costs. The social as well as private benefits of sprinkler versus gravity flow systems will depend on a complex combination of economic, agronomic, and climate factors.

U.S. acreage irrigated with sprinkler systems increased from 36% to 54% between 1979 and 2008. Acreage devoted to drip irrigation rose from 1% to 7%, while acreage irrigated by gravity systems declined from 63% to 39% over the same period (Table 1). Drip irrigation is most commonly used for vegetables, orchards, vineyards, nuts, and other perennial crops. In the 17 Western States, 81% of acreage under drip irrigation is in California, primarily on citrus and specialty crops. While drip irrigation accounts for 32% of California's irrigated acreage, it accounts for just 1.6% of irrigated acreage in the remaining 16 states (USDA, NASS 2010). Because of limited adoption of drip systems outside of California, we focus here on the choice between sprinkler and gravity systems.

Table 1. Shares of irrigated acreage in the United States by method of irrigation.

Year	Agency Report	Gravity	Sprinkler	Drip and Subsurface
1979	NASS	63%	36%	1%
1995	USGS	52%	40%	3%
1998	NASS	50%	45%	5%
2000	USGS	48%	46%	7%
2003	NASS	43%	51%	6%
2005	USGS	44%	50%	7%
2008	NASS	39%	54%	7%

Sources: USDA NASS (1998, 2004, 2010); Aillery and Gollehon (2003); Solley et al. (1998); Hutson et al. (2004); Kenny et al. (2009).

Irrigation Technology Choice: The Literature

Caswell and Zilberman (1986) introduced a theoretical model of irrigation technology choice that characterized modern irrigation systems (sprinkler and drip systems) as land quality augmenting. Negri and Brooks (1990, 214) noted, "traditional gravity systems applied to land with high water-holding capacity due to high clay content and level slopes can achieve application efficiencies comparable to sprinkler irrigation. Conversely, lands with porous soils or steep slopes are unsuitable for gravity irrigation because of excessive deep percolation or runoff."

Because modern systems compensate for soils with poor water-holding capacity, improved irrigation technologies would be relatively more profitable on such soils. Subsequent empirical studies supported Caswell and Zilberman's theoretical specification (Dinar and Yaron 1990; Negri and Brooks 1990; Dinar et al. 1992; Green and Sunding 1997; Schuck and Green 2001; Mendelsohn and Dinar 2003).

Economic theory and previous empirical findings suggest there are systematic relationships between operation scale and irrigation methods. Leib et al. (2002) found significant positive relationships between farm size and adoption of scientific irrigation scheduling methods (use of crop evapotranspiration data and soil moisture testing) among Washington farmers. In a study of New Mexico irrigators, Skaggs and Samani (2005, 43) reported a "lack of interest in making improvements to current irrigation systems or methods on the smallest farms." Comparing irrigation districts in Alberta, Canada, Bjornlund et al. (2009) found evidence of greater adoption of information-intensive irrigation management in areas with larger farms. Frisvold and Deva (2012, 569) found that smaller farms in the U.S. West were, "less likely to investigate irrigation improvements, use management-intensive methods for irrigation scheduling, or participate in cost-share programs to encourage adoption of improved irrigation practices."

Sprinkler systems require fixed capital investments, so average fixed costs fall with scale of operation. Evidence of the impact of farm size on sprinkler adoption is mixed, however. Skaggs (2001) and Schuck et al. (2005) found evidence of a positive relationship between farm size and adoption by New Mexico and Colorado irrigators, but Negri and Brooks (1990) reported the opposite result in a large sample of western groundwater-using farms from the 1984 FRIS. Frisvold and Deva (2013), in contrast, found a positive relationship between sprinkler adoption and farm size using 1998 FRIS data. Green et al. (1996), Green and Sunding (1997), and Dinar et al. (1992) observed positive relationships between sprinkler adoption and farm field size.

Different ways that researchers measure scale of operation complicates comparisons of studies considering farm size and irrigation technology adoption. Some measure scale in terms of sales volume. In addition to areal scale, this may be also capturing the effects of growing high-value specialty crops. Thus, cropping decisions may drive technology choices. Analyses that focus on total farm area may be, in contrast, capturing the effects of low-value, land extensive crops (e.g., irrigated pasture). There is a distinction between farm scale and field scale. Technological factors may account for economies of scale at the field level. Scale of farm operation may affect risk aversion or access to credit.

A number of studies have examined the role of water cost on sprinkler adoption. Caswell and Zilberman (1985), Negri and Brooks (1990), and Dinar et al. (1992) found a positive relationship between water costs and sprinkler adoption. Green et al. (1996) reported that water price had a negative but statistically insignificant effect, whereas Green and Sunding (1997) found water price had a positive effect on sprinkler adoption in citrus production, but not in vineyards.

Empirical findings for the relationship between water source and technology choice have been mixed. According to Negri and Brooks (1990), greater reliance on surface water decreased the probability of sprinkler adoption. Moreno and Sunding (2005) found such a negative relationship to be statistically significant. Caswell and Zilberman (1985), Dinar et al. (1992), and Green et al. (1996) found a negative, but statistically insignificant relationship between reliance on surface water and sprinkler adoption. Mendelsohn

and Dinar (2003), in contrast, found a significant, positive relationship between reliance on surface water and sprinkler adoption.

The number of studies on the effects of climate variables is limited because smaller-scale studies at the irrigation district level (for example) may not exhibit sufficient cross-sectional variation in climate. Studies covering wider geographic areas found climate to have important effects on irrigation technology choice. Both Negri and Brooks (1990) and Mendelsohn and Dinar (2003) found higher rates of sprinkler adoption in areas with greater rainfall. Dinar et al. (1992) did not find a significant rainfall effect, but this analysis was confined to California's San Joaquin Valley, which has less variation in rainfall than the other two studies. Negri and Brooks (1990) argued that sprinkler adoption would be greater in areas with greater rainfall. This is because growers in high-rainfall areas face a greater risk of crop damage from unexpected rainfall following flood irrigation. Here, sprinklers provide growers with greater control over applications. In contrast, in hot, arid regions, evaporation losses are large with sprinkler technology. Evaporation losses in sprinkler systems can reach levels close to 50% under the hot, arid conditions found in Arizona or Southern California (McLean et al. 2000). Negri and Hanchar (1989, 9) stated, "Farmers in hot or windy regions are more likely to adopt gravity since a large fraction of water applied with sprinkler systems evaporates under these climate conditions." Based on farm-level analysis of California, Oregon, and Washington irrigators, Olen et al. (2016) argue that above a critical temperature threshold, high evaporative losses from sprinklers negate efficiency advantages of sprinklers, making them less attractive relative to gravity or drip systems. Mendelsohn and Dinar (2003, 338) also pointed out the problem of large evaporation losses and noted "sprinkler systems are more frequently adopted in cooler locations with a lot more rainfall."

Negri and Brooks (1990) reported that sprinkler adoption was lower in areas with more frost-free days and a longer growing season (measured in growing degree days) and greater in areas with more frost. They argued that sprinklers are better suited for irrigation for frost protection, whereas longer growing seasons are associated with warmer climates where sprinkler evaporation losses are greater. Olen et al. (2016) cited several studies reporting the frost protection advantages of sprinkler irrigation over other systems. Mendelsohn and Dinar (2003) also pointed out that sprinkler adoption rates were inversely related to temperature. Dinar et al. (1992) again found no significant effect, but as before, this may be because of the low variance of weather and climate variables in their sample.

Finally, because sprinkler systems tend to be labor saving, some studies have considered impacts of labor costs on sprinkler adoption. Results have been mixed. Negri and Brooks (1990) found that higher farm labor wages were associated with greater adoption of sprinkler irrigation. Mendelsohn and Dinar (2003) reported greater use of sprinklers in counties with higher farm wage rates in January, but lower wage rates in April. Frisvold and Deva (2013) found no significant relationship between current farm wages and sprinkler adoption, but found growth in wage rates over the previous five years was associated with higher sprinkler adoption rates.

Data and Regression Model Specification

Irrigation data come from special cross-tabulations of the 1998 and 2008 Farm and Ranch Irrigation Survey (FRIS) (USDA, NASS, 1998; 2010) reported by USDA's Economic Research Service (USDA ERS 2016). The FRIS collects the most detailed, comprehensive data on irrigation practices and water use at the national and state level in the United States. Yet, survey data are seldom used for statistical analysis of irrigator behavior. This is because USDA reports data as tables of state-level aggregates, providing far too few observations for multivariate analysis. To protect respondent confidentiality, access to farm-level FRIS data is restricted. For this reason, rigorous research using FRIS data has been limited to a few studies conducted by USDA economists (and collaborators) (e.g., Negri and Brooks 1990). A notable exception to this rule is a recent study by Olen et al. (2016) that analyzed farm-level FRIS data for >1,000 irrigators in California, Oregon, and Washington. This study has sufficient

geographic scope to assess effects of climate and irrigator behavior in detail.

The ERS special cross-tabs report data for the 17 western contiguous states (North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, Texas, Montana, Wyoming, Colorado, New Mexico, Idaho, Utah, Arizona, Nevada, Washington, Oregon, and California) by four farm sales classes:

- small farms, with sales less than $100,000;
- medium farms, with sales from $100,000 to $249,999;
- large farms, with sales from $250,000 to $499,999; and
- very large farms, with sales of $500,000 or greater.

Let the proportion of hectares irrigated with sprinkler systems (as a share of acreage under sprinkler plus gravity irrigation) by irrigators in sales class i and state j be PS_{ij} while the proportion of hectares irrigated with gravity systems be PG_{ij}. Drip and subsurface irrigation acreage were not included, as data were not reported for many states and farm sales classes because of insufficient observations. The proportion of hectares irrigated with sprinklers as a share of hectares irrigated with sprinklers and gravity systems ($PS/(PS + PG)$) has grown for all farm size classes between 1998 and 2008 (Figure 1). Following previous work on irrigation adoption using proportions data (Caswell and Zilberman 1985; Mendelsohn and Dinar 2003; Schaible et al. 1991), the regression equation explaining adoption is specified as a logistic function:

$$ln\ (PS_{ij}/(PG_{ij})) = \alpha_0 + \alpha_1 \text{Small} + \alpha_2 \text{Medium} + \alpha_3 \text{Large} + \boldsymbol{\beta}' \boldsymbol{X}_{ij} + u_{ij} \quad (1)$$

where:
Small = 1 for small farms, = 0 otherwise;
Medium = 1 for medium farms, = 0 otherwise;
Large = 1 for large farms, = 0 otherwise;
$\boldsymbol{X}_{ij}$ = a vector of other explanatory variables (discussed below); and
u_{ij} = a stochastic error term.

A regression intercept α_0 is included; so, one farm class dummy (for very large farms) is omitted. Coefficients for the other sales-class variables represent differences from the very large



Figure 1. Percentage of hectares irrigated with sprinklers. Weighted average from 17 western states by farm sales class (Small: $0 to $99,999 in sales; Medium: $100,000 to $249,999; Large: $250,000 to $499,999; Very Large: $500,000 and greater). Source: USDA, ERS (2016).

farm class. Operations may be in the larger sales classes if they have more acreage, grow higher-value crops, or both. The sales class variables may therefore capture some combination of acreage and crop value effects. Table 2 shows descriptive statistics for regression variables.

The variable *ln Water Cost* is measured as (the natural log of) water cost per acre-foot applied. An acre-foot is volume of a sheet of water one acre (0.405 hectare) in area and one foot (30.48 cm) in depth. It equals 1,233.5 m^3 of water. The per-unit-of-volume water cost is a weighted average that combines the costs of pumping surface water and groundwater as well as prices paid for off-farm delivered surface water.

The variable *Surface Water* is the share of surface water to total irrigation water applied by farm size and state, for 1998 and 2008 FRIS irrigated farms. Irrigators' relative reliance may affect sprinkler adoption in at least two ways. First, marginal costs of surface water are usually lower than costs for groundwater. Previous research suggests that sprinkler adoption increases with water costs. Second, surface water diversions naturally complement gravity flow systems.

We include two climate variables. *LNTEMP5-9* is (the log of) monthly 40-year mean temperatures in °F weighted by cropland from May through September. *LNTEMP10-3* is (the log of) the average of monthly 40-year mean temperatures of cropland from October through March, adjusted for growing season. Sprinkler irrigation is often used to protect crops from frost (Negri and Brooks 1990; Skaggs 2001; Dressing 2003; Moreno and Sunding 2005; Olen et al. 2016). *LNTEMP10-3* is intended to measure such frost risk. Winter months with average temperatures below 32 °F (0 °C) are not included in *LNTEMP10-3.* Temperatures in months of freezing temperatures represent the off-season and are not likely to affect this decision. Above freezing, but low Fall/Winter temperatures suggest areas where frost risk is greater, and sprinkler adoption is higher. We hypothesize an inverse relationship between *LNTEMP10-3* and frost-protecting uses of sprinkler irrigation.

The climate variables come from Teigen and Singer (1988), who weighted average weather station measurements by harvested cropland. The climate variables give more weight to temperature readings where crops are grown. For example, readings from Death Valley or high in the Rocky Mountains with no agricultural production would receive no weight. In contrast, readings from major agricultural production areas would receive great weight.

The variable *LNEROSION* measures average annual sheet and rill erosion on cultivated cropland in each state, measured in tons of soil per acre per year. Data come from the 1997 Natural Resources Inventory of the USDA's Natural Resources Conservation Service (USDA NRCS 2000). According to the NRCS (Potter et al. 2006, 73), "Sheet erosion takes place when the dislodged soil particles are moved by thin sheets of water flowing over the surface. Rill erosion occurs when the surface flow of water establishes paths and the flowing water detaches soil particles from the sides and bottoms of the rills that are formed." *LNEROSION* is derived from the universal soil loss equation (USLE) (Wischmeier and Smith 1978; IWR-MSU 2002). It is included because it comprises several variables that past studies have found affect irrigation technology choice.

LNEROSION = ln(RKLSCP), where *R* is a rainfall factor, *K* is a soil erodibility factor, *L* is a slope length factor, *S* is a slope steepness factor, *C* is a cover and management factor, and *P* is a conservation practice factor. *R* is a factor measuring erosivity of soil from rainfall runoff that increases with the total amount and peak intensity of rainfall. The USLE's *R* factor is highly correlated with measures of precipitation (with $R^2 \sim 0.86–0.91$) (Yu and Rosewell 1996; de Santos Loureio and Azevedo Coutinho 2001; Diodato 2004). The USLE measure of erosion is more sensitive to changes in the *R* factor than to other environmental variables (Nearing 2001).

The factor *K* assigns values to different types of soil based on susceptibility to erosion and rate of runoff. *K* is thus related to the water-holding capacity of soils. Sprinkler irrigation technology enhances water-holding capacity and previous studies have found higher adoption rates on soils with low water-holding capacity. For example, clay soils have *K* values ranging from about 0.05 to 0.15 because they resist detachment. Loam soils tend to have moderate *K* values, and sandy soils have higher *K* values. Both Negri and

Table 2. Descriptive statistics for variables used in Irrigation Choice Regression.

Variable Name	Variable Description	Mean	Minimum	Maximum
$PS_{ij}/(PS_{ij} + PG_{ij})$	Acreage irrigated with sprinkler systems as a proportion of the sum of area to irrigate with sprinkler and gravity systems	0.53	0.02	0.95
Ln (PS_{ij} / PG_{ij})	Ln of the proportion hectares irrigated with sprinklers over the proportion of hectares irrigated with gravity methods	0.14	-3.95	2.92
Ln Water Costs	Ln of average water cost, $ per acre-foot (AF) applied	$3.01	$1.32	$4.34
	Antilog of ln water costs ($ / AF)	$20.22	$3.76	$77.00
	Antilog of ln water costs ($ / thousand cubic meters)	$16.39	$3.05	$62.42
Surface Water	Proportion of surface water to total irrigation water from all sources	0.578	0.000	0.996
LNTEMP5-9	Ln of average of long-run monthly temperatures from May to September	4.207	4.107	4.413
	Antilog of *LNTEMP5-9* (F)	67.2	60.8	82.5
	Antilog of *LNTEMP5-9* (C)	19.6	16.0	28.1
LNTEMP10-3	Ln of growing season adjusted average of long-run month temperatures from October to March	3.78	3.66	4.04
	Antilog of *LNTEMP10-3* (F)	43.9	38.8	57.0
	Antilog of *LNTEMP10-3* (C)	6.7	3.8	13.9
LNEROSION	Ln of average annual sheet and rill erosion on nonfederal land	0.48	-1.61	1.55
	Antilog of *LNEROSION*	1.61	0.20	4.70
ΔLNWAGE	Continuous growth rate of farm wages	3.01	1.32	4.34

Brooks (1990) and Mendelsohn and Dinar (2003) reported that farming on sandy soils encouraged sprinkler adoption, while farming on clay soils discouraged sprinkler adoption, relative to farming on loam soils. These results suggest that sprinkler adoption would increase in *LNEROSION*, via the relationship with *K*.

Land with steeper slopes has been associated with greater adoption of sprinkler irrigation relative to gravity irrigation. Mendelsohn and Dinar (2003), however, argued that slope length should be positively associated with gravity irrigation because it implies flatter fields. They found empirical evidence to support this argument. Slope length appears to be the only variable that increases *LNEROSION* and has been associated with less sprinkler irrigation use. All other components of the USLE appear to both increase *LNEROSION* and contribute to greater sprinkler adoption, so slope length may have a confounding effect. Yet, computed values for *LNEROSION* are not very sensitive to slope length, particularly on flat landscapes (IWR-MSU 2002). Overall, then, one might expect a positive association between *LNEROSION* and sprinkler adoption.

Several studies suggest gravity systems are more labor intensive than sprinkler systems (Maddigan et al. 1982; Bernardo et al. 1987; Negri and Hanchar 1989; Negri and Brooks 1990; Sauer et al. 2010). Sprinkler systems may thus represent a laborsaving change from gravity systems. Following Frisvold and Deval (2013), we estimated the effects of growth rates in farm labor wages on sprinkler system adoption. The variable Δ*LNWAG*E was the log of hired farm labor wages minus the log of wages from five years earlier. This represents the continuous growth rate in hired labor wages from five years before both the 1998 and 2008 FRIS observations. Annual wage rates are reported by multi-state region rather than by state in the USDA's Quick Stats database (http://quickstats.nass.usda.gov/).

Regression Results

The regression equation was estimated using ordinary least squares (Table 3). With the log proportions transformation of the dependent variable, we failed to reject the null hypothesis of homoscedasticity. The coefficients for small, medium, and large farms were negative and statistically significant, indicating that these operations had a lower percentage of acreage under sprinkler systems than did very large farms. The coefficient for small farms was the most negative. The coefficients for farm size variables grow increasingly negative as farm size grows smaller. This suggests there is a positive, monotonic relationship between farm size and sprinkler adoption.

Water costs had a positive and significant effect on sprinkler adoption, consistent with earlier findings suggesting that sprinkler irrigation increases with water costs (Caswell and Zilberman 1985; Negri and Brooks 1990; Dinar et al. 1992; Green et al. 1996; Moreno and Sunding 2005). As noted above, the water cost variable may be measured with error because the measure captures average rather than marginal costs, irrigators are quantity constrained, or both. Our water cost variable may be incorrectly measuring the true water costs irrigators face. This "errors-in-variables" can lead to attenuation bias such that the coefficient estimate is biased downward. The measurement error presents a problem if the *lnWaterCost* variable is correlated with the regression error term. We performed a Hausman-Wu test as follows. First, *lnWaterCost* was regressed on the percentage reliance on surface water, farm size dummies, a year dummy, and fixed effects for USDA production regions (Northern Plains, Southern Plains, Southern Mountain, Northern Mountain, and Pacific Northwest, with California as the default). Next, residuals from this first stage regression were included as an additional variable in the irrigation adoption regression. The null hypothesis that the *lnWaterCost* variable is exogenouse is a test that the residuals are significantly different from zero. We could not reject the null hypothesis that the residuals coefficient is zero ($p = 0.768$). This suggests that *lnWaterCost* is not correlated with the adoption equation error term, biasing the OLS estimates.

The proportion of hectares irrigated with sprinklers decreased with greater reliance on surface water. Reliance on surface water appears to have a negative effect on sprinkler

Table 3. Factors affecting the adoption of sprinkler irrigation. Ordinary least squares* results.

Variable	Parameter Coefficient	Standard Error	P-value (significance level)
Intercept	22.789	3.518	0.000
D2008	0.274	0.113	0.017
Small	-0.667	0.158	0.000
Medium	-0.600	0.153	0.000
Large	-0.342	0.153	0.028
Surface Water	-1.936	0.332	0.000
LNTEMP5-9	-3.591	1.247	0.005
LNTEMP10-3	-2.083	0.917	0.025
LNEROSION	0.760	0.078	0.000
Ln Water Cost	0.448	0.155	0.004
ΔLNWAGE	0.184	1.205	0.879

Dependent variable: *ln* [(*PSij/PGij*)] where *PSij* is the proportion of acreage irrigated with sprinkler systems and *PGij* as a proportion of acreage irrigated with gravity systems.
Adjusted R^2 = 0.754. Number of observations = 136.
*The null hypothesis of homoscedasticity could not be rejected.

adoption beyond differences in water pumping costs (pumping surface water costs less than groundwater). Negri and Brooks (1990) noted that marginal costs for water from surface sources can be quite low. Surface water supplies in the West are often quantity-rationed rather than price-rationed. In many western surface water projects, average costs can deviate substantially from marginal costs. Indeed, in many cases, irrigators are charged per hectare of irrigated land, rather than based on the volume of water applied. In these cases, the marginal cost of water (per volume of water) is zero. The fact that sprinkler irrigation adoption rates increase with reliance on groundwater in the regression may also reflect the lower *marginal* cost of surface water. Moreno and Sunding (2005) noted that costs of surface water are less variable than costs of groundwater, which can fluctuate based on changes in the water table and volatile energy costs. Thus, irrigators relying on groundwater may face both higher and more variable marginal costs of water. Both factors may encourage adoption of more water-efficient technologies.

There is a significant negative relationship between May-September temperatures and sprinkler adoption. We experimented with quadratic and log quadratic specifications of this temperature variable. In no case was the quadratic term significant, suggesting that sprinkler adoption is monotonically decreasing in May-September temperatures. Using the same procedure, it was also found that sprinkler adoption was monotonically decreasing in growing-season-adjusted Winter/Fall temperatures (*LNTEMP10-3*).

These results suggest that, in areas with warmer climates, there may be limited scope to adapt to

climate warming via adoption of sprinkler irrigation systems. In cooler climates, the relationship between climate warming and sprinkler adoption is more complex. The variable *LNTEMP10-3* does not include fall and winter months with average temperatures below 32 °F (0 °C). This is because one would not expect changes in off-season temperatures to affect irrigation decisions for the growing season. However, climate warming may convert an off-season month to a relatively cold growing-season month. If it does so, it is possible that this effect will *lower* the value of *LNTEMP10-3* and encourage sprinkler adoption. This is a mathematical explanation, but not an agronomic one. In agronomic terms, warming in cold climates increases the number of months where plant growth is possible, but simultaneously may increase frost risk in this expanded growth window. Whether climate warming has a positive effect on sprinkler adoption in colder states will depend on a complex combination of changes in the length of growing season, Fall/Winter temperatures, and Spring/Summer temperatures. Our results do suggest, however, that in warmer western states (Arizona, California, New Mexico, Oklahoma, Oregon, and Texas), warmer temperatures would discourage sprinkler adoption.

These results regarding temperature apply only to the choice of sprinkler irrigation relative to gravity irrigation (i.e., changes at the intensive margin). They say nothing about whether sprinkler adoption may encourage more land to be brought under irrigated cultivation. Caswell and Zilberman (1986) discuss conditions where introduction of sprinklers would increase total acreage under irrigation. Caswell et al. (2001) found evidence of warmer temperatures increasing sprinkler irrigation adoption, relative to dryland production. Some irrigators may also switch to drip irrigation in response to warmer temperatures (Mendelsohn and Dinar 2003; Olen et al. 2016). Other climate adaptations include switching the mix of crops grown and deficit irrigation (Moreno and Sunding 2005; Schoengold et al. 2006; Frisvold and Konyar 2012; Olen et al. 2016).

The coefficient on average annual sheet and rill erosion is positive and significant, indicating that in areas with high erosion, sprinkler irrigation systems are more likely to be adopted. Because sheet and rill erosion is more likely on areas with greater rainfall, steeper slopes, and poorer water-holding capacity, it is likely this variable is picking up the effects of combinations of these factors. The results here are consistent with previous research suggesting sprinkler adoption will be greater in areas with greater precipitation where it can supplement rainfall (Negri and Brooks 1990; Mendelsohn and Dinar 2003) and on lower quality soils (Caswell and Zilberman 1985; Negri and Brooks 1990; Mendelsohn and Dinar 2003). The wage growth rate variable was positive as in Frisvold and Deva (2013), but it was not statistically significant. We experimented with current wage levels and lagged levels. These were statistically insignificant as well.

Policy Implications and Conclusions

This study illustrated how special tabulations of the USDA Farm and Ranch Irrigation Survey (FRIS) can be used for multivariate regression analysis of irrigation technology adoption across 17 western states. First, results corroborated findings of a number of previous studies finding that adoption of sprinkler (versus gravity) irrigation increased with farm operation scale, water pumping and purchase costs, and reliance on groundwater. Sprinkler adoption was greater in states with a higher index of sheet and rill erosion, which captures effects of greater rainfall, steeper slopes, and soils with less water-holding capacity. Previous studies have found these individual factors to be important predictors of sprinkler adoption, so the erosion index may serve as a useful summary index of these effects in future research.

Second, the study took advantage of the West-wide scope of the FRIS data to examine the effects of climate on sprinkler adoption. Several commentators have suggested adoption of improved irrigation will be important for agricultural adaptation to climate change (e.g., Burton 2000; Smit and Skinner 2002; Kurukulasuriya and Rosenthal 2003; Cavagnaro et al. 2006; Jackson et al. 2009; Joyce et al. 2009). Sprinkler adoption is less likely in warmer climates and under drier climate change scenarios. An implication for climate adaptation policy is that for hot, arid parts of the West, other adaptation strategies – such as deficit irrigation, adoption

of drip irrigation, and use of advanced water management practices (e.g., soil- or plant-moisture sensing devices, commercial irrigation scheduling services, or computer-based crop-growth simulation models) – may be more important to pursue. Adoption of sprinkler irrigation increased with an index of sheet and rill erosion, which increases with precipitation amounts and intensity. Nearing (2001) projected that the rainfall component of the erosivity index would increase in some parts of the West under climate change, but decrease in other parts. The Third National Climate Assessment of the U.S. Global Change Research Program projects a decrease in spring and winter precipitation in the Southwest, which would reduce the erosion index (Walsh and Wuebbles 2013). However, the Assessment also projects an increase in the frequency and intensity of extreme precipitation events (Walsh and Wuebbles 2013), which would have the opposite effect. In sum, our results suggest sprinkler irrigation is more likely to be a potential climate adaptation in areas that are relatively cooler and where extreme precipitation events increase.

Third, results suggest smaller scale producers adopt sprinkler systems to a significantly lower extent than larger-scale producers. A number of studies have raised questions about whether smaller-scale agricultural operators will have the financial or technical capacity to adequately adapt to climate change (Joyce et al. 2001; McLeman et al. 2008; Wolfe et al. 2008; Walthall et al. 2012). The lower rates of sprinkler adoption among smaller-scale producers raise the question of whether policy interventions are warranted to encourage greater adoption by these producers. The Environmental Quality Incentives Program (EQIP) provides cost share payments to U.S. farmers, subsidizing adoption of a variety of conservation practices and investments. Sprinkler adoption is the third most common EQIP-subsidized practice, with growers receiving more than $400 million in payments from 1997 to 2010 (Wallander et al. 2013). Farmers with lower sales have participated less in EQIP and other water or energy conservation programs, (Nickerson and Hand 2009; Frisvold and Deva 2012).

The 2002 and 2008 Farm Bills included provisions to increase use of EQIP among beginning, socially disadvantaged, or limited resource farmers (Nickerson and Hand 2009). Schaible (2004) noted there may be an efficiency-equity trade-off in targeting smaller scale irrigators for greater conservation program participation. Large-scale operations account for the bulk of irrigation water use. Targeting larger-scale producers may thus lead to greater overall improvements in irrigation technology and water conservation.

Fourth, sprinkler adoption is lower where water costs are low and where reliance on surface water is greater. One policy option to encourage greater sprinkler adoption may be incentive pricing for surface water. Many irrigation districts do not charge irrigators the marginal cost of surface water, but apply quantity rationing, price water for district infrastructure cost-recovery, or follow other allocation schemes that do not reflect the true scarcity of water. Movements to marginal cost pricing, perhaps under tiered pricing schemes, may act to encourage adoption of more efficient irrigation technologies such as sprinklers and drip systems.

Finally, a goal of this article is to illustrate that Special Tabulations of the FRIS provided by USDA's Economic Research Service are a valuable data product for researchers without access to farm-level FRIS data. If USDA made more such special tabulations available, then researchers could use FRIS data for more than simple descriptions of general irrigation trends and patterns. The geographic scope of the FRIS means it is a valuable source of detailed data to assess climate-irrigation relationships. More and better use of this data would significantly increase our understanding of such relationships.

Acknowledgments

This work was supported by the National Oceanic and Atmospheric Administration's Climate Program Office through Grant NA07OAR4310382.

Author Bio and Contact Information

George Frisvold (corresponding author) is a Professor and Extension Specialist of Agricultural and Resource Economics at the University of Arizona. He received his degrees from the University of California, Berkeley and has been a visiting scholar at India's National Institute of Rural Development, a lecturer at Johns Hopkins

University, and Chief of the Resource Policy Branch of USDA's Economic Research Service. From 1995 to 1996, Dr. Frisvold served as a Senior Economist for the President's Council of Economic Advisers responsible for agricultural, natural resource, and international trade issues. He is currently an associate editor of the *Water Economics and Policy*. Dr. Frisvold may be reached at the Department of Agricultural & Resource Economics, 319 Cesar Chavez Building, University of Arizona, Tucson, AZ 85715; Phone (520) 621-6269; Email frisvold@ag.arizona.edu.

Ting Bai is a graduate student in the Department of Agricultural and Resource Economics at the University of Arizona. Her M.S. thesis is titled, "Farmers' Demand for Water Management Information."

References

Aillery, M. and N. Gollehon. 2003. Irrigation water management. In: *Agricultural Resources and Environmental Indicators,* R. Heimlich (Ed.). Agriculture Handbook No. AH722. Washington, D.C., U.S. Department of Agriculture, Economic Research Service.

Bernardo, D.J., N.K. Whittlesey, K.E. Saxton, and D.L. Bassett. 1987. An irrigation model for management of limited water supplies. *Western Journal of Agricultural Economics* 12: 164-173.

Bjornlund H., L. Nicol, and K. Klein. 2009. The adoption of improved irrigation technology and management practices: A study of two irrigation districts in Alberta, Canada. *Agricultural Water Management* 96: 121-131.

Burton, I. 2000. Adaptation to climate change and variability in the context of sustainable development. In: *Climate Change and Development*, L. Gómez-Echeverri (Ed.). Yale School of Forestry and Environmental Studies and United Nations Development Programme (UNDP), New Haven, CT, and New York, pp. 153-173.

Caswell, M., K. Fuglie, C. Ingram, S. Jans, and C. Kascak. 2001. *Adoption of Agricultural Production Practices: Lessons Learned from the U.S. Department of Agriculture Area Studies Project*. Agricultural Economic Report No. AER792.

Caswell, M.F. 1991. Irrigation technology adoption decisions: Empirical evidence. In: *The Economics and Management of Water and Drainage in Agriculture*, A. Dinar and D. Zilberman (Eds.). Springer Science & Business Media, New York, pp. 295-312.

Caswell, M., E. Lichtenberg, and D. Zilberman. 1990. The effects of pricing policies on water conservation and drainage. *American Journal of Agricultural Economics* 72: 883-890.

Caswell, M. and D. Zilberman. 1985. The choices of irrigation technologies in California. *American Journal of Agricultural Economics* 67: 224-324.

Caswell, M. and D. Zilberman. 1986. The effects of well depth and land quality on the choice of irrigation technology. *American Journal of Agricultural Economics* 68: 798-811.

Cavagnaro, T., L. Jackson, and K. Scow. 2006. Climate change: Challenges and solutions for California agricultural landscapes. White Paper CEC-500-2005-189-SF. California Climate Change Center. www.energy.ca.gov/2005publications/CEC-500-2005-189/CEC-500-2005-189-SF.PDF. Accessed June 14, 2016.

de Santos Loureio, N. and M. de Azevedo Coutinho. 2001. A new approach to estimate the RUSLE EI30 index based on monthly rainfall data and applied to the Algarve Region, Portugal. *Journal of Hydrology* 250: 12-18.

Diodato, N. 2004. Estimating RUSLE's rainfall factor in the part of Italy with a Mediterranean rainfall regime. *Hydrology and Earth System Sciences* 8: 103-107.

Dinar, A., M. Campbell, and D. Zilberman. 1992. Adoption of improved irrigation and drainage reduction technologies under limiting environmental conditions. *Environmental and Resource Economics* 2: 373-398.

Dinar, A. and D. Yaron. 1990. Influence of quality and scarcity of inputs on the adoption of modern irrigation technologies. *Western Journal of Agricultural Economics* 15: 224-233.

Dressing, S.A. 2003. National Management Measures to Control Non-point Source Pollution from Agriculture. EPA- 841-B-03-004. United States Environmental Protection Agency, Office of Water, Washington, D.C.

Fernald, A.G. and S.J. Guldan. 2006. Surface water–Groundwater interactions between irrigation ditches, alluvial aquifers, and streams. *Reviews in Fisheries Science* 14(1-2): 79-89.

Fleischer, A., I. Lichtman, and R. Mendelsohn. 2008. Climate change, irrigation, and Israeli agriculture: Will warming be harmful? *Ecological Economics* 67: 109-116.

Frisvold, G.B. and S. Deva. 2012. Farm size, irrigation practices, and conservation program participation in the U.S. Southwest. *Irrigation and Drainage* 61: 569-582.

Frisvold, G.B. and S. Deva. 2013. Climate and choice of irrigation technology: Implications for climate adaptation. *Journal of Natural Resources Policy Research* 5: 107-127.

Frisvold, G.B. and K. Konyar. 2012. Less water: How will agriculture in southern mountain states adapt? *Water Resources Research* 48: W05534, doi:10.1029/2011WR011057. Accessed June 14, 2016.

Gollehon, N. and W. Quinby. 2000. Irrigation in the American West: Area, water and economic activity. *International Journal of Water Resources Development* 16: 187-195.

Green, G. and D. Sunding. 1997. Land allocation, soil quality and irrigation technology choices. *Journal of Agricultural and Resource Economics* 27: 267-275.

Green, G., D. Sunding, D. Zilberman, and D. Parker. 1996. Explaining irrigation technology choices: A microparameter approach. *American Journal of Agricultural Economics* 78: 1064-1072.

Hutson, S.S., N.L. Barber, J.F. Kenny, K.S. Linsey, D.S. Lumia, and M.A. Maupin. 2004. Estimated use of water in the United States in 2000. U.S. Geological Survey Circular, 1268.

IWR MSU (Institute of Water Research, Michigan State University). 2002. RUSLE Online Soil Erosion Assessment Tool. Available at www.iwr.msu.edu/rusle/. Accessed June 14, 2016.

Jackson, L., F. Santos-Martin, A. Hollander, W. Horwath, R. Howitt, J. Kramer, A. O'Geen, B. Orlove, J. Six, S. Sokolow, D. Sumner, T. Tomich, and S. Wheeler. 2009. Potential for adaptation to climate change in an agricultural landscape in the Central Valley of California. Final Paper CEC-500-2009-044-F. California Climate Change Center. Available at www.energy.ca.gov/2009publications/CEC-500-2009-044/CEC-500-2009-044-F.PDF. Accessed June 14, 2016.

Joyce, B., V. Mehta, D. Purkey, L. Dale, and M. Hanemann. 2009. Climate change impacts on water supply and agricultural water management in California's Western San Joaquin Valley, and potential adaptation strategies. Final Paper CEC-500-2009-051-F. California Climate Change Center. Available at www.energy.ca.gov/2009publications/CEC-500-2009-051/CEC-500-2009-051-F.PDF. Accessed June 14, 2016.

Joyce, L.A., D. Ojima, G. Seielstad, R. Harriss, and J. Lackett. 2001. Potential consequences of climate variability and change for the Great Plains. In: *Climate Change Impacts on the United States: The Potential Consequences of Climate Variability and Change*. Cambridge University Press, Cambridge.

Kenny, J.F., N.L. Barber, S.S. Hutson, K.S. Linsey, J.K. Lovelace, and M.A. Maupin. 2009. Estimated use of water in the United States in 2005: U.S. Geological Survey Circular 1344: 52.

Kurukulasuriya, P. and S. Rosenthal. 2003. *Climate Change and Agriculture: A Review of Impacts and Adaptations*. Climate Change Series No. 91. World Bank, Washington, D.C.

Leib B., M. Hattendorf, T. Elliott, and G. Matthews. 2002. Adoption and adaptation of scientific irrigation scheduling: Trends from Washington, USA as of 1998. *Agricultural Water Management* 55: 105-120.

Maddigan, R.J., W. Chern, and C.G. Rizy. 1982. The irrigation demand for electricity. *American Journal of Agricultural Economics* 64: 673-680.

McLean, R.K., R. Sri Ranjan, and G. Klassen. 2000. Spray evaporation losses from sprinkler irrigation systems. *Canadian Agricultural Engineering* 42:1-8.

McLeman, R., D. Mayo, E. Strebeck, and B. Smit. 2008. Drought adaptation in rural eastern Oklahoma in the 1930s: Lessons for climate change adaptation research. *Mitigation and Adaptation Strategies for Global Change* 13: 379-400.

Mendelsohn, R. and A. Dinar. 2003. Climate, water, and agriculture. *Land Economics* 79: 328-341.

Moreno, G. and D. Sunding. 2005. Joint estimation of technology adoption and land allocation with implications for the design of conservation policy. *American Journal of Agricultural Economics* 87: 1009-1019.

Nearing, M.A. 2001. Potential changes in rainfall erosivity in the U.S. with climate change during the 21st century. *Journal of Soil and Water Conservation* 56: 229-232.

Negri, D.H. and D.H. Brooks. 1990. Determinants of irrigation technology choice. *Western Journal of Agricultural Economics* 15: 213-223.

Negri, D.H. and J. Hanchar. 1989. Water conservation through irrigation technology. Agriculture Information Bulletin No. 576, U.S. Department of Agriculture, Economic Research Service, Washington, D.C.

Negri, D.H., N.R. Gollehon, and M.P. Aillery. 2005. The effects of climatic variability on U.S. irrigation adoption. *Climatic Change* 69: 299-323.

Nickerson, C. and M. Hand. 2009. Participation in conservation programs by targeted farmers:

Beginning, limited-resource, and socially disadvantaged operators' enrollment trends. Economic Information Bulletin No. 62, USDA, Economic Research Service, Washington, D.C.

Olen, B., J. Wu, and C. Langpap. 2016. Irrigation decisions for major west coast crops: Water scarcity and climatic determinants. *American Journal of Agricultural Economics* 98: 254-275.

Potter, S.R., S. Andrews, J.D. Atwood, R.L. Kellogg, J. Lemunyon, L. Norfleet, and D. Oman. 2006. Model simulation of soil loss, nutrient loss, and change in soil organic carbon associated with crop production. USDA-NRCS, Conservation Effects Assessment Program (CEAP). Available at http://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_013138.pdf. Accessed June 5, 2016.

Sauer T., P. Havlik, U.A. Schneider, E. Schmid, G. Kindermann, and M. Obersteiner. 2010. Agriculture and resource availability in a changing world: The role of irrigation. *Water Resources Research* 46: W06503.

Schaible, G. 2004. Irrigation, water conservation, and farm size in the western United States. *Amber Waves* 2: 8.

Schaible, G., C. Kim, and N. Whittlesey. 1991. Water conservation potential from irrigation technology transitions in the Pacific North-West. *Western Journal of Agricultural Economics* 16: 194-206.

Schoengold, K., D. Sunding, and G. Moreno. 2006. Price elasticity reconsidered: Panel estimation of an agricultural water demand function. *Water Resource Research* 42: W09411, doi/10.1029/2005WR004096. Accessed June 14, 2016.

Schuck, E.C., W.M. Frasier, R.S. Webb, L.J. Ellingson, and W.J. Umberger. 2005. Adoption of more technically efficient irrigation systems as a drought response. *International Journal of Water Resources Development* 21: 651-662.

Schuck, E.C. and G.P. Green. 2001. Field attributes, water pricing, and irrigation technology adoption. *Journal of Soil and Water Conservation* 56(4): 293-298.

Skaggs, R.K. 2001. Predicting drip irrigation use and adoption in a desert region. *Agricultural Water Management* 51: 125-142.

Skaggs, R.K. and Z. Samani. 2005. Farm size, irrigation practices, and on-farm irrigation efficiency. *Irrigation and Drainage* 54: 43-57.

Sloggett, G. 1985. Energy and U.S. Agricultural Irrigation Pumping, 1974-1983. Agricultural Economics Report No. 545, Washington, D.C., U.S. Department of Agriculture, Economic Research Service.

Smit, B. and M.W. Skinner. 2002. Adaptation options in agriculture to climate change: A typology. *Mitigation and Adaptation Strategies for Global Change* 7: 85-114.

Solley, W.B., R.R. Pierce, and H.A. Perlman. 1998. Estimated use of water in the United States in 1995, U.S. Geological Survey Circular 1200, 71.

Teigen, L. and F. Singer. 1988. Weather in U.S. agriculture: Monthly temperature and precipitation by state and farm production region, 1950–86. Statistical Bulletin No. 765. Department of Agriculture, Economic Research Service,Washington, D.C., U.S.

USDA, ERS (U.S. Department of Agriculture, Economic Research Service). 2016. Data Sets: Western Irrigated Agriculture: Tables. Available at http://www.ers.usda.gov/data-products/western-irrigated-agriculture.aspx. Accessed June 28, 2016.

U.S. Department of Agriculture (USDA, NASS). 1998. Census of Agriculture, AC97-SP-1, vol. 3, Special Studies, part 1, 1998 Farm and Ranch Irrigation Survey, USDA NASS, Washington, D.C.

U.S. Department of Agriculture (USDA, NASS). 2004. Farm and Ranch Irrigation Survey (2003), vol. 3, Special Studies, part 1 of the 2002 Census of Agriculture, AC-02-SS-1, National Agricultural Statistics Service, Washington, D.C.

U.S. Department of Agriculture (USDA, NASS). 2010. Farm and Ranch Irrigation Survey (2008), vol. 3, Special Studies, part 1 of the 2007 Census of Agriculture, AC-07-SS-1, National Agricultural Statistics Service, Washington, D.C.

USDA, NRCS (U.S. Department of Agriculture, Natural Resources Conservation Service). 2000. Summary Report: 1997 National Resources Inventory. Rev. December 2000. Natural Resources Conservation Service and Iowa State University Statistical Laboratory, Washington, D.C., and Ames, IA.

Wallander, S., M. Aillery, D. Hellerstein, and M. Hand. 2013. The role of conservation programs in drought risk adaptation, ERR-148, USDA, Economic Research Service, Washington, D.C.

Walsh, J. and D. Wuebbles. 2013. Our changing climate. Chapter 2. In: *Third National Climate Assessment: Draft Report for Public Comment*, U.S. Global Change Research Program, Washington, D.C.

Walthall, C.L., J. Hatfield, P. Backlund, L. Lengnick,

E. Marshall, M. Walsh, S. Adkins, M. Aillery, E.A. Ainsworth, C. Ammann, C.J. Anderson, I. Bartomeus, L.H. Baumgard, F. Booker, B. Bradley D.M. Blumenthal, J. Bunce, K. Burkey, S.M. Dabney, J.A. Delgado, J. Dukes, A. Funk, K. Garrett, M. Glenn, D.A. Grantz, D. Goodrich, S. Hu, R.C. Izaurralde, R.A.C. Jones, S.-H. Kim, A.D.B. Leaky, K. Lewers, T.L. Mader, A. McClung, J. Morgan, D.J. Muth, M. Nearing, D.M. Oosterhuis, D. Ort, C. Parmesan, W.T. Pettigrew, W. Polley, R. Rader, C. Rice, M. Rivington, E. Rosskopf, W.A. Salas, L.E. Sollenberger, R. Srygley, C. Stöckle, E.S. Takle, D. Timlin, J.W. White, R. Winfree, L. Wright-Morton, and L.H. Ziska. 2012. Climate change and agriculture in the United States: Effects and adaptation. USDA Technical Bulletin 1935. USDA Climate Change Program Office, Washington, D.C.

Ward, F.A. and M. Pulido-Velazquez. 2008. Water conservation in irrigation can increase water use. *Proceedings of the National Academy of Sciences* 105: 18215-18220.

Wischmeier, W.H. and D.D. Smith. 1978. *Predicting Rainfall Erosion Losses: A Guide to Conservation Planning*. Agricultural Handbook No. 537. U.S. Department of Agriculture, Soil Conservation Service, Washington, D.C.

Wolfe, D., L.H. Ziska, C. Petzoldt, A. Seaman, L. Chase, and K. Hayhoe. 2008. Projected change in climate thresholds in the Northeastern U.S.: Implications for crops, pests, livestock, and farmers. *Mitigation and Adaptation Strategies for Global Change* 13: 555-575.

Yu B. and C.J. Rosewell. 1996. A robust estimator of the r factor for the universal soil loss equation. *Transactions of the American Society of Agricultural Engineers* 39: 559-561.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 78-84, August 2016

Building Watershed Management Capacity in Nigeria: Expanding the Role of Agriculture Colleges

John W. Groninger

Southern Illinois University, Carbondale, IL

Abstract: Developing countries face many critical social, agricultural, and natural resource-related issues where watershed management plays a prominent role. Short-term gains in productivity often occur at the expense of long term system resiliency needed to achieve sustainability. Post-secondary (tertiary) education institutions are underfunded and hampered by bureaucratic inertia limiting their relevance toward addressing emerging water resources concerns. As populations urbanize, rural constituencies continue to lose influence. Using the case of Samaru College of Agriculture, part of Ahmadu Bello University in north-central Nigeria, this paper identifies institutional challenges and suggests solutions to help similar institutions better address watershed management issues. Given current constraints, a watershed management curriculum could be built upon existing courses offered in related majors. Developing partnerships with an expanding private sector could help create opportunities for meaningful solutions to management issues and foster problem solving skills among students. Outside investment in increasing university capacity to address watershed education, including hiring staff focused on these issues, is justifiable given the humanitarian and security crises that have been afflicting northern Nigeria's vulnerable agricultural landscape.

Keywords: *climate change, counter insurgency, food security, hydrophilanthropy, Kaduna, Sahel*

Many developing regions across Africa face a common set of challenges encompassing issues of water supply and quality that directly threaten social stability. In Nigeria, a rapidly expanding and urbanizing population has long exceeded domestic food production capacity and is expected to more than double in the next 35 years (Adesina 2012). In an effort to restore food self-sufficiency, the Nigerian federal government, backed by domestic and international investors, has developed ambitious plans to increase production of primary food crops including rice, cassava, yam, sorghum, maize, groundnut, millet, common beans, and livestock. Presently, irrigation covers only seven percent of irrigable land. However, Nigeria is undergoing a rapid expansion in agricultural capacity development, including private investment (Adesina 2012). More lands are becoming productive in a step toward meeting the Nigerian food deficit. Drought, overgrazing, erosion, and gully formation are an ongoing threat to existing and expanding agricultural initiatives across Africa (World Bank 2012). Finally, security instability, most notably posed by Boko Haram, limits mobility and discourages investment in the institutions needed to address these problems.

Staff and administrators at Nigeria's tertiary agricultural institutions are acutely aware of the value of water and watershed management as critical to securing environmental and agricultural resiliency. However, facilities and curriculum are outdated and remain focused toward production and training protocols that were generally developed prior to the 1980's. Many of these challenges are common at agriculture colleges throughout the developing world (Maguire 2011; Baker et al. 2015). Additionally, unpredictable budgets, bureaucratic restrictions on curriculum reform, and a negative perception of agriculture as a career choice among aspiring young people are barriers faced by those hoping to build the

capacity developing nations need in order to address watershed management issues.

Threats to water supply and quality in developing countries are most successfully addressed through a broad adaptation strategy (Hurd 2012). Several projects pertinent to watershed management have been conducted throughout Nigeria, including water quality monitoring, agricultural system adaptation, and capacity development within government management agencies (Lal 1985; Igboekwe et al. 2008; Macheve et al. 2015). However, formal efforts to increase capacity through reform of existing post-secondary education programs have received little attention. This paper presents challenges and identifies opportunities faced by the federally-funded Samaru College of Agriculture, Ahmadu Bello University in expanding its own capacity to address key watershed management challenges in Nigeria and across West Africa. A strategy is presented to increase the role of Nigerian tertiary agriculture institutions in addressing the country's watershed management challenges as part of a larger strategy to make these institutions more broadly relevant to the agriculture industry and society as a whole.

Study Methods

The information presented here is based on documents and interviews conducted by the author at the request of Samaru College of Agriculture (SCA) in Zaria, Nigeria. A rigorous qualitative approach was used employing triangulation; including interviews, document analysis, and observation (McNiff and Whitehead 2006). All interviews were conducted on the SCA campus from July 31 to August 13, 2014 by the author with the assistance of Mr. John Iruaga of Winrock International. Group interviews with SCA academic departments were typically attended by greater than half of the teaching and support staff housed therein. Open-ended interviews typically lasted 60 to 120 minutes and included a preliminary SWOT (strengths, weaknesses, opportunities, and threats) analysis (Yin 1984; Merriam 1998). Discussions were sometimes continued during short tours of laboratories and other classroom facilities. Additionally, farms, administrative facilities, and the library were toured. At the time of the interviews, security concerns, as ascertained by university personnel, restricted the interviewer's mobility to the campus, university farms, and closely surrounding environment. Enrollment data were provided by SCA Student Affairs office and curriculum summaries by the respective academic departments.

The SCA campus is situated within the Guinea savanna region of northern Nigeria with annual rainfall averaging 1211 mm (Climate-data.org 2016). From its origin in 1921 as an Agricultural training center focused on increasing cotton and groundnut exports from the region, the college was formally named in 1932 and evolved under the supervision of the agricultural research arm during British colonial rule and, subsequently, the government of the Federal Republic of Nigeria. In 1962, the newly-founded Ahmadu Bello University (ABU) absorbed SCA and quickly grew to become one of the largest universities in Africa. In 2006, SCA came under the authority of the National Board of Technical Education (NBTE). Since that time, there is a perceived disconnect reported by SCA personnel between the traditional SCA mission and the decisions rendered by NBTE regarding proposed curricular content, changes, and enrollment caps. Presently, the college has a total of 40 full-time academic staff and 544 students enrolled in National Diploma and Higher National Diploma programs across several traditional agriculture disciplines.

The student body is drawn from across Nigeria and beyond, in part attracted by the ABU university environment. Agriculture career paths are perceived as undesirable among many students attending university because they rarely see its relevance to future personal prosperity. Many students enroll at SCA with little commitment to pursuing the agriculture curriculum, instead planning to re-apply for more preferred, and competitive, programs elsewhere at ABU.

Student extracurricular activities are largely limited to participation in general student government and social organizations. There are few special interest and professional development clubs as compared to U.S. colleges of agriculture of similar size. The college has several teaching laboratories varying greatly in adequacy of equipment, but the new agriculture engineering

facility, including an irrigation hydrology lab, stands out above others. Farm facilities are extensive, serving as teaching tools and revenue generators by selling grain crops, primarily maize. Planting and tillage are mechanized with other aspects of production using labor from surrounding villages. A pasture seed facility is under development with the hope of generating revenue from seed sales. Additional arable land is available to dramatically expand seed and row crops for market or to help increase the college's existing animal production operations.

The practical agriculture curriculum at SCA prepares students for middle level employment upon entry into the workforce. All curricula are approved by NBTE and reflect a balance between theoretical and practical knowledge. Students and staff consistently express a desire for more hands-on practical exercises that are relevant to the current state of the industry. Critical equipment for these are, in many instances, partially functional, non-functional, or non-existent. The curriculum guides specify a large amount of rote memorization and suggest little opportunity for development of higher-level creative problem solving skills. In-class training lacks audio-visual support resources in almost all classrooms. Fortunately, there are many opportunities for field trips in Zaria and these are portrayed by staff as a significant part of the student training experience. However, staff lament the removal of agricultural extension and de-emphasis of field trips by NBTE.

Prior to graduation, students serve a compulsory internship (Attachment) supervised by SCA staff and typically hosted by industry and government agencies. Staff must periodically travel to the location of the Attachment to assess the student's progress and speak with supervisors. In some fields, there appears to be disconnection between Attachment content and host and the actual employers of graduated students. In particular, many Attachments take place in government ministries where little relevant experience is gained. Given the potential for regular interaction between SCA staff and attachment hosts, there are many missed opportunities to use Attachments as a means for establishing the kinds of university-industry partnerships and collaborations that benefit U.S. agricultural universities.

At the time of writing, the Nigerian economy is severely weakened by low petroleum prices. Consequently, sizeable internal investment in education, including new program development, is unlikely to occur in the near term. However, much may be accomplished with presently available resources to increase focus on watershed management while setting the stage for broader institutional growth if additional resources become available. Following are suggestions to increase university capacity to address watershed management in a manner that is realistic within present constraints and priorities.

Watershed Management and the Role of Agriculture Colleges

A first consideration is educating the public about the critical linkages between improvements in agriculture and natural resources management and securing a prosperous future. Agriculture colleges and career paths are already burdened by negative attitudes among prospective students and their families that are in place prior to university enrollment (Rivera 2011). Education initiatives should help students recognize the importance of sanitation, soil and water conservation, and water conflict that are pertinent to agriculture and to Nigeria as a nation. Introducing students to watershed issues, management, and technologies prior to university matriculation could contribute toward overcoming the agriculture stigma held by potential students. Prospective students also need to see themselves as having options for success in these critical areas as evidenced by clear and reliable education and career paths. Outreach efforts targeted toward secondary school students and broader society could positively influence attitudes regarding agriculture-related disciplines as relevant and desirable. The SCA staff currently employed within the extension program would be natural leaders in this arena provided they are able to adapt their communication strategies targeting the traditionally rural audiences they served earlier in their careers toward strategies reflecting the interests and sensibilities of increasingly urban populations. These outreach efforts could target intended audiences through available media outlets (Baker et al. 2015). Post-secondary

schools could pursue campaigns independently, as a consortium, or collaboratively with NGOs, such as Global Environment Facility and United Nations Environment Program World Water Day activities, to build awareness regarding the link between watershed management and daily life. The next step would be promoting realistic education paths to help potential students address these issues (Rivera 2011).

Once on campus, students face many barriers toward receiving training focused on watershed management issues. Appropriate host degree programs containing elements relevant to watershed management do exist, particularly in the areas of range and pasture management and irrigation agronomy (soil and water engineering). However, these degree programs are highly prescriptive in terms of course requirements. Increasing flexibility within existing programs to permit student focus on watershed management would help graduates better respond to watershed management problems while avoiding the challenges inherent to restructuring academic departments. Leadership, or at least increased support, from NBTE encouraging curriculum revision and new academic program development teaching practical problem solving approaches to emerging challenges would help facilitate such initiatives at SCA and in other agriculture institutions nationally.

Any new approaches to education and research programs would be aided by hiring staff from universities or work environments already supporting such programs. This would require a significant deviation from current hiring practices that reward students who complete their academic training at SCA and ABU with permanent employment on the teaching staff. Even if staff hired from outside eventually seek employment elsewhere, influx of new staff to replace them would continue institutional exposure to new ideas and initiatives. While all staff should be encouraged to innovate, younger staff tend to have greater interest and aptitude for pursuing new initiatives. These individuals would be the best candidates to lead university efforts in these areas if properly incentivized.

Agriculture industry is expanding rapidly in Nigeria. Experience elsewhere in Africa suggests that increased productivity could be accompanied by myriad problems associated with higher runoff, erosion, and sedimentation rates. Agriculture colleges, with their technical expertise, could serve as leaders in helping ensure that producers are able to sustain increased productivity while applying principles of sound watershed management. At this time, there is little engagement between SCA, government agencies, and agricultural industries to pursue such productive partnerships (Rivera 1995; Rivera 2011). This is despite the fact that university staff routinely supervises students pursuing their Attachments. As of summer 2014, ABU was building a large conference facility that would be well-suited to serve as a regional meeting place to facilitate industry/stakeholder engagement. The value of partnerships to increase research potential and ensure that classroom material is current, a model well-developed in the U.S., is enthusiastically received by SCA staff and regional industry, alike. However, this approach strongly differs from the compartmentalized status of education that remains deeply ingrained in many old world universities. Expanding partnerships had been previously identified as a key recommendation for tertiary education institutions in Nigeria (Baker et al. 2015).

SCA curriculum guides suggest that many courses potentially relevant to watershed management-centered studies are already available (Table 1). However, actual course content is geared toward memorization of facts and presently lacks the critical thinking skills needed by students to address emerging and complex problems. This approach to post-secondary education is deeply rooted in colonial and early post-colonial education philosophies (Teferra and Altbach 2004). A dialogue is needed among agriculture colleges, NBTE, and stakeholders in both public and private sectors to modernize curricula so that graduates are better able to identify and solve unique and emerging problems throughout their careers (Davis et al. 2008). The physical facilities on and off campus and some potential partnerships are already in place. Encouraging the pursuit of problem-based teaching among staff who rose through the ranks under the present system poses a more formidable challenge. However, the presence of instructors exposed to problem-based education earlier in their training offers a starting point for reform.

Table 1. Courses presently available to students at Samaru College of Agriculture that would have special relevance to a Watershed Management course of study.

Department	Course Title	Contact Hours Per Week
Animal Production Technology	Pasture and Range Management	5
Agriculture and Bio-Environmental Engineering	Land Clearing and Development	3
	Soil Mechanics	5
	Land Law	2
	Principles of Surveying and Farm Buildings	5
	Tillage and Planting Machinery	5
	Engineering Properties of Biological Materials	3
	Farm Machinery Design and Construction	4
	Farm Waste Management	4
	Agricultural Engineering in Development	2
Pasture and Range Management	Range and Pasture Plant Identification	3
	Principles of Range Management	3
	Sown Pastures Production	TBD
	Advanced Range Management	3
	Range Utilization Systems	2
	Range Patrol Management	TBD
	Range Inventory Evaluations and Improvement	3
Agriculture Engineering	Irrigation and Drainage	4
	Soil and Water Conservation	2
Mechanical Engineering	Computer-Aided Design and Drafting	3
	Technical Drawing	4
Agricultural Technology	Introduction to Soil Science	2
	Introduction to Farm Woodland Management	4
	Principles of Crop Production	4
	Industrial Crop Production I and II	8
	Annual Crops	3
	Agroclimatology	2
	Soil Fertility and Crop Nutrition	4
	Farm Soil Management	4
	Farm Management	5
	Agricultural Extension and Rural Sociology	2

Administrators and staff at SCA, and undoubtedly others in similar positions in Nigerian institutions of higher learning, could pursue some meaningful incremental changes in watershed management given current capacities. However, increased and long-term investments are needed to support education overhaul to include effective principles of watershed management appropriate for a country that is growing, both as an urban society and as an agriculture producer. Instituting specific programs addressing watershed management, such as forestry/natural resources management, would be welcomed by SCA administrators. Improving computer/GIS facilities, internet connectivity, and increased power supply reliability would also be productive. Internal allocation of resources

at ABU tends to underestimate the value of agriculture as more prominent programs in newer fields of technology compete successfully for funds. Nationally, economic realities make this unlikely to be an internally generated initiative. Public universities throughout Nigeria face a large number of serious challenges, made worse by serious macroeconomic challenges, suggesting that meaningful education reforms will be slow to happen if dependent upon internally-generated funds (Iruonagbe et al. 2015).

World investment in Nigerian education and conservation agriculture should be given serious consideration, both on a humanitarian basis and in the interest of promoting global security, since terrorist groups have proven adept at exploiting tensions between grazers and farmers (OECD 2013). Changing land use patterns across Africa coupled with land degradation, precipitation variability, and policies to encourage sedentary agriculture are increasing grazing pressure at the expense of soil and field crop productivity. In turn, farmers are expanding operations or compensating for degraded lands by establishing row cropping on lands traditionally used for grazing, typically by using pre-conservation agriculture practices. Resulting runoff and gully development imperil downstream lands and communities. Insurgents capitalize on local tensions and grievances, including those rooted in water and land resource management, to increase their influence over the rural territories they use to support their logistical operations (Benjaminsen 2008; Aduku 2015). Natural resource sustainability and resiliency through improved watershed management should be part of a broader security strategy that increases government capacity and in turn builds trust among the otherwise disaffected rural populations. In this way, watershed management education and technology transfer could be a meaningful part of a broader strategy to help prevent armed conflicts in this highly populated country and beyond (Lowi 1999; Miller 2011; Groninger and Ruffner 2014). Recently, USAID has supported an initiative to increase grant writing capacity among SCA staff. Hopefully, these efforts will ultimately support changes and potential solutions pertinent to those closest to the challenges.

Conclusion

Productive watersheds and abundant water resources are important to the success of a rapidly growing Nigeria. Tertiary agricultural institutions are necessary for students and society to prepare for a future that includes more effective watershed management to protect and efficiently use water and land resources. Broader institutional relevance is challenged by an education system that is highly bureaucratic, insular, and slow to respond to societal changes. However, many of the challenges encountered in Nigeria higher education are shared by their counterparts in the U.S. Both parties would gain immensely from sharing perspectives and strategies to address these common problems, while setting the stage for mutually beneficial collaboration. Maintaining open lines of communication and providing opportunities for bilateral professional exchanges among technical personnel and leaders in academic, development, and security sectors could help address watershed related issues that span regional and global scales.

Acknowledgements

This research was supported by U.S. Agency for International Development through the John Ogonowski and Doug Bereuter Farmer to Farmer Program for Agricultural Education and Training, administered by Winrock International. The author was a volunteer and received no compensation related to this work. All opinions expressed herein are those of the author and do not reflect those of USAID or Winrock International.

Author Bio and Contact Information

John Groninger is Professor of Forestry at Southern Illinois University, a position he has held since 1997. His teaching, research, and outreach interests include silviculture, agroforestry, urban forestry, and watershed rehabilitation. He can be reached at groninge@siu.edu and at 1205 Lincoln Drive, Room 184, Carbondale, IL 62901.

References

Adesina, A. 2012. Investing in Nigeria's Agricultural Value Chain. Presentation at the Bank of Industry's Nigerian Investment Forum, London, July 30, 2012. Available at http://www.newworldnigeria.com/pdf/AdesinaLondonBOIForumfinal.pdf. Accessed May 4, 2016.

Aduku, E.B. 2015. A study of violence-related deaths in Karasuwa, Machina, Nguru, and Yusufari local government areas of Yobe State North-Eastern Nigeria. *IFRA-Nigeria Working Papers Series*, *No. 55*. December 11, 2014. Available at http://www.nigeriawatch.org/media/html/WP24ADUKU.pdf. Accessed May 4, 2016.

Baker, M., M. Bassey, A.A. Jimoh, and T. Akande. 2015. An expertise-oriented evaluation of the Center for Entrepreneurship Development and Vocational Studies, Federal Polytechnic, Ado-Ekiti, Nigeria: Implications for assessing and improving the Federal Polytechnic System in Nigeria. *Journal of International Agriculture and Extension Education* 22(1): 63-74.

Benjaminsen, T.A. 2008. Does supply-induced scarcity drive violent conflicts in the African Sahel? The case of the Tuareg Rebellion in Northern Mali. *Journal of Peace Research* 45(6): 819-836.

Climate-data.org. 2016. Climate: Kaduna. Available at http://en.climate-data.org/location/539/. Accessed May 4, 2016.

Davis, K.E., J. Ekboir, and D.J. Spielman. 2008. Strengthening agricultural education and training in sub-Saharan Africa from an innovation systems perspective: A case study of Mozambique. *Journal of Agricultural Education* 14(1): 35-51.

Groninger, J.W. and C.M. Ruffner. 2014. Learning rural engagement: Afghan agriculture and the future of rural stabilization. *Small Wars Journal* 10(3): March 13.

Hurd, B. 2012. Climate vulnerability and adaptive strategies along the Rio Grande/Rio Bravo Border of Mexico and the United States. *Journal of Contemporary Water Research and Education* 149: 56-63.

Igboekwe, M.U., V.V.S. Gurunadha Rao, and E.E. Okwueze. 2008. Groundwater flow modelling of Kwa Ibo River watershed, southeastern Nigeria. *Hydrological Processes* 22: 1523-1531.

Iruonagbe, C.T., D. Imhonopi, and M.E. Egharevba. 2015. Higher education in Nigeria and the emergence of private universities. *International Journal of Education and Research* 3(2): 49-64.

Lal, R. 1985. Mechanized tillage systems effects on properties of a tropical alfisol in watershed cropped to maize. *Soil and Tillage Research* 6: 149-161.

Lowi, M.R. 1999. Water and conflict in the Middle East and South Asia: Are environmental issues and security issues linked? *The Journal of Environment and Development* 8(4): 376-396.

Macheve, B., A. Danilenko, R. Abdullah, A. Bove, and L.J. Moffitt. 2015. State water agencies in Nigeria: A performance assessment. Washington, DC: World Bank. © World Bank. Available at https://openknowledge.worldbank.org/handle/10986/22581. License: CC BY 3.0 IGO. Accessed May 4, 2016.

Maguire, C. 2011. *Building the Base for Global Food Security - Agricultural Education and Training.* United States Agency for International Development, AID-AEG-I-00-04-00010, AID-OA-TO-10-00017, Washington D.C., April 6, 2011.

Merriam, S.B. 1998. *Qualitative Research and Case Study Applications in Education.* Jossey-Bass Publishers, San Francisco, CA.

Miller, D. 2011. Going outside the wire: Liaising with special operation forces to rebuild agriculture in Afghanistan. *Small Wars Journal* 7(5): May 27.

McNiff, J. and J. Whitehead. 2006. *All You Need to Know about Action Research*. Sage Publications, London.

OECD (Organization for Economic Co-operation and Development). 2013. *Conflict over Resources and Terrorism Two Facets of Insecurity*. OECD Publishing. Available at http://dx.doi.org/10.1787/9789264190283-en. Accessed May 4, 2016.

Rivera, W.M. 1995. Human resource development in the agriculture sector: Three levels of need. *International Journal of Lifelong Education* 14:1, 65-73.

Rivera, W.M. 2011. *Covering All the Angles: Using a Spectrum of Ideas to Expand and Transform Agricultural Education and Training*. United States Agency for International Development, AID-AEG-I-00-04-00010, AID-OA-TO-10-00017, Washington D.C., June 20, 2011.

Teferra, D. and P.G. Altbach. 2004. African higher education: Challenges for the 21st century. *Higher Education* 47: 21-50.

World Bank. 2012. *World Development Indicators 2012*. World Bank Publications, Washington D.C.

Yin, R.K. 1984. *Case Study Research: Design and Methods*. Sage Publications, Beverly Hills, CA.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 85-97, August 2016

Heard it through the Grapevine - Social Network Analysis of a Hydrology Class

*E. Annette Hernandez and Venkatesh Uddameri

Department of Civil, Environmental, and Construction Engineering, Texas Tech University, Lubbock, TX
**Corresponding Author*

Abstract: Social network analysis (SNA) was investigated as an efficient way to understand the dynamics behind collaborative endeavors and group formation in project based courses. The SNA was applied to a junior level undergraduate engineering hydrology class with two sections taught by two professors with varying levels of structure. It was found that students tended to use formal sources of information (i.e., professor, teaching assistant, textbook, or class notes); however, the use depended on the amount of structure in the class. More independent students did not collaborate in assigned groups, preferring to self-select study groups. Furthermore, the less structured class created a split atmosphere of "connected" students who had many interactions as well as "disconnected" students on the other end of the spectrum. The more structured class created fewer but more even distribution of interactions between students. The SNA study corroborated that collaborations correlate positively with class grade. In fact, those students with fewer interactions with their classmates performed poorly in the class.

Keywords: *social network analysis, group based project, engineering hydrology course, teamwork, peer learning*

Water resources engineering is inherently a collaborative endeavor and each project poses unique multi-faceted challenges that need to be overcome for successful design. As such, water resources engineers and hydrologists regularly work in (often interdisciplinary) teams. The ability to work in teams is therefore an important skill necessary for successful water resources engineering practice and must be cultivated in students as quickly and frequently as possible. The first year introductory "cornerstone" design and senior year "capstone" design have been the typical academic responses to the industry criticism of not training engineering students in soft-skills such as teamwork and communication (Mills and Treaqust 2003; Dunlap 2005; McGann and Cahill 2005; Russell et al. 2005; Shuman et al. 2005; Janicki et al. 2006; Shelby et al. 2013; Liu and Dou 2015). A variety of different team-based activities such as allowing students to collaboratively complete assignments, work together on short class projects, and even take group exams have been adopted by faculty in various engineering courses (Felder et al. 1998; Johnson 1999; Mills and Treaqust 2003; Tonso 2006; Freeman et al. 2014; Huang and Pierce 2015). These approaches not only instill and impart teamwork and collaboration skills, but have the added advantage of facilitating "peer learning," the value of which is increasingly being recognized in academic circles (Boud et al. 2014).

While the benefits of collaborative learning are readily recognized, concerns exist as to whether all students are contributing equally within their teams. Some team-members may choose to be "free-riders" which causes undue burden on the other participating members of the team. On the other hand, certain groups (or teams) may contain a strong individualistic member who may dominate and not listen or allow other team-members to contribute. Teams may also form cliques (or sub-groups) and selectively exclude certain members. This issue may arise due to perceived educational competencies as well as differences in learning and operational styles between students of different genders, races, or nationalities. These dysfunctionalities not only take away from the benefits of team-based

learning paradigms, but also increase the student's dissatisfaction with the subject matter, lower self-esteem, increase self-doubt regarding the subject matter, and eventually may contribute to retention issues (Oakley et al. 2004; Knight et al. 2007; Oakley et al. 2007; Stump et al. 2011). Tackling these issues is somewhat easier in capstone (or cornerstone) design courses as teamwork is an integral part of the outcomes and deliverables of the course. On the other hand, dysfunctional team issues may largely go unnoticed in hybrid classes which include both individualistic content-based assessment, and team-oriented problem-based learning pedagogy. Often times, the team project is a small part of the overall student's grade and as such students may be less inclined to discuss team dysfunctionalities. Furthermore, team projects are usually assigned late in the semester as a way to tie-up individual concepts presented throughout the semester; therefore, the students most affected by team dysfunctionalities may give up hope and think it is too late to discuss matters.

In the traditional mode of undergraduate instruction, students formally obtain their information from the instructor and reinforce it by reading the textbook and other supplemental material (usually provided by the instructor) as well as carrying out prescribed assignments. Most classes also have graduate teaching assistants (TA) who help students with their queries informally during their office hours or semi-formally through supplemental instruction. In addition, students form study groups and share information informally. As TAs tend to be closer in age to students, and are in a somewhat similar situation (i.e., juggling their time between classes and other student-type activities) some students feel a higher level of comfort obtaining information from TAs rather than the instructor. This level of comfort may be further enhanced if the instructor has other administrative responsibilities and is perceived to be busy by students. In a similar vein, some students find it more comfortable to obtain information from a peer (who may have a better grasp of the subject matter) or one who has already met with the instructor or TA (despite the second-hand nature of the information) rather than go themselves. The proliferation of web-based materials, the advent of social media, and the availability of online resources have increased the informal channels through which students can receive information. Information presented with high graphical content (e.g., animations and YouTube videos) may be appealing to some students instead of the static presentations found in a textbook. Students may also find greater comfort and convenience in obtaining information from digital sources as they are available 24/7 and provide anonymity to the student which shields them from feeling inadequate in front of their instructors and peers.

Understanding the information flow in a classroom is important on several fronts: 1) it provides an understanding of how the textbook and assigned readings are being perceived and used by the students; 2) it helps evaluate the role of the TA and make appropriate adjustments mid-course; 3) it aids in the identification of online materials that students have found helpful but are otherwise unknown to the instructor; 4) it helps identify those students who have formed study-groups and those that appear to be working individually; and 5) it enables the instructor to understand the differences in working with students belonging to under-represented minorities versus those belonging to the majority population to find better ways to reach out to the class as a whole.

Social network analysis (SNA) is an information-theoretic approach that seeks to characterize and quantify the structure of social interactions among a set of actors (individuals within a system). SNA uses concepts derived from network and graph theories to map interactions within the network (Otte and Rousseau 2002; Freeman 2005). SNA has not only found widespread acceptance in various social sciences and natural resources disciplines, but has also been used to understand computer and telecommunication networks (Pinheiro 2011). Martinez et al. (2003) made use of social network analysis to study social interactions in a project-based computer architecture class. de Laat et al. (2007) used SNA in conjunction with content analysis (CA) and critical event recall (CER) to understand how students learned in a computer-assisted network learning environment. Amory (2007) presented a game object model (GOM) for transformative learning and acknowledged the important role of social networks in promoting

gender-inclusive, non-confrontational outcomes and building democracy and social capital through computer-mediated communication tools. More recently, Vaquero and Cebrian (2013) reported that students' performances are linked to social structures and interactions. Lee and Bonk (2016) evaluated the role of blogs in improving peer connectivity among students and use social network analysis to quantify improvements in classroom interactions that are critical to building learning communities. However, the use of SNA as a mid-course assessment tool has not been reported in the literature and as such forms the focus of this study.

The present study is an innovative application of social network analysis in a project-based introductory junior-level engineering hydrology course. The study demonstrates how social network analysis of data generated from a simple, less than 10 minute survey that asks students to list the names of students and other sources (TA, textbook, instructor, etc.) they use to solve homework problems, study for exams, and work on the project during the semester can yield valuable insights related to information flow in the class. The results of the study lead to a fundamental understanding of where the students are getting their information and help identify key resources and personnel upon which students most rely. The analysis also provides other insights critical to reaching out to students in the class and thus promotes effective, information-oriented teaching practices.

Structural Characteristics of Networks - Social Network Analysis

The mathematical theory of SNA provides a formal framework to assess the interactions taking place in a social system. The basic vocabulary and key concepts pertinent to understanding social networks, particularly in the context of this study, are summarized here. The interested readers are referred to several excellent textbooks such as Knocke and Yang (2008) and Borgatti et al. (2013) for more detail. SNA begins with the definition of a social system of interest. Just as with physical systems, a social system is conceptualized to have well-defined boundaries. The social system is comprised of a set of individuals (referred to as **actors** in SNA parlance). There is a certain degree of interaction between the actors within a network. Characterizing the nature and extent of these interactions is the primary focus of SNA.

Networks within a social system can be defined at several levels. The **ego network** maps the interactions of an individual actor within the social system. The focal actor is referred to as the **ego** and those connected to this actor are referred to as **alters**. An ego network can be constructed for every actor within the social system. In contrast, a **complete network** simultaneously maps the interactions between a set of actors within a social system. A **directional graph** or a **digraph** is a graphical representation of the social network. The actors of the network form the **nodes** of the digraph and directed lines called **arcs** connect the nodes. The arcs have a direction and connect senders to receivers. **Senders** are actors who nominate and **receivers** are the nominees. A **reciprocal** or a **mutual tie** exists between two actors when they both nominate each other. Unlike a digraph, a **graph** has no directional arcs and simply connects the actors with arrowless lines called **edges**. In a classroom setting, links to formal information sources (i.e., instructor, textbook, and TA) are the senders of information while students are mostly receivers. In groups with high collaboration and teamwork, there are a large number of mutual ties.

Ego networks are useful to compare various actors within a social system. For example, some egos (actors) have larger networks than others which may have an effect on external measure (e.g., performance in the class). In addition to the size, it is also important to understand: 1) how the alters of an ego share ties with others (referred to as **ego network density**); 2) the attributes of egos and alters (e.g., gender, race, grade point average (GPA)); and 3) the tie-strengths across the ego networks (i.e., is there a shared learning or one-directional teaching type behavior). **Centrality** is an actor-level measure that reflects the ideas of relative importance of an individual within a larger (complete) network. Actors who have more ties compared to others in the network are considered to be more **central** to the network. Several measures of centrality have been proposed in the literature and are summarized in Table 1. Many of these centrality measures can only be measured on graphs

(and not digraphs) and do not account for mutual ties. In a classroom context, formal information sources (instructor, textbook, and TA) should have high centrality scores relative to individual students. Centrality measures are also useful to identify connected students who are sought after by other classmates for information. In a similar vein, actors having low centrality scores are not well connected within the network and could be used to identify students who are not fully integrated within their groups and/or information sources that are not perceived as effective by the students.

Subgroups of networks can be viewed as cohesive subsections of a network and as such could represent a study group or a design team within the class. Alternatively, subgroups can also be conceived as subsets of actors (e.g., instructors of different sections, TAs) that share a similar role. Positional analysis is a class of SNA that seeks to evaluate networks associated with actors who are in a similar position (e.g., TAs) within the social system of interest. **Cohesion** is an important concept associated with the network as a whole. A perfect cohesion implies that all actors of the social system are connected through some kind of social tie. **Density** is a measure of cohesiveness and measures the proportion of all possible ties that are realized within the system. If the density is high there are considerable interactions among different members. Density in itself is, however, an incomplete measure of the network. Clearly, density will be expectedly low when there are a large number of actors. A large value of density could arise because a single actor has a large number of ties. In a classroom system, there could be a popular instructor, TA, or student from whom many students may be obtaining information. These actors will increase the density measure, but do not necessarily contribute to the overall cohesiveness of the class. **Degree centralization score** (or centralization score) looks at the extent to which one actor in a network is holding all the ties and can vary between zero and one. For example, if the centralization score is unity then there is a single actor who is connected to all other actors. At the other extreme, a score of zero indicates a perfectly diffused network where all the actors have the same number of ties. A simultaneous evaluation of density and centrality provides vital clues towards network cohesion. A network with a high density and centralization score is less cohesive than a network with the same density score but a lower centralization score (Prell 2011). A network with a high density and centralization score where the central actor is a formal information source (i.e., instructor, textbook) is preferred in classroom settings. In a similar manner, less dense and more diffuse networks are preferred when evaluating teamwork and collaboration efforts among students. As can be seen, SNA provides a useful set of tools to evaluate the effectiveness of classroom and group work.

Table 1. Various centrality measures used in Social Network Analysis.

Centrality Metric	What does it Measure?
In-Degree	How many people come to this person for information?
Out-Degree	How many people does this person go to for information?
Betweenness	How likely (or how many times) is this person between two other people in network?
Closeness	How fast can this person get information to others in network?
Eigenvector	How well is this person connected to other well connected persons in network?

Illustrative Study

CE 3354: Engineering Hydrology is an early junior level class that is required for the four year Bachelors of Science in Civil Engineering (BSCE) and a five-year integrated Masters of Environmental Engineering (MSEE) degree offered at Texas Tech University. The course is the first water-related design course in the curriculum and covers traditional concepts related to watersheds, water budgets, hydrologic processes (rainfall,

initial abstractions, infiltration, evapotranspiration, and runoff), river and reservoir routing, and basic concepts of groundwater hydrology. The course also provides the first design experience within the curriculum and requires students to work on a team project which accounts for 30% of the overall grade. The design project is assigned roughly a month after class begins and is due at the end of the semester. Students are semi-randomly divided into teams of four or five members. The project provides an opportunity for the students to apply certain concepts learned in the class (e.g., watershed delineation, flood routing) in a "real-world" like setting. The project also expands the coverage of certain concepts, such as the time-distribution of design storms, which are only cursorily covered in the lecture. The students make use of the industry-standard HEC-HMS software (USACE, 2015) for the project. The projects generally require the students to evaluate and pick the best (from a hydrologic standpoint) land-development (urbanization) option among several potential alternatives and/or require students to evaluate the adequacy of detention structures to minimize flooding hazards downstream. The students are required to work in groups, keep track of their effort using time-sheets (that are signed by all team members), and write technical memos during the course of the semester. The class is designated as a writing-intensive course and students are provided feedback on the technical memos without penalty to improve their writing skills. These memos are then aggregated with appropriate additions and submitted as a final report which is graded. Each team submits one final report which is awarded an average grade. This average, in conjunction with the reported time and effort, is used to assign project grades to individual group members.

Project teams are identified early in the semester to ensure that students become aware of the working habits of their team members. A short team-building exercise is conducted in class to help students brainstorm and learn how other members on the team think. To further develop teamwork and collaboration skills, students also work collaboratively on several homework assignments. Students are required to make individual submissions and must explicitly acknowledge the help and assistance obtained from other classmates and other sources. Students are also encouraged to sit together as a team during class hours and work together on in-class assignments. This project-based seating is, however, not strictly enforced in the class. Semi-random assignment of team-members based on overall GPA and grade in Fluid Mechanics is used to increase diversity of teams and spread out students of different academic preparation, backgrounds, gender, and race. However, collaborations outside the project team are not discouraged for other class activities. Students can form their own study groups to study for exams and solve homework assignments.

A short-survey was conducted roughly two months into the Fall 2013 semester. Two sections of the class were taught separately by the authors of this paper and each section had a separate TA. While the instructors followed the same textbook and roughly followed the same sequence of topics, the homework assignments and in-class lectures were not synchronized by the instructors. Furthermore, the instructors were free to employ their own pedagogic style. Professor 1 mostly provided hints to solve problems or provided an overview of necessary steps, whereas Professor 2 walked students through the various steps involved. At the time the survey was administered, the project teams had been identified and had performed a few tasks (e.g., watershed delineation and synthetic hyetograph generation) for their final project in both sections. Also, students in both sections had completed several homework assignments where they had the option to work collaboratively, and had taken a test for the class.

The survey was short and administered at the beginning of the class, without any prior notice to the students, and consisted of the following question:

> *List the names of students as well as other resources (TA, Professor, Textbook, Notes) that you get information from to solve homework problems, study for exams, work on the project for CE 3354 Engineering Hydrology. (Please be as complete as possible).*

Students were given an opportunity to provide additional comments. Student names were coded to preserve their anonymity. The gender, ethnicity, and overall GPA of each student were ascertained and summarized for additional analysis. Most students in the class were either in the four-year BSCE or five-year integrated MSEE program,

except for one Engineering Technology student who was taking this class as a technical elective. Figure 1(A) shows the incoming GPA for each section and there was no statistical difference between the two sections. While both sections had similar class sizes (25 and 24 students in Sections 1 and 2, respectively), the gender and ethnicity of the students varied considerably in both classes; these characteristics are summarized in Table 2. In addition, a majority of the students belonged to the same cohort. Similarly, many students had taken earlier engineering classes together and belonged to the same student organizations. As such, many of these students had prior interactions and had established informal coalitions (e.g., study groups) before entering this course. However, the survey did not ask any questions to determine these prior interactions. The required data were compiled separately for each section in a spreadsheet and SNA was carried out using the R programming environment (RDC Team 2011).

Table 2. Gender and ethnic characteristics of students in CE 3354 Hydrology Class.

Gender/Race	Section 1	Section 2
Male	19	20
Female	6	4
Hispanic	9	1
White	15	19
Black	0	2
Other	1	2

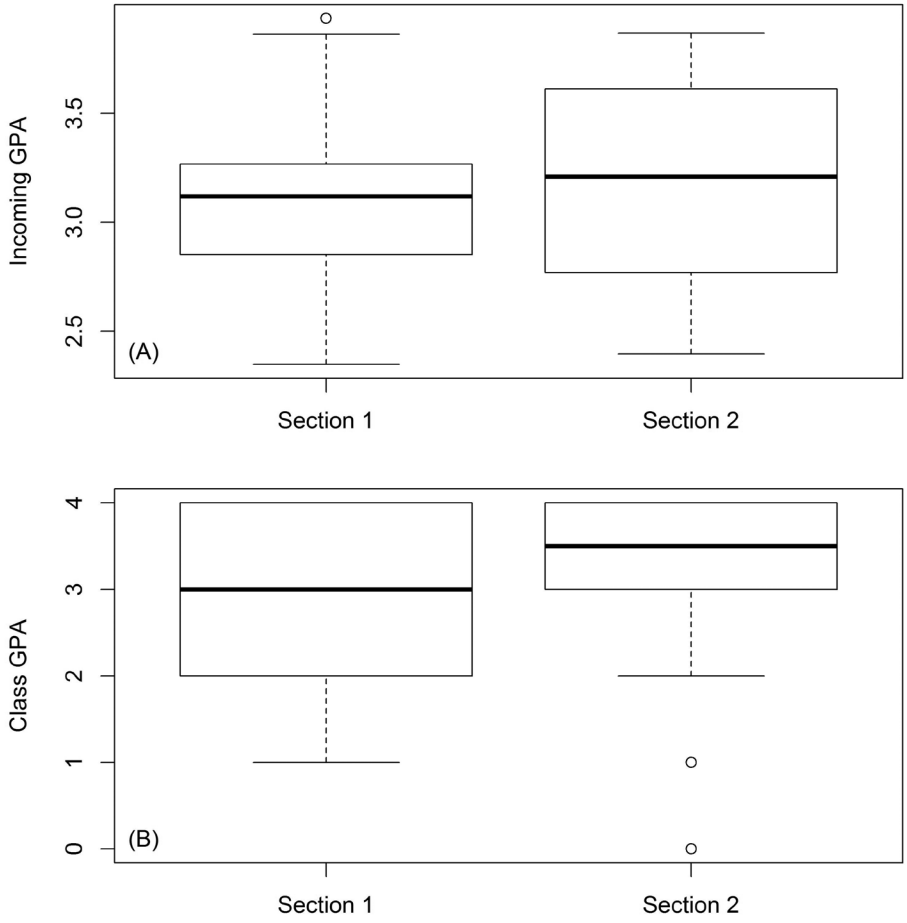


Figure 1. (A) Distribution of cumulative GPAs of students entering CE 3354: Engineering Hydrology. **(B)** Distribution of student grades in CE 3354: Engineering Hydrology.

Results and Discussion

How Are Students Seeking Information from Formal Information Sources?

The ego networks for professors and TAs are shown in Figure 2. Approximately 10 students in each class went to their respective professors for information outside the class. In addition, five students from Section 1 sought help from the professor of Section 2 (Professor 2). It is evident there were greater (two-way) interactions among students who visited Professor 1 outside the classroom than those who visited Professor 2. This situation can be attributed to the pedagogic styles of instructors. The teaching style of Professor 1 required students to compare notes and solutions more frequently than Professor 2. These differences in pedagogy may also explain why very few students in Section 2 went to their TA (TA 2), while the TA for Section 1 (TA 1) was sought after by many students.

Are Students Reading Their Textbook and Referring to Their Notes?

The ego networks for the textbook and class notes as sources of information are depicted in Figure 3. The in-degree centrality for the textbook was 14 and 16 in Sections 1 and 2, respectively. The reliance on the textbook was higher than on professors and can perhaps be attributed to the fact that the course required students to pull out design information (e.g., curve numbers, Manning's coefficients) from the textbook to solve homework

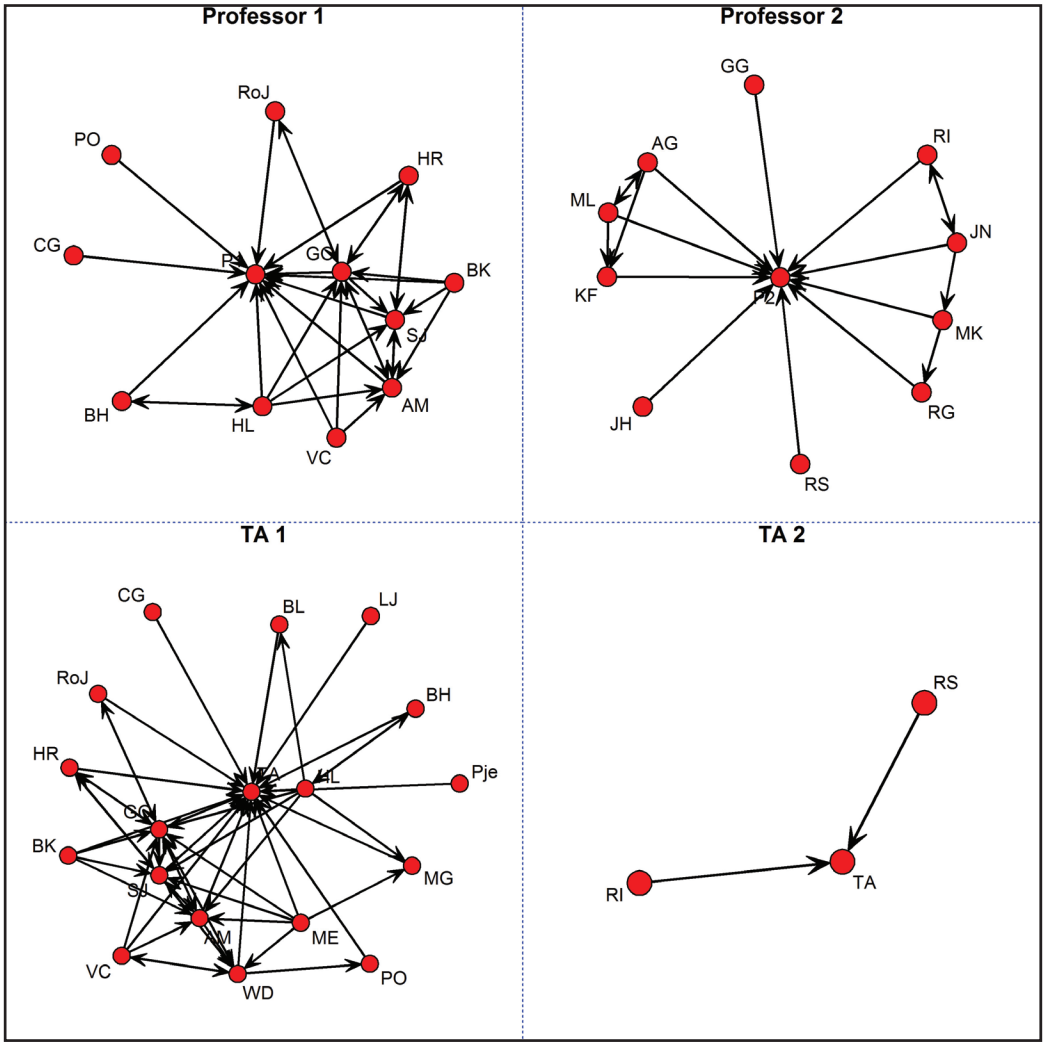


Figure 2. Ego network of professors and graduate teaching assistants.

problems. This result also indicates that nearly 44% of students in Section 1 and 33% in Section 2 did not refer to their textbook and possibly relied on other sources. Furthermore, 16% of students in Section 1 and 12.5% of students in Section 2 who did not rely on the textbook, sought help from the instructors; therefore, a significant number of students (over 20%) relied solely on class notes and/or did not indulge in additional reading outside of the classroom. The ties between students who indicated using the textbook for information are low. This result indicates that students are probably reading their textbook on their own rather than in group study sessions.

Professor 1 mostly used a non-structured lecture style (i.e., devoid of PowerPoint or other electronic technologies) and provided printed problem statements on blank sheets that students used to take notes during class. On the other hand, Professor 2 mostly used the Document Cam in class and provided these structured in-class lecture notes online (via Blackboard) after the completion of the lecture. Nearly 2/3rd of the class in Section 2 indicated they relied on these structured in-class notes, while only 36% of the students relied on in-class notes (that they took on their own during the class) in Section 1. Again, the lack of structured notes increased collaborations between students in Section 1 as they had to compare notes to ensure they had all the necessary information captured from the lecture. While the lack of structured notes allowed for greater collaboration, it is important to note that nearly 66% of the class did not rely

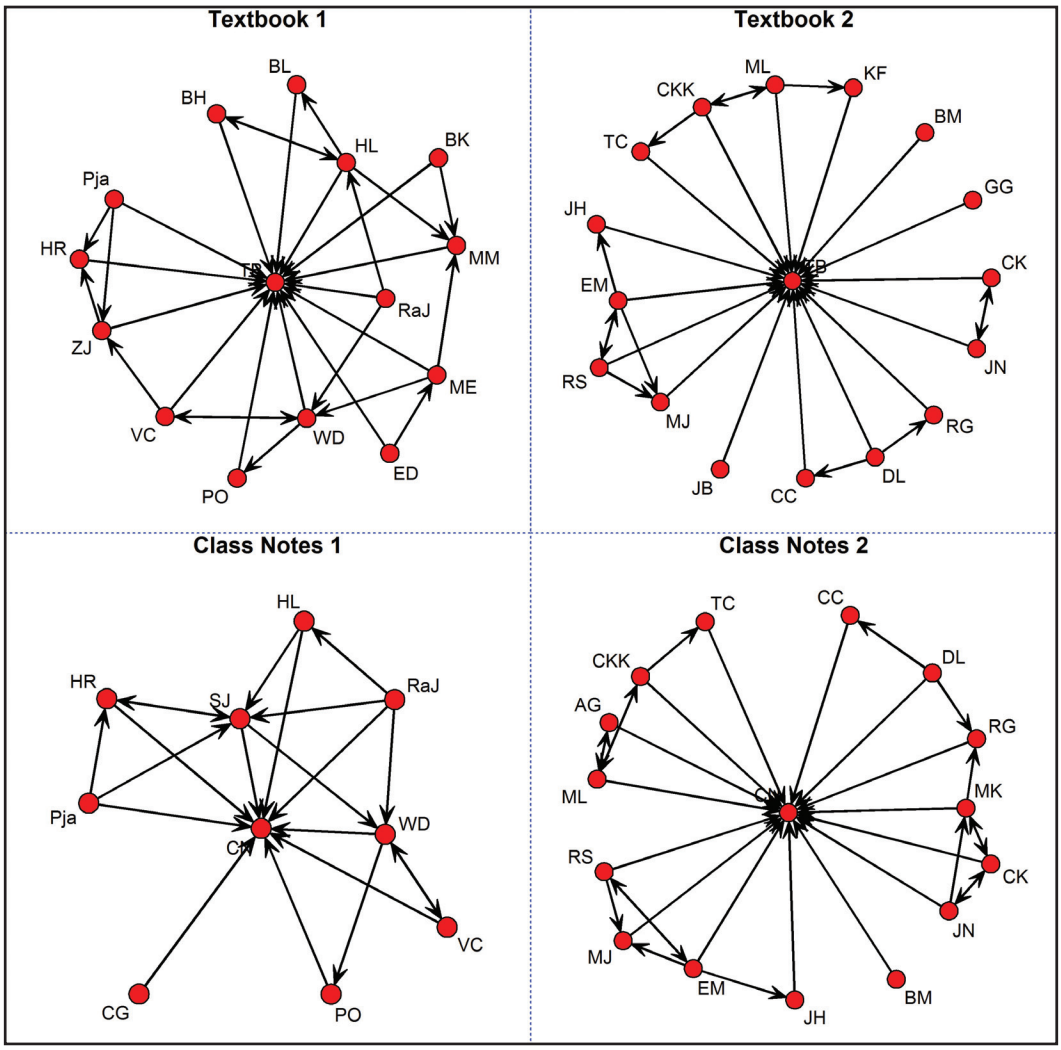


Figure 3. Ego network of textbook use and class notes use.

on class notes in Section 1 and had to depend on other sources of information.

Are We Creating Connected and Disconnected Students in Class?

Connected students are those sought by other students in the class for information. The degree of in-centrality can be used to identify such connected students and their ego-networks (Figure 4) and help identify students who rely on them. All these connected students made use of formal information resources and made regular appearances during office hours. The eigenvector centrality values (ranging from a high interaction value of 1 to a low interaction value of 0) presented in Figure 4 also indicate that these students were interacting among each other to a relatively higher degree. Interestingly, all the connected students identified were in Section 1 which was far less-structured than Section 2.

There were a total of nine students (six in Section 1 and three in Section 2) that had an in-degree centrality of zero. These students were not sought out by their classmates for information. On the other hand, five students (two in Section 1 and three in Section 2) had an out-degree of zero indicating that while these students were sought by some students for information, they did not rely on other students to get their information. These students largely worked independently and did not often collaborate with their classmates. A total of two students (both in Section 1) had both their in- and out-degree centralities as zero, indicating that they were

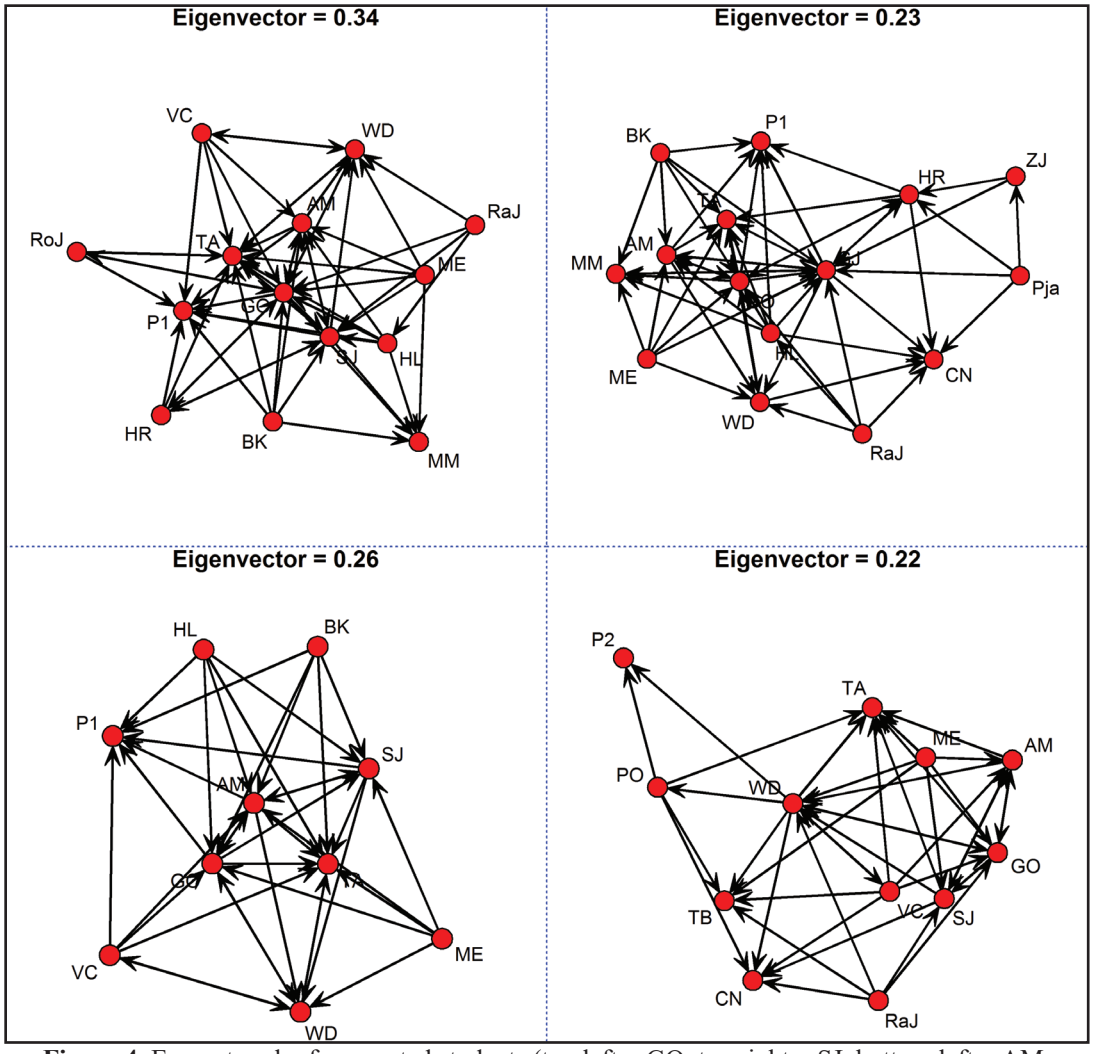


Figure 4. Ego network of connected students (top left = GO, top right = SJ, bottom left = AM, bottom right = WD) and their eigenvector centrality measures.

completely disconnected and did not interact with others in the class even on project tasks that required teamwork. These disconnected students were mostly present in the far less-structured Section 1.

Identification of disconnected students is essential to correct "freerider" problems in teams. Discussing students' situations and reasons for their inability to work with groups, and discussion with other team members with regards to their inclusiveness efforts, provide suitable interventions to improve team efforts. In a similar fashion, information content of connected students must be carefully evaluated throughout the semester to ensure that "misinformation" is not accidently propagated through the class. Scheduling regular meetings with these students can help get the pulse of the class and provide the right structure and guidance to ensure information presented in the class is being transferred effectively throughout. It is perhaps best to use TAs for this purpose to avoid any semblance of favoritism to students.

Are Students Working with Their Assigned Teams or Self-Selecting Study Groups?

For the class design project, both sections divided students into six groups of four students each (one group in Section 1 had five students). The assignment of teams was done early in the semester to improve teamwork and collaboration within the groups. The degree value for each team was computed by quantifying the number of times the students within a group identified other members of the group as someone they went to for information, summing up all such interactions, and dividing the result by the number of all possible interactions. For example, a team of four would have 12 potential interactions as each member can interact with three other teammates. If the total number of actual interactions was six, then the degree of the project team was 50%. The results presented in Table 3 indicate minimal interactions between teammates, suggesting that students primarily created study groups prior to the design team assignments and mostly worked with them unless required to work with other teammates. Extremely low group interactions were seen when disconnected students were paired with connected students. This result corroborates an earlier study by Vaquero and Cebrian (2013) who found evidence of "rich-club phenomenon" in unstructured classes. The "rich club phenomenon" states that certain students have a disproportionately high level of information when compared to others in the class and some students find it hard to fit in with these popular students, perhaps due to large differences in their academic preparation, and essentially become disconnected. The group degree of centrality measures could also be used to test the veracity of time-sheets submitted by students and also discuss the importance of teamwork with students and find ways to reduce dysfunctionality and increase collaboration.

How Does Collaboration Correlate with Class Grade?

The in-degree and out-degree centralities were plotted against the final GPA that students received in the class (Figure 5). In the less structured Section 1, the median in-degree centrality increases exponentially with GPA. Clearly, students with higher GPA are sought after by students who eventually received lower GPA in the class. This trend is not as evident in Section 2 which was more structured; and, as seen previously, students relied more on the instructor and the class notes provided to them. The median out-degree centrality also increases with eventual GPA in both sections. Furthermore, from Figure 1(B), it can be seen that while the individual GPA within each section varied, the less structured Section 1 showed less instances of students being left behind. In Section

Table 3. Degree measures for project teams (Number of Interactions/Possible Number of Interactions).

Group	Section 1	Section 2
1	4/12	3/12
2	6/20	5/12
3	1/12	3/12
4	3/12	9/12
5	7/12	1/12
6	1/12	5/12

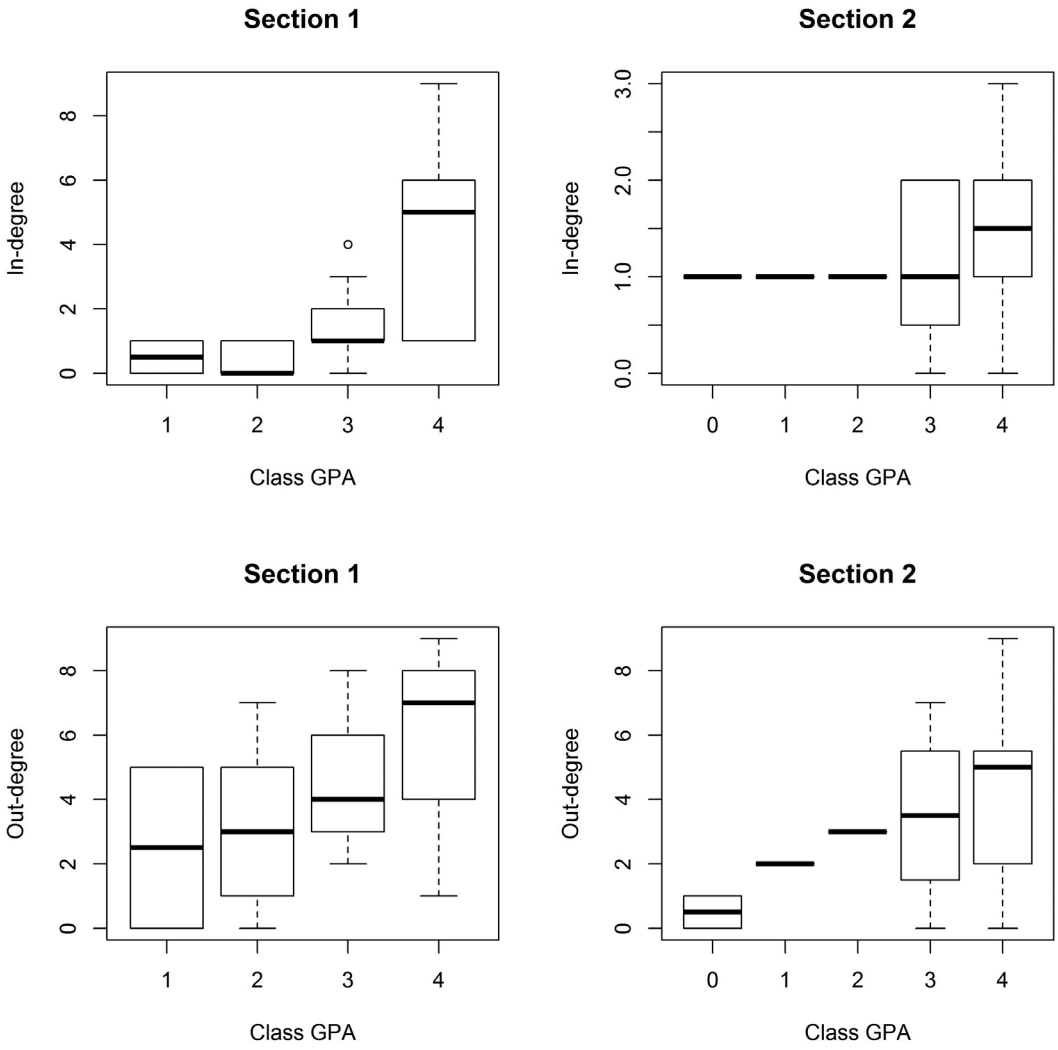


Figure 5. Variation of in-degree and out-degree by class grade.

2, while the average class GPA was higher, there were outliers of students at the lower grade level. These results suggest that collaborative learning correlates well with better class grades and corroborates earlier empirical findings that indicate collaboration among students improves learning the objectives of the course (Prince 2004).

Summary and Conclusions

This study presented the use of SNA in a project-based introductory junior-level engineering hydrology course to understand where students were getting their information and to identify students who were disconnected. Furthermore, insights into the efficacy of different teaching methods (structured vs less structured) and how they impacted student learning were discussed in the results of this study. Students were equally likely to seek information from the professor regardless of teaching style; however, assistance from the graduate TAs was sought to augment professor instruction in less structured classes. Conversely, the independence promulgated by the less structured class resulted in more mature and self-selected collaborations, but created extremities in classroom interactions (i.e., connected students and disconnected students). Not all students relied on the textbook for information, even though the homework assignments required students to pull

design information directly from the textbook. The design team assignments helped students who relied solely on their professor more so than students who were well connected in the classroom. Regardless of teaching method, students with greater collaborations resulted in higher individual final class grades.

SNA in the classroom has proven to be a useful tool for understanding the needs of the students and for recognizing which students may be falling behind their classmates. It aids in identifying specific weaknesses in information distribution and dissemination. Secondarily, SNA can be used to adjust teaching styles to fit the learning style of the class as a whole. Furthermore, implementing SNA along the entire undergraduate curriculum can aid in evaluating when students reach a level of independence that brings about more meaningful collaborations.

Acknowledgements

The authors would like to acknowledge and thank the students who chose to participate in this survey. We would also like to thank the two anonymous reviewers and the editors for the constructive comments and suggestions which have greatly improved the presentation of the material.

Author Bio and Contact Information

E. Annette Hernandez (corresponding author) is an Associate Professor in the Department of Civil, Environmental, and Construction Engineering. Her area of research includes sustainable watershed management through the development of scientifically rigorous consensus-driven decision making tools, the role of uncertainty regarding conceptualizations and climate change, and technology transfer across international boundaries. Dr. Hernandez is faculty advisor for Engineers Without Borders and as such has led students into Panama and Brazil. She has published research on inclusive management of water resources. Dr. Hernandez has been recognized nationally for her efforts in teaching and research. She may be reached at the following email: annette.hernandez@ttu.edu, or at her mailing address: Texas Tech University, Box 41023, Lubbock, TX 79409-1023.

Venkatesh Uddameri is the Director of the Texas Tech University Water Resources Center and also a professor in the Department of Civil, Environmental, and Construction Engineering. His research interests are in the broad area of sustainable water resources management with a particular emphasis on food-water-energy nexus in groundwater dependent environments. He currently serves as the associate editor for the *Journal of American Water Resources Association* and has recently co-authored a textbook - GIS and Geocomputation in Water Resources Science and Engineering published John Wiley and Sons (ISBN: 978-1-118-35413-1). Dr. Uddameri may be contacted at venki.uddameri@ttu.edu or at Texas Tech University, Box 41023, Lubbock, TX 79409-1023.

References

Amory, A. 2007. Game object model version II: A theoretical framework for educational game development. *Educational Technology Research and Development* 55(1): 51-77.

Borgatti, S.P., M.G. Everett, and J.C. Johnson. 2013. *Analyzing Social Networks*. SAGE Publications Limited, Los Angeles, CA.

Boud, D., R. Cohen, and J. Sampson. 2014. *Peer Learning in Higher Education: Learning From and With Each Other*. Routledge, New York, NY.

de Laat, M., V. Lally, L. Lipponen, and R.J. Simons. 2007. Investigating patterns of interaction in networked learning and computer-supported collaborative learning: A role for Social Network Analysis. *International Journal of Computer-Supported Collaborative Learning* 2(1): 87-103.

Dunlap, J.C. 2005. Problem-based learning and self-efficacy: How a capstone course prepares students for a profession. *Educational Technology Research and Development* 53(1): 65-83.

Felder, R.M., G.N. Felder, and E.J. Dietz. 1998. A longitudinal study of engineering student performance and retention. V. Comparisons with traditionally-taught students. *Journal of Engineering Education* 87(4): 469.

Freeman, L.C. 2005. Graphic techniques for exploring social network data. In: *Models and Methods in Social Network Analysis,* P.J. Carrington, J. Scott, and S. Wasserman (Eds.). Cambridge University Press, New York, NY, pp. 248-269.

Freeman, S., S.L. Eddy, M. McDonough, M.K. Smith, N. Okoroafor, H. Jordt, and M.P. Wenderoth. 2014. Active learning increases student performance in science, engineering, and mathematics. *Proceedings of the National Academy of Sciences* 111(23): 8410-8415.

Huang, S. and E. Pierce. 2015. The impact of a peer learning strategy on student academic performance

in a fundamental engineering course. In: *Frontiers in Education Conference (FIE).* IEEE (Ed.). pp. 1-4.

Janicki, T.N., D. Fischetti, and A.T. Burns. 2007. Incorporating Real World Projects and Emerging Technologies into One MIS Capstone Course. Available at http://proc.isecon.org/2006/2712/ISECON.2006.Janicki.pdf. Accessed July 11, 2016.

Johnson, P.A. 1999. Problem-based, cooperative learning in the engineering classroom. *Journal of Professional Issues in Engineering Education and Practice* 125(1): 8-11.

Knight, D.W., L.E. Carlson, and J.F. Sullivan. 2007. Improving engineering student retention through hands-on, team based, first-year design projects. In: *Proceedings of the International Conference on Research in Engineering Education,* ASEE (Ed.). Honolulu, HI.

Knoke, D. and S. Yang. 2008. *Social Network Analysis* (Vol. 154). Sage Publications, Limited, Thousand Oaks, CA.

Lee, J. and C.J. Bonk. 2016. Social network analysis of peer relationships and online interactions in a blended class using blogs. *The Internet and Higher Education* 28: 35-44.

Liu, Y. and Y. Dou. 2015. Design of an industry-tied and team-oriented course for mechanical engineering senior. In: *2015 ASEE Southeast Section Conference,* ASEE (Ed.).

Martınez, A., Y. Dimitriadis, B. Rubia, E. Gómez, and P. De la Fuente. 2003. Combining qualitative evaluation and social network analysis for the study of classroom social interactions. *Computers & Education* 41(4): 353-368.

McGann, S. and M. Cahill. 2005. Pulling it all together: An IS capstone course for the 21st century emphasizing experiential and conceptual aspects, soft skills and career readings. *Issues in Information Systems* 6(1): 1-7.

Mills, J.E. and D.F. Treagust. 2003. Engineering education—Is problem-based or project-based learning the answer. *Australasian Journal of Engineering Education* 3(2): 2-16.

Oakley, B., R.M. Felder, R. Brent, and I. Elhajj. 2004. Turning student groups into effective teams. *Journal of Student Centered Learning* 2(1): 9-34.

Oakley, B.A., D.M. Hanna, Z. Kuzmyn, and R.M. Felder. 2007. Best practices involving teamwork in the classroom: Results from a survey of 6435 engineering student respondents. *Education, IEEE Transactions on Education* 50(3): 266-272.

Otte, E. and R. Rousseau. 2002. Social network analysis: A powerful strategy, also for the information sciences. *Journal of Information Science* 28(6): 441-453.

Pinheiro, C.A.R. 2011. *Social Network Analysis in Telecommunications* (Vol. 37). John Wiley & Sons, Hoboken, NJ.

Prell, C. 2011. Some basic structural characteristics of networks. In: *Social Networks and Natural Resource Management: Uncovering the Social Fabric of Environmental Governance.* Ö. Bodin & C. Prell (Eds.). (Vol. 3). Cambridge University Press, Cambridge, UK, pp. 29-43.

Prince, M. 2004. Does active learning work? A review of the research. *Journal of Engineering Education* 93(3): 223-231.

Russell, J., B. Russell, and W.J. Tastle. 2005. Teaching soft skills in a systems development capstone class. *Information Systems Education Journal* 3(19): 3-19.

Shelby, R., E. Patten, F. Ansari, L. Pruitt, G. Walker, and J. Wang. 2013. Implementation of leadership and service learning in a first-year engineering course enhances professional skills. *International Journal of Engineering Education* 29(1): 1-14.

Shuman, L.J., M, Besterfield-Sacre, and J. McGourty. 2005. The ABET "professional skills" -Can they be taught? Can they be assessed? *Journal of Engineering Education* 94(1): 41-55.

Stump, G.S., J.C. Hilpert, J. Husman, W.T. Chung, and W. Kim. 2011. Collaborative learning in engineering students: Gender and achievement. *Journal of Engineering Education* 100(3): 475.

RDC Team. 2011. A language and environment for statistical computing. *R Development Core Team.* Retrieved from https://www.r-project.org/. Accessed June 2, 2016.

Tonso, K.L. 2006. Teams that work: Campus culture, engineer identity, and social interactions. *Journal of Engineering Education* 95(1): 25.

USACE. 2015. Hydrologic Engineering Center - Hydrologic Modeling System (HEC-HMS). Retrieved from http://www.hec.usace.army.mil/software/hec-hms/. Accessed June 2, 2016.

Vaquero, L.M. and M. Cebrian. 2013. The rich club phenomenon in the classroom. *Scientific reports* 3.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 98-119, August 2016

Economic Value of Nature and Ecosystems in the Delaware River Basin

*Gerald J. Kauffman

University of Delaware Water Resources Center, Newark, Delaware

Abstract: The Delaware River basin is a valuable ecological and economic resource that supplies drinking water to five percent of the population of the United States. Located in Delaware, New Jersey, New York, and Pennsylvania, the basin supplies drinking water to the nation's first (New York City) and seventh (Philadelphia) largest metropolitan economies, and supports the largest freshwater port in the world while sustaining a recovering anadromous shad and striped bass fishery. The Delaware basin contributes over $22 billion in annual economic activity from potential Marcellus Shale gas extraction ($425 million), recreation ($1.2 billion), fish/wildlife ($1.5 billion), public parks ($1.8 billion), water quality ($2.5 billion), navigation ($2.6 billion), agriculture ($3.4 billion), water supply ($3.8 billion), and forest ($5.1 billion) benefits. The value of natural goods and services from Delaware basin ecosystems is $21 billion ($2010) with net present value of $683 billion with contributions from Delaware ($2.5 billion), New Jersey ($6.6 billion), New York ($3.5 billion), and Pennsylvania ($8.6 billion). The Delaware basin supports 600,000 direct/indirect jobs with $10 billion in wages in the coastal, farm, ecotourism, water/wastewater, ports, and recreation industries. This research demonstrates that the Delaware River basin provides significant economic benefits to the region and is worthy of priority investments by elected officials and decision-makers to protect and restore these natural resources.

Keywords: *watershed, river basin, ecological economics, ecosystem services*

Many economists agree that water is an undervalued resource. The dilemma inherent in defining the economic value of water goes back two centuries to the 1776 *Wealth of Nations* when Adam Smith described the "diamond-water paradox" (U.S. Environmental Protection Agency (U.S. EPA) 2012). If water is more valuable to society than a precious gem, then why is drinking water sold for a penny per 100 gallons or not valued at all as an ecological resource in the river? Society tends to underprice water based on its marginal value for single uses (i.e. drinking water) and not consider the full value for its myriad of uses. If water is society's most valuable chemical, then the Delaware River, with a mean annual flow of 2.7 trillion gallons, is an invaluable economic resource.

Natural systems have substantial economic value in terms of ecosystems goods and services. The estimated annual value of global natural capital ranges from $33 trillion (Costanza et al. 1997) to $125 trillion (Costanza et al. 2014). Ingraham and Foster (2008) concluded the ecosystem services value of the U.S. National Wildlife Refuge System was $27 billion. The New Jersey Department of Environmental Protection (NJDEP) found the value of habitat in the Garden State was $20 billion (Mates 2007). The watersheds and ecosystems in the First State of Delaware are valued at $6.7 billion annually (Narvaez et al. 2012). The economic value of the Barnegat Bay in New Jersey is at least $2.3 billion (Kauffman and Cruz-Ortiz 2012).

Watersheds have significant economic value and restoration of these resources can result in favorable benefit-cost ratios for society. The University of Maryland reported in 1988 that the Chesapeake Bay was worth $678 billion and with the rise in the cost of living the value of the bay exceeded $1 trillion (Chesapeake Blue Ribbon Panel 2003). The Brookings Institution (Austin et al. 2007) estimated restoration of the Great

Lakes would cost $26 billion and provide tourism, fishing, recreation, property value, and water treatment benefits that would exceed $50 billion, a 2:1 benefit/cost ratio. The Everglades Foundation concluded the Comprehensive Everglades Restoration Plan (CERP) would provide $6 billion in benefits with a benefit/cost ratio of 4:1 and generate 443,000 jobs over 50 years (McCormick 2010).

Two decades ago, economists concluded the Delaware River and Bay in Delaware, New Jersey, and Pennsylvania possessed significant economic value. Latham and Stapleford (1987) estimated the Delaware Estuary within Delaware accounted for 10,500 jobs with $222 million in annual wages and each direct estuary job accounted for 2.2 indirect jobs. The Greeley-Polhemus Group (1993) estimated the Delaware Estuary supported 123,000 jobs, $4.3 billion in wages, $24 billion in sales, $25 million in sport fishing non-market value, $1 million in commercial fish landings, and a wetlands replacement value of $638 million.

Economic benefits are the maximum dollar value of goods and services that individuals are voluntarily willing to pay (WTP) for improved water quality (Cech 2005). In environmental economics, WTP measures how much people are willing to pay for a given good or service regardless of whether they actually pay or not (Goulder and Kennedy 1997). Consumer surplus is the area under the demand (marginal benefit) curve above its price (or value) measured by the difference between the amount individuals actually pay and the amount they are willing to pay for a benefit such as clean drinking water or enhanced fishing provided by improved water quality (Figure 1). That is, consumer surplus is the amount people are willing to pay above the price they pay for it (Thurston et al. 2009). If an individual is willing to pay $6.00 per 1000 gallons for drinking water and the price is $5.00, the consumer surplus is $1.00.

Research Objectives

The following research quantifies the multi-objective value of water *in toto* for its wide range of habitat, recreation, water quality, and water supply uses in the Delaware River basin along the East Coast of the United States. We update the 1990's economic analyses of the Delaware River and Bay and incorporate modern valuation methods from the emerging fields of ecological economics and ecosystem services to estimate the "21st century" value of the interstate Delaware basin in Delaware, New Jersey, New York, and Pennsylvania. The objective of this research is

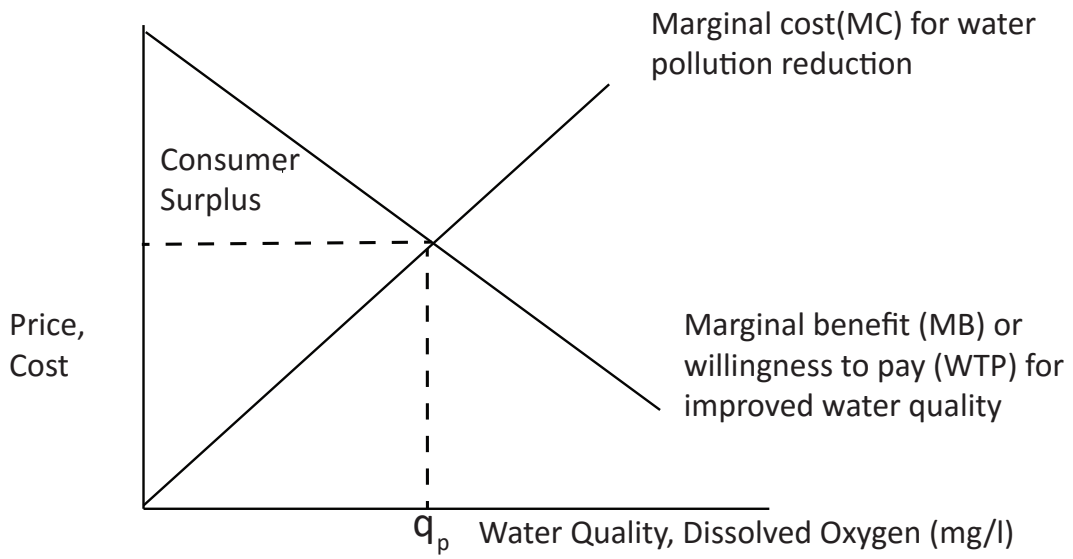


Figure 1. Consumer surplus as willingness to pay for improved water quality. Consumer surplus is the difference between the amount that individuals actually pay and the amount that they would have been willing to pay for improved water quality.

to estimate the economic value of the Delaware River basin in the aforementioned states by examining: (1) economic activity derived from market and nonmarket use and nonuse value of water supply, recreation, fishing, hunting, boating, ecotourism, agriculture, and navigation sectors, (2) ecosystem services value (natural capital) of goods and services provided by habitat such as wetlands, forests, farms, and open water, and (3) jobs/wages directly and indirectly associated with the river, bay, and tributaries.

The Delaware River Basin

For over four centuries, the Delaware River has been utilized for economic gain. Henry Hudson discovered the bay off Cape May in August 1609 during an unsuccessful quest to find an inner trade route to Asia for the Dutch East India Company. When William Penn founded Philadelphia in 1682 seeking refuge from religious persecution in Europe, he also found a safe harbor between the Delaware and the Schuylkill in a colony rich with lumber, fertile land, beaver pelts, and, in later centuries, coal and oil. By the 18th century, prosperous Philadelphia Quaker merchants established triangular trade route connections to Europe and the Caribbean from their home port along the Delaware. During the American Revolution, Philadelphia was the largest city in the colonies and the third largest port in the British Empire after London and Liverpool. In 1790 Ben Franklin was so concerned about pollution that he willed funds to build the first municipal water system in the U.S. for Philadelphia.

The river as an economic engine kicked into high gear during the Industrial Revolution with hydropower and steam power. In 1802, the DuPont family searched up and down the Atlantic Seaboard and established gunpowder mills along the Brandywine River falls above Wilmington, one of the first industries in the Delaware Valley. Delaware River ports grew when anthracite coal was discovered in the Lehigh Valley in 1792 and steam railroads were built in the 1830s. By the Gay Nineties, every Philadelphia wharf had railroad access and steam ships made fast transatlantic trips. In 1895, the Corps of Engineers dredged the Delaware River to 26 ft. from the natural depth of 17 ft. (Economy League of Greater Philadelphia 2008).

By the end of the 19th century, the Delaware River supported the largest commercial American shad and sturgeon fishery along the Atlantic coast, highlighting an annual spring rite that fascinated artists like Thomas Eakins and the poet Walt Whitman. The sturgeon was such a lucrative fish that boom town Caviar near Greenwich, New Jersey processed the roe for worldwide export. By the 1880s, 1,400 sailing vessels harvested 22 million lbs. of oysters from the Delaware Bay. In 1886, nationally famous hotels in Gloucester, New Jersey served 10,000 planked shad dinners at events that resembled modern day blue crab feasts. In 1896 over 14 million lbs. of shad were caught with a value of $400,000 ($10 million in $2008). Also in 1896, a fisheries report to the Governor of Pennsylvania listed the catch of a 76 lb. striped bass just across the river from Philadelphia in Gloucester, New Jersey (Kauffman 2010).

During the First World War, the Delaware became known as the "Clyde of America," with ship building that rivaled its Scottish cousin. In 1912, Philadelphia manufactured 5% of all goods in the United States and exported coal, iron, cotton, leather, grain, lumber, tobacco, and gunpowder. By 1914, the Panama Canal opened access from the East Coast to the Hawaiian sugar cane fields and Philadelphia refined 1/6 of all the sugar in the United States (Kauffman 2010).

The Delaware River ship channel was deepened to 41 ft. in 1941 and the port economy boomed during World War II as the Philadelphia Navy Yard employed 40,000 workers who built 53 ships and repaired over 500 vessels. After the war, the "Arsenal of America" manufacturing base declined due to waning demand for Pennsylvania coal and Lehigh Valley steel. In 1995, the U.S. Navy closed the Philadelphia Navy Yard and decommissioned the ghost fleet.

In 1931 and 1954, the U.S. Supreme Court issued decrees authorizing New York City to divert 800 million gallons per day from three reservoirs in the upper Delaware basin to the Hudson River basin. The Delaware River supplies half of the city's drinking water via an 85-mile long, 13-feet diameter concrete aqueduct running from the Catskill reservoirs to the five boroughs.

By 2010, a billion gallons per day were withdrawn from the Delaware basin to sustain the region's economy and supply drinking water to the 1st (New York City) and 7th (Philadelphia) largest metropolitan economies in the U.S. In 1965, Congress formed the Delaware Water Gap National Recreation Area that receives 5 million visits annually, the 8th most visited unit in the National Park System. The Delaware River is the largest freshwater port in the world yet supports a recovering American shad and striped bass fishery. After the turn of the 21st century, new horizontal drilling and hydraulic fracturing technology kicked off the Marcellus Shale natural gas drilling boom that threatened to take place in the upper 36% of the upper Delaware basin in New York and Pennsylvania.

Over the last half century, Federal, state, and local governments, nonprofits, and the private sector have focused on restoring the Delaware basin (Figure 2). In 1961, President John F. Kennedy and the governors of Delaware, New Jersey, New York, and Pennsylvania signed the 100-year Delaware River Basin Compact (DRBC) as the first ever Federal-state watershed accord. Four years before Congress passed the Clean Water Act, the DRBC in 1968 issued waste load allocations to reduce pollutant discharges from over 80 wastewater treatment plants. In 1996, Congress designated the Delaware Bay as one of only 28 National Estuary Programs in the United States and the Partnership for the Delaware Estuary was established to implement a Comprehensive Conservation and Management Plan. In 2011, the DRBC celebrated the 50th anniversary of its founding by JFK and the four governors (Kauffman 2010).

The Delaware basin occupies 12,769 square miles where Delaware, New Jersey, New York, and Pennsylvania cover 8%, 23%, 20%, and 49% of the basin, respectively. The basin covers 50% of Delaware and is home to 74% of the First State's population. The basin covers 40% of New Jersey and holds 22% of the Garden State's population. The basin covers 5% of New York but holds less than 1% of the Empire State's population. The basin covers 14% of Pennsylvania yet is home to 43% of the Keystone State's population. The Delaware basin drains just 0.4% of the continental U.S. yet supplies drinking water to 5% of the nation's population.

In 2010, 8.2 million people lived in the basin including 640,000 people in Delaware (9%), 2,300 in Maryland, 1,950,000 in New Jersey (24%), 125,000 in New York (2%), and 5,530,000 in Pennsylvania (66%). An additional 8 million people in New York City and northern New Jersey draw drinking water from the Delaware River via interbasin transfers. The Delaware basin population exceeds 8.2 million, which as a single jurisdiction would be the 12th most populous state in the nation after New Jersey but ahead of Virginia. Between 2000 and 2010, the basin population increased by 7% or a half million people (Figure 3). In 2010, nearly 3,500,000 people worked in the Delaware basin with 316,000 jobs in Delaware, 1,200 jobs in Maryland, 823,000 jobs in New Jersey, 70,000 jobs in New York, and 2,271,000 jobs in Pennsylvania.

Methods

This research quantifies the economic value of the Delaware basin based on three levels of analysis: (1) annual economic activity, (2) ecosystem goods and services, and (3) basin-related jobs and wages. The study area is defined by the hydrologic boundaries of the basin from the headwaters in the Catskill Mountains of New York to the mouth of the bay at Cape Henlopen, Delaware. We drew from data gathered by the U.S. Census Bureau, U.S. Bureau of Labor Statistics, U.S. Department of Agriculture, U.S. Forest Service, and U.S. Fish and Wildlife Service, and ArcGIS map layers of census blocks.

When primary ecological valuation data from the Delaware basin was not available, benefits transfer was utilized to translate data to the basin from other watersheds. Benefits transfer involves extrapolating the benefits calculated by previous studies in other sites to the watershed in question with appropriate adjustments (U.S. EPA 2012). Benefits transfer is relatively inexpensive to implement; however, it must be applied carefully to avoid redundancy and double-counting of benefits. The benefit transfer method is most reliable when the original site and study site are similar in location and population characteristics,

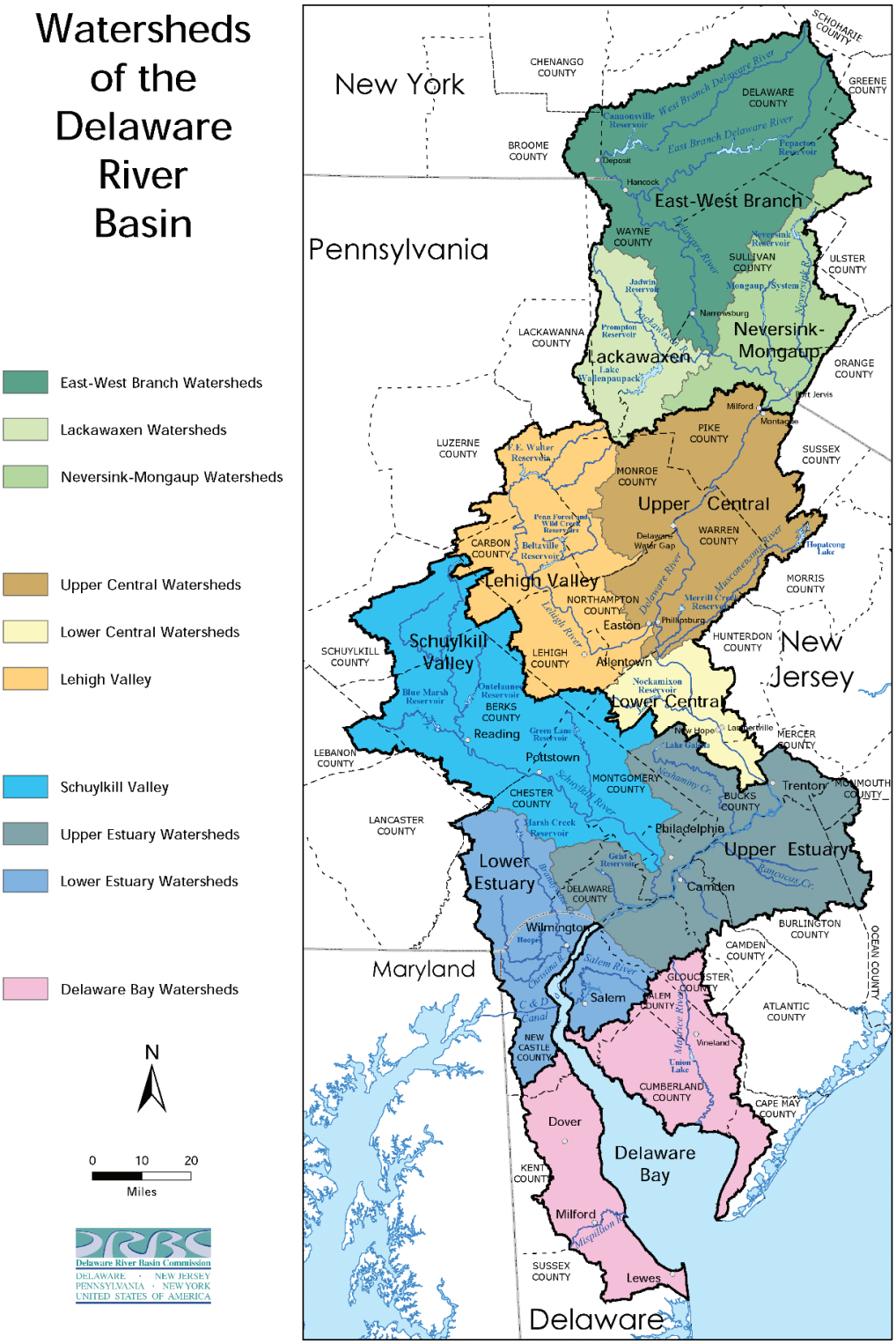


Figure 2. The Delaware River basin drains 13,000 square miles in Delaware, New Jersey, New York, and Pennsylvania (DRBC 2013).

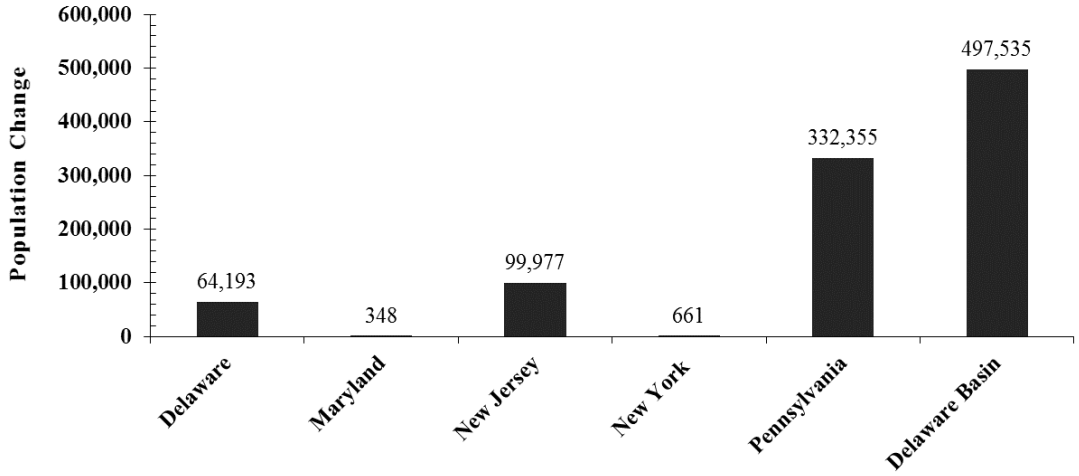


Figure 3. Population change in the Delaware basin for Delaware, New Jersey, New York, and Pennsylvania from 2000 to 2010 (U.S. Bureau of the Census 2010).

when water quality change is similar for the two sites, and when the original study used sound valuation techniques (WBCSD 2011). EPA employs benefit transfer to estimate nonmarket benefits of proposed water quality regulations from the Federal Clean Water Act. While it has limitations, the benefit transfer method was used here to estimate the economic benefits of the Delaware River basin.

To transfer data to the Delaware River, benefits are defined based on the population in the basin who benefit from improved water quality according to the following four steps. The first step in benefits transfer is to identify existing study values that can be utilized for the site in question (U.S. EPA 2012). The second step is to decide whether existing values are transferable based on several criteria. Is the benefit valued comparable to the value in the existing studies? The third step is to evaluate the quality of transferred studies. If the quality of the initial study is good, then the transferred value will be more accurate. The final step is to adjust original values to reflect the characteristics of the study site.

Benefits are converted to 2010 dollars based on the average annual change (3%) in the Consumer Price Index (CPI) in the Northeast Region from 1991-2010 as reported by the Bureau of Labor Statistics (2015) using the following formula. The discount rate of 3% was chosen as representative of the change in interest rates for borrowing over a 20-year period up to and including the base year of 2010.

$$B_{\$2010} = B_b(1+r)^t \quad (1)$$

Where:

$B_{\$2010}$ = Benefit in 2010 dollars;

B_b = Benefits estimated for the base year from the literature;

r = Annual change in Consumer Price Index (3%); and

t = Time in years counted from the base year through 2010.

Annual Economic Activity

For the first level of analysis, we estimated the annual value of agriculture, water quality, water supply, fishing, hunting, recreation, boating, ecotourism, and navigation in the basin from population, employment, industrial activity, and land use data. Total economic activity is defined as the sum of direct/indirect use, option, and nonuse values (Ingraham and Foster 2008). Direct use values are from natural goods such as drinking water, boating, recreation, and commercial fishing. Indirect values are benefits from ecosystems such as water filtration by forests and flood control/habitat protection from wetlands. Option

demand is public willingness to pay for benefits from improved water quality or knowing that the scenic value of the bay will be preserved. Nonuse (existence) values accrue to a public who may never visit the resource, but are willing to pay to preserve the resource for future generations.

Valuation methods include avoided cost which accounts for the costs if certain ecosystems are not present or lost such as the loss of wetlands that may increase economic flood damages. Replacement costs account for natural services lost and replaced by more expensive manmade systems. For instance, forests provide water filtration benefits that are replaced by costly water filtration plants. Net factor enhancement of income is derived where improved water quality water enhances fisheries and crabbing industries and, in turn, boosts jobs and wages. Travel cost involves visitors willing to pay to travel and visit ecosystems and natural resources for hunting, fishing, and birding. The hedonic pricing method involves residents willing to pay more for the higher property values along scenic bay and river coastlines. Contingent valuation surveys estimate how much more people may be willing to pay in source water fees.

Ecosystem Goods and Services

For the second level of analysis, we tabulated the natural capital or value of habitat in the watershed provided by wetlands, forests, farmland, and open water. Ecosystem services (ecological services) are provided by nature and represent benefits such as water filtration, flood reduction, and drinking water supply. Using GIS, we defined ecosystem areas using 2006 National Oceanic and Atmospheric Administration (NOAA) Coastal Services Center land cover data for wetlands, marine, farmland, forest, barren, urban, beach/dune, and riparian buffer habitats.

Ecological resources provide marketable goods and services such as timber, water, and fish and wildlife and equipment for recreation, hiking, and boating/kayaking. Natural capital are goods (commodities like water, crops, and timber that can be sold) and services (functions such as flood control, water filtration, and wildlife/fisheries habitat) provided by ecosystems such as wetlands, forests, farms, and open water. Ecosystems provide benefits for hunters, fishermen, boaters, and hikers who spend money to visit natural sites due to improved water quality.

Ecosystem services were estimated using value (benefits) transfer where published data and literature from other watersheds are transferred to the resource in question (the Delaware basin). We computed ecosystem services value by multiplying land use area by the ecosystem value ($/acre). Value transfer involves selecting data from published literature from another watershed and applying the dollar per acre values to the basin in question. While primary research data from the Delaware basin is preferable and used where available in this analysis, value transfer is a practical way to value ecosystems especially when in the absence of such data the worth of ecosystems have previously been deemed zero. Values from previous studies were adjusted to $2010 based on 3% change in CPI annually. Net present values were calculated based on an annual discount rate of 3% in perpetuity (over 100 years in the future).

Jobs and Wages

For the third level of analysis, we obtained employment and wage data from the U.S. Department of Labor, U.S. Census Bureau, and National Ocean Economics Program and calculated direct/indirect jobs by North American Industry Classification System (NAICS) code for shipbuilding, marine transportation/ports, fisheries, recreation, minerals, trade, agriculture, and other industries tied to the function of the rivers. Jobs/wages were scaled for each basin county from census block data. NAICS data were supplemented with farm jobs data from the USDA Agricultural Statistics Bureau, U.S. Fish and Wildlife Service ecotourism jobs data, and water supply and wastewater treatment utilities. Jobs and salaries were obtained from U.S. Bureau of Labor Statistics and U.S. Census Bureau data bases for the following scenarios:

- Total jobs in each county within the Delaware basin classified by NAICS industry code (formerly SIC code) and then grouped by census tract.
- Direct Delaware basin-related jobs such as water/sewer construction, living resources,

maritime, tourism/recreation, ports, environmental services, and water/wastewater management for each NAICS code by state and county within the basin boundary.

- Indirect jobs/wages from purchases of goods/services by direct jobs earners in the Delaware basin in the interlinked regional economy. Indirect jobs were estimated by multipliers of 1.2 applied to direct jobs and 0.8 to direct wages (Latham and Stapleford 1990), i.e. 100 direct jobs support 120 indirect jobs and direct wages of $1,000 provide $800 indirect wages.

Results

Annual Economic Activity

The economic value of the Delaware basin exceeds $22 billion/yr. from potential Marcellus Shale gas extraction ($425 million), recreation ($1.2 billion), fish/wildlife ($1.5 billion), public parks ($1.8 billion), water quality ($2.5 billion), navigation ($2.6 billion), agriculture ($3.4 billion), water supply ($3.8 billion), and forest ($5.1 billion) benefits (Table 1).

Recreation. The annual economic value of recreation in the Delaware basin is $1.2 billion. Swimming ($57 million), fishing ($24 million), boating ($6 million), and viewing ($5 million) benefits due to improved water quality total $92 million ($11.10 per person) for a basin population of 8,255,000 as scaled from Parsons et al. (2003). Travel cost demand data transferred from the Peconic Estuary (Johnston et al. 2002) indicate the annual consumer surplus for swimming, boating, fishing, and bird watching/wildlife viewing totals $211 million. Nine ski areas draw water (1 mgd) for snowmaking on 1,005 ac (407 ha) where the economic value ranges from $88 million (based on 1.9 million ski visits at a lift ticket rate of $45/day) to $325 million (based on scaled estimates from the Pennsylvania Ski Areas Association (2009)).

Paddling-based recreation such as canoeing, kayaking, and rafting in the Delaware basin involves 621,000 participants, $362 million in gear retail and trip sales, and 4,226 jobs based on scaled estimates by population from the Outdoor Industry Association (2006). River recreation economic output along the upper Delaware River and Delaware Water Gap National Recreation Area is roughly $41 million in $2010 based on Cordell et al. (1990) from the U.S. Forest Service and National Park Service. Thirty-seven canoe and kayak liveries along the Delaware, Lehigh, Schuylkill, and Brandywine Rivers lease watercraft to 225,000 visitors with earnings of $9 million/yr. assuming a daily rental fee of $40/person (Canoe and Kayak Liveries 2011). The National Marine Manufacturers Association (2010) found that New York, Delaware, Pennsylvania, and New Jersey ranked 3rd, 7th, 17th, and 23rd in the U.S., respectively, in expenditures for recreational powerboats which, scaled by population, totaled $395 million/yr. in the Delaware basin.

Water Quality. The annual economic value of water quality benefits in the Delaware basin is $2.4 billion. Willingness to pay to make Delaware Estuary water quality acceptable to the public was $658 million in $2010 for a basin population of 6,700,000, based on value transfer from the Chesapeake Bay where 43% of the population were users willing to pay $121/yr. and 57% of the population were nonusers willing to pay $38/yr. to restore the bay (Bockstael, McConnell, and Strand 1989). Loss of forests that occupy 53% of the Delaware basin would increase treatment costs for public water supplies by $63 million/yr. given that for every 10% increase in forest area, water treatment costs decline by 20% (Trust for Public Land and AWWA 2004). The value of wastewater treatment is $1.7 billion/yr. for National Pollutant Discharge Elimination System (NPDES) permit holders that discharge 1,180 mgd to the Delaware basin based on a mean wastewater rate of $4.00/1000 galions (Yagecic et al. 2014). While wastewater treatment may be construed as a cost, the waters of the Delaware River basin provide assimilative capacity to treat these wastes and the rate paid by the public is an estimate of what consumers are willing to pay for this resource use and is therefore counted as a benefit. The value of property within 2000 feet of the 143-mile shoreline will increase by 8% ($256 million or $13 million/yr. over 20 years) due to water quality improvements in the Delaware Estuary watershed (U.S. EPA 1973).

Water Supply. The annual economic value of water supplies in the Delaware basin is $3.8 billion.

The value of public drinking water supplies (Barr and Sayers 2013) in the Delaware basin (1,803 mgd) is $3.1 billion based on an average water rate set by public and private water purveyors of $4.78/1000 gallons (Corrozi and Seymour 2008). The annual value of storage (369 billion gallons) at 11 reservoirs in the Delaware basin is $145 million based on an estimated market price of $0.394/1,000 gallon that water purveyors are willing to pay to access this storage when needed during drought or emergency (NJWSA 2011). Irrigation withdrawals, mainly for corn, soybean, and vegetable crops, are rising due to warmer temperatures during the summer. The value of irrigation is $12 million/yr. based on DRBC allocations of 36.5 mgd (Barr and Sayers 2013), given the median value of withdrawals is $300/ac-ft. ($0.92/1000 gallons) in 2010, adjusted from Frederick et al. (1996). The economic value of water to irrigate 141,138 ac of cropland (USDA 2009) in the Delaware basin is $32 million, based on irrigation water needs of nine inches from June through September at a unit value of $300/ac-ft. (Frederick et al. 1996). The value of thermoelectric power plant water withdrawals (Barr and Sayers 2013) generating 13,458 megawatts of electricity in the Delaware basin is $297 million/yr., based on a unit value of $0.14/1000 gallons developed by Frederick et al. (1996). The annual value of industrial water supply withdrawals (804 mgd) in the Delaware basin (Barr and Sayers 2013) is $179 million based on a unit value of $200/ac-ft. (Frederick et al. 1996), adjusted for 2010. The annual value of hydropower based on DRBC allocations of 539 mgd is $20 million based on the median value of $0.10/1000 gallons in 2010, adjusted from Frederick et al. (1996).

Fish/Wildlife. The annual economic value of fish and wildlife in the Delaware basin is $1.5 billion. The annual value of commercial fish landings was $25.4 million in 2000 or $34.1 million in 2010 according to the National Marine Fisheries Service (NOEP 2009), including blue crab ($14.4 million), summer flounder ($5.3 million), Atlantic menhaden ($4.3 million), eastern oyster ($3.7 million), striped bass ($2.3 million), and American eel ($0.8 million) in the Delaware Estuary. Scaled by ratio of basin area to state area, the annual value of fishing ($576 million), hunting ($340 million), and wildlife viewing recreation ($561 million) trip and gear expenditures is $1.5 billion in the Delaware basin or $134 million in Delaware, $574 million in New Jersey, $160 million in New York, and $608 million in Pennsylvania (US DoI, FWS 2006). Based on 63,000 angler days, the annual value based on willingness to pay $102/trip/angler for the Delaware River shad fishery is $6.5 million in 2010 (The Pennsylvania Fish and Boat Commission 2011). Along the east branch, west branch, and upper Delaware River in the Catskills of New York, wild trout fishing contributed over $29 million in economic activity and nearly 350 jobs with $3.6 million in wages (Maharaj, McGurrin, and Carpenter 1998).

Agriculture. The Delaware basin covers 12,769 mi^2 or just 13% of the combined land area of Delaware, New Jersey, New York, and Pennsylvania, yet accounts for $3.4 billion or 27% of farm products sold in the four states from 3,010 mi^2 of farmland (USDA 2009).

Forests. The U.S. Forest Service and Delaware Center for Horticulture (Nowak et al. 2008) estimated 7,137 acres of forests in New Castle County, Delaware have carbon storage ($79/ac), carbon sequestration ($29/ac), air pollution removal ($266/ac), building energy savings ($56/yr.), and avoided carbon emissions ($3/ac) benefits of $5.9 million ($827/ac). Applying these per acre multipliers, 4,343,190 forested acres in the Delaware basin provide economic benefits that total $5.1 billion/yr. These economic benefits in carbon, air pollution, and energy savings are quite significant as half of the Delaware basin is covered by forests.

Parks. Public parks provide $1.8 billion in annual economic value to the Delaware River basin. The Trust for Public Land (2009) found the 444-acre City of Wilmington park system provides annual economic savings to the public due to health benefits from exercise in the parks ($9,734/ac), community cohesion benefit from socializing in the parks ($2,383/ac), water pollution benefits from treating stormwater ($921/ac), and air pollution mitigation by tree and shrub absorption ($88/ac). Transferring the unit values from the City of Wilmington study, public parks (169 mi^2) within the Delaware basin provide health ($1.3 billion), community cohesion ($314 million), water pollution ($121 million), and air pollution

Table 1. Annual economic activity in the Delaware River basin in 2010.

Activity	2010 ($ mil)	Source
Recreation (Boating, Fishing, Swimming)		
Clean Water Act Restoration		
• Viewing/Aesthetics ($0.58/person)	5	Parsons et al. (2003), University of Delaware
• Boating ($0.76/person)	6	Parsons et al. (2003), University of Delaware
• Fishing ($2.95/person)	24	Parsons et al. (2003), University of Delaware
• Swimming ($6.88/person)	57	Parsons et al. (2003), University of Delaware
Water Quality Based Recreation		
• Swimming ($13.40/trip)	9	Johnston et al. (2002), University of Rhode Island
• Boating ($30/trip)	47	Johnston et al. (2002), University of Rhode Island
• Fishing ($62.79/trip)	52	Johnston et al. (2002), University of Rhode Island
• Wildlife/Bird-watching ($77.73/trip)	104	Johnston et al. (2002), University of Rhode Island
Skiing (1.9 million ski-days @$45/day)	88-325	Pennsylvania Ski Areas Association (2009)
Paddling-based Recreation (621,000 paddlers)	362	Outdoor Industry Association (2006)
Del. Water Gap River Recreation (267,000 visitors)	41	Cordell et al. (1990), National Park Service
Canoe/Kayak/Rafting (225,000 visits)	9	Canoe and Kayak Liveries (2011)
Powerboating (232,000 boat registrations)	395	National Marine Manufacturers Association (2010)
Water Quality		
Willing to Pay for Clean Water ($38-$121/user)	659	Bockstael et al. (1989), University of Maryland
Water Treatment by Forests ($96/mgd)	63	Trust for Public Land and AWWA (2004)
Wastewater Treatment ($4.00/1000 gallons)	1,722	Yagecic et al. (2014)
Increased Property Value (+8%)	13	USEPA (1973)
Water Supply		
Drinking Water Supply ($4.78/1000 gallons)	3,145	Corrozi and Seymour (2008); Barr and Sayers (2013)
Reservoir Storage ($0.394/1000 gallons)	145	NJWSA (2011)
Irrigation Water Supply ($300/ac-ft)	32	Frederick et al. (1996); USDA (2009)
Thermoelectric Power Water Supply ($44/ac-ft)	297	USEIA (2002); USNETL (2009)
Industrial Water Supply ($200/ac-ft)	179	Frederick et al. (1996); Barr and Sayers (2013)
Hydropower Water Supply ($32/ac-ft)	20	Frederick et al. (1996); Barr and Sayers (2013)

Table 1 (continued). Annual economic activity in the Delaware River basin in 2010.

Activity	2010 ($ mil)	Source
Fish/Wildlife		
Commercial Fish Landings ($0.60/lb)	34	National Ocean Economics Program (2009)
Fishing (11-18 trips/angler, $17-$53/trip)	576	U.S. Fish and Wildlife Service (2008)
Hunting (16 trips/hunter, $16-50/trip)	340	U.S. Fish and Wildlife Service (2008)
Wildlife/Bird-watching (8-13 trip/yr, $15-$27/trip)	561	U.S. Fish and Wildlife Service (2008)
Shad Fishing (63,000 trips, $102/trip)	6	Pennsylvania Fish and Boat Commission (2011)
Wild Trout Fishing	29	Maharaj et al. (1998)
Agriculture		
Crop, Poultry, Livestock Value ($1,180/ac)	3,371	USDA Census of Agriculture 2007 (2009)
Forests		
Carbon Storage ($827/ac)	3,592	Nowak et al. (2008)
Carbon Sequestration ($29/ac)	126	Nowak et al. (2008)
Air Pollution Removal ($266/ac)	1,155	Nowak et al. (2008)
Building Energy Savings ($56/ac)	243	Nowak et al. (2008)
Avoided Carbon Emissions ($3/ac)	13	Nowak et al. (2008)
Public Parks		
Health Benefits ($9,734/ac)	1,283	Trust for Public Land (2009)
Community Cohesion ($2,383/ac)	314	Trust for Public Land (2009)
Stormwater Benefit ($921/ac)	121	Trust for Public Land (2009)
Air Pollution ($88/ac)	12	Trust for Public Land (2009)
Del. Water Gap Natl. Rec. Area (5.6 million visits)	160	Stynes (2011)
Marcellus Shale Gas		
Natural Gas (4.0 trillion ft^3@ $2.68/1000 ft^3)	425	USEIA (2010); Coleman et al. (2011)
Navigation		
Port Activity	2,400	Economy League of Greater Philadelphia (2008)
Delaware River Basin	**≥$22 billion**	

mitigation ($12 million) benefits. Stynes (2011) from Michigan State University estimated the National Park system at the Delaware Water Gap National Recreation Area and Upper Delaware National Wild and Scenic River drew 5,592,229 recreation visits in 2010 with total visitor spending of $160 million, generating 2,198 jobs and $61 million in wages.

Marcellus Shale Natural Gas. While a moratorium on natural gas drilling holds in place in the Delaware basin, it remains a critical water-energy nexus issue for the region and the value of the fossil fuel activity is discussed here. Approximately 9% of the 54,000 mi^2 Marcellus Shale Formation lies in the upper third of the Delaware basin. Buffers set by proposed DRBC and New York State (NYSDEC) rules, existing Pennsylvania Act 13, and drilling bans in four New York towns would exclude 2,363 mi^2 or nearly half of the 4,940 mi^2 Marcellus Shale region in the Delaware basin from natural gas drilling, leaving 2,577 mi^2 available for extraction (Kauffman and Homsey 2013). With buffers in place to shield sensitive water resources from hydraulic fracturing, the estimated economic value of 4.0 trillion cubic feet (tcf) of potentially recoverable Marcellus Shale gas in the basin at the 2012 wellhead price ($2.68/1000 cf) is $425 million/yr. or three times less than the value at the 2008 price (Table 2).

Navigation. The Delaware River docked nearly 2,900 ships in 2006 and is the largest freshwater port in the world with $2.4 billion in total economic output. The five Delaware River ports had combined imports of $41 billion, the 5th largest port in the U.S. after Los Angeles, Newark (NJ), Houston, and Long Beach (CA) and ahead of Seattle, Norfolk (VA), and Baltimore. The Delaware River port, from Wilmington to Philadelphia to Trenton, imports 1/2 of the nation's cocoa beans, 1/3 of the bananas, 1/4 of all fruit and nuts, and 51% of container trade value nationwide (Economy League of Greater Philadelphia 2008).

Ecosystem Services

Ecosystem services provided by natural habitat include air filtration, water filtration, recycling nutrients, soil conservation, pollinating crops and plants, climate regulation, carbon sequestration, flood/stormwater control, and hydrologic cycle regulation. Ecological resources provide marketable goods and services such as timber, fish and wildlife recreation, hiking, and boating/kayaking. Natural capital is the sum of goods (commodities like water, crops, and timber that can be sold) and services (functions like flood control, water filtration, and wildlife/fisheries habitat) provided by watershed ecosystems such as wetlands, forests, farms, and open water. In addition to these direct benefits, ecosystems also provide indirect benefits such as ecotourism by hunters, fishermen, boaters, and hikers who spend money to visit natural sites and realize the value of improved water quality and habitat.

Other studies have defined ecosystem services in watersheds in and around the Delaware basin. A Cecil County, Maryland study by the Conservation Fund found that riparian forest wetlands provided significant stormwater/flood control ($32,000/ac), water supply ($8,630/ac), and clean water ($1,925/ac) functions (Weber 2007). The NJDEP and University of Vermont estimated the value of New Jersey's natural capital to be $20 billion/yr. (+/- $9 billion/yr.) in 2004 with a net present value of $681 billion based on a discount rate of 3% calculated over 100 years in the future (Mates 2007). The Wilderness Society (Krieger 2001) concluded that temperate forest ecosystem services from climate regulation, water supply, water quality, and recreation totaled $392/ac in 1994. A contingent value study by University of Rhode Island economists found that natural resources values in the Peconic Estuary watershed on Long Island in New York ranged from $6,560/ac for wetlands to $9,979/ac for farmland in 1995 (Johnston et al. 2002). Ingraham and Foster (2008) from the University of Maryland determined ecosystem values of forests and freshwater wetlands in the National Wildlife Refuge System were $845/ac and $6,268/ac, respectively. The Audubon Society found the economic value of ecosystems in Massachusetts ranged from $984/ac for forests to $15,452/ac for saltwater wetlands (Breunig 2003). The ecosystem services value of agriculture captures the value of the soil and soil moisture. The market value of agricultural crops, poultry, and livestock sold from 1,926,524 acres of farmland in the Delaware River basin was $3.37

Table 2. Potential value of Marcellus Shale gas in the Delaware basin.

Year	Available Marcellus Shale Area (mi^2)	Available Natural Gas[1] (tcf)	Annual Wellhead Price[2] ($/1000 cf)	Natural Gas Value ($ mil)	Annual Natural Gas Value[3] ($ mil/yr)
without buffers					
2008	4,940	7.6	$7.97	60,572	2,423
2009	4,940	7.6	$3.67	27,892	1,116
2010	4,940	7.6	$4.48	34,048	1,362
2011	4,940	7.6	$3.95	30,020	1,201
2012	4,940	7.6	$2.68	20,368	815
with buffers[4]					
2008	2,363	4.0	$7.97	31,880	1,275
2009	2,363	4.0	$3.67	14,680	587
2010	2,363	4.0	$4.48	17,920	717
2011	2,363	4.0	$3.95	15,800	632
2012	2,363	4.0	$2.68	10,625	425

[1]Source: Coleman et al. 2011. [2]Source: USEIA 2010. [3]Assumes 25-year recovery period. [4]w/ DRBC, NY, and PA buffers.

billion ($1,676/ac) or $2,502/ac in Delaware, $1,192/ac in New Jersey, $562/ac in New York, and $2,070/ac in Pennsylvania (USDA 2009).

The estimated value of natural goods and services provided by ecosystems in the Delaware River basin (12,742 mi^2) is $21 billion in 2010 with a net present value (NPV) of $683 billion (based on an annual interest rate of 3%) which includes $2.5 billion for Delaware, $6.6 billion for New Jersey, $3.5 billion for New York, and $8.6 billion for Pennsylvania (Table 3). If the lowest or highest per acre estimates of ecosystem services values from other studies were employed for value transfer in place of the NJDEP values, the value of natural resources in the Delaware basin would range from $9.6 billion to $94.7 billion. These estimates do not include the ecosystem services value of $61 billion (1,946/ac) of open water habitat (720 mi^2) in the tidal Delaware River and Bay between the shores of Delaware, Pennsylvania, and New Jersey.

Ecosystems within the Delaware basin are comprised of forests (53%), farmland (24%), freshwater wetlands (5%), saltwater wetlands (2%), and open water/marine (1%) (Figure 4). Over 15% of the Delaware basin is urban (Figure 5). Forests ($8.6 billion or $1,978/ac) provide the largest ecosystem services values in the Delaware basin followed by freshwater wetlands ($5.8 billion or $13,261/ac), farmland ($4.8 billion or $2,503/ac), and saltwater wetlands ($1.1 billion or $7,235/ac).

Jobs and Wages

The Delaware basin supports 600,000 direct and indirect jobs with $10 billion in annual wages in the coastal, farm, ecotourism, recreation, water/wastewater, and port industries (Table 4). It is noted that use of indirect jobs benefits is debated in many economic circles. Indirect jobs benefits calculated by multipliers are summarized here to roughly estimate the spinoff effect to the economy from jobs directly related to the waters of the Delaware basin.

The Delaware basin is home to 3,480,483 jobs with $172.6 billion in wages in Delaware (316,014 jobs, $16.5 billion wages), New Jersey (823,294

jobs, $38.1 billion wages), New York (69,858 jobs, $2.5 billion wages), and Pennsylvania (2,271,317 jobs, $115.5 billion wages).

According to the National Ocean Economic program (NOEP 2009), coastal counties within the Delaware basin boundary contribute 44,658 coastal jobs with $947 million in annual wages with contributions of $1.8 billion toward the GDP. Table 5 summarizes employment, wages, and employment within the Delaware basin by multiplying the 2009 NOEP report county-wide values by the ratios of coastal county area within the basin by total coastal county area within the state which are 80% for Delaware, 5% for New Jersey, and 86% for Pennsylvania.

Jobs directly associated with the Delaware basin (Table 6), such as water/sewer construction, water utilities, fishing, recreation, tourism, and ports, employed a total of 240,621 people with $4.9 billion in wages. The breakdown per state follows: Delaware (15,737 jobs, $340 million wages), New Jersey (62,349 jobs, $1.3 billion wages), New York (32,171 jobs, $550 million wages), and Pennsylvania (130,364 jobs, $2.8 billion wages). Jobs indirectly related to the Delaware basin (based on multipliers of 1.2 for jobs and 0.8 for salaries) employed 288,745 people with $4.0 billion in wages between the states of Delaware (18,884 jobs, $270 million wages), New Jersey (74,819 jobs, $1.0 billion wages), New York (38,605 jobs, $400 million wages), and Pennsylvania (156,437 jobs, $2.2 billion wages).

In 2007 there were 30,455 farms in Delaware basin counties or 21,840 farms within the basin boundary. The USDA (2009) estimates each farm employs 2.1 full time workers. Farming provides 45,865 jobs with $1.9 billion in wages in the Delaware basin. The New Jersey Department of Environmental Protection (Mates 2007) estimated that the average annual salary per ecotourism job in 2007 was $32,843 using data from the U.S. Department of Interior Fish and Wildlife Service and U.S. Census Bureau (2008) report on fishing, hunting, and wildlife associated recreation. If fishing, hunting, and bird/wildlife associated recreation in the Delaware River basin accounts for $1.5 billion in annual economic activity ($2006), then ecotourism provides for 44,941 jobs.

Over 300 public and private water utilities withdraw up to 1,800 mgd of drinking water from the Delaware basin (Barr and Sayers 2013) and employ at least 8,750 jobs with annual wages of $485 million. More than 60 wastewater utilities discharge almost 1.2 billion gallons per day of treated wastewater to the Delaware basin (Yagecic et al. 2014) and employ 1,298 employees who earn $61 million in annual wages. Over 100 nonprofit watershed and environmental organizations (DRBC 2013) employ at least 200 professionals who earn at least $9.5 million in wages on programs to restore the watersheds in the Delaware basin.

Recreational pursuits that rely on clean and plentiful water resources in the Delaware basin provide thousands of jobs in the regional economy. In the Pocono Mountains of Pennsylvania, nine ski resorts provide 1,753 direct jobs in the Delaware basin generating aggregate annual revenues of $87,655,063 from 1,908,228 skier visits. In the Mid-Atlantic census division (NY, NJ, PA), the Outdoor Industry Association (2006) estimated that paddling-based recreation is practiced by 11% of the population and is responsible for 22,844 jobs or 4,226 jobs when scaled by population to the Delaware basin. Stynes (2011) from Michigan State University estimated that the Delaware Water Gap National Recreation Area and Upper Delaware Wild and Scenic River drew 5,592,229 recreation visits in 2010 with total visitor spending of $160 million that generated 2,198 jobs and $61 million in wages. The 37 canoe and kayak liveries along the Delaware, Lehigh, Schuylkill, and Brandywine Rivers employ 225 people to lease watercraft to approximately 225,000 visitors with earnings of $9 million per year, assuming a daily rental fee of $40 per person. Along the Beaverkill, east branch, west branch and upper Delaware River in New York, wild trout fishing provides for 350 jobs with $3.6 million in wages (Maharaj, McGurrin, and Carpenter 1998)

The Economy League of Greater Philadelphia (2008) reported that Delaware River ports employ 4,056 workers earning $326 million in wages. Each port activity job indirectly supports an additional two jobs and employee spending for a total of 12,121 port jobs with $772 million in wages and a $2.4 billion annual economic output.

Table 3. Ecosystem services values in the Delaware River basin.

Ecosystem	Area (ac)	Unit Value ($/ac/yr 2010)	Present Value (2010 $)	Net Present Value ($)
Low Range				
Freshwater Wetlands	422,838	6,268[5]	2,650,346,040	86,136,246,300
Marine	16,588	8,670[2]	143,820,496	4,674,166,116
Farmland	1,926,524	1,387[6]	2,672,088,886	86,842,888,779
Forest Land	4,343,190	641[3]	2,783,984,500	90,479,496,255
Saltwater Wetland	145,765	6,269[2]	913,802,685	29,698,587,269
Barren Land	18,630	0	0	0
Urban	1,206,504	296[2]	357,125,287	11,606,571,818
Beach/Dune	900	42,149[2]	37,915,873	1,232,265,862
Open Water	92,615	217[5]	20,097,408	653,165,771
Total	8,173,554		9,579,181,174	311,323,388,171
Mid-Point Estimate				
Freshwater Wetlands	422,838	13,621[2]	5,759,329,048	187,178,194,067
Marine	16,588	10,006[2]	165,982,947	5,394,445,767
Farmland	1,926,524	2,503[2]	4,823,030,404	156,748,488,136
Forest Land	4,343,190	1,978[2]	8,591,367,360	279,219,439,184
Saltwater Wetland	145,765	7,235[2]	1,054,617,851	34,275,080,170
Urban	1,206,504	342[2]	412,157,579	13,395,121,322
Beach/Dune	900	48,644[2]	43,758,633	1,422,155,566
Open Water	92,615	1,946[2]	180,210,703	5,856,847,857
Total	8,154,924		$21,030,454,525	$683,489,772,069
High Range				
Freshwater Wetlands	422,838	43,685[1]	18,471,660,300	600,328,959,736
Marine	16,588	8,670[2]	143,820,496	4,674,166,116
Farmland	1,926,524	9,979[4]	19,224,783,698	624,805,470,173
Forest Land	4,343,190	12,033[1]	52,261,599,829	1,698,501,994,444
Saltwater Wetland	145,765	28,146[1]	4,102,710,221	133,338,082,193
Barren Land	18,630	0	0	0
Urban	1,206,504	296[2]	357,125,287	11,606,571,818
Beach/Dune	900	42,149[2]	37,915,873	1,232,265,862
Open Water	92,615	1,686[2]	156,148,527	5,074,827,144
Total	**8,173,554**		**94,755,764,230**	**3,079,562,337,486**

[1]Weber 2007. [2]Mates 2007. [3]Krieger 2001. [4]Johnston et al. 2002. [5]Ingraham and Foster 2008. [6]Breunig 2003.
[7]Value of goods only as measured by agricultural crops, livestock, and poultry sold from USDA 2009.

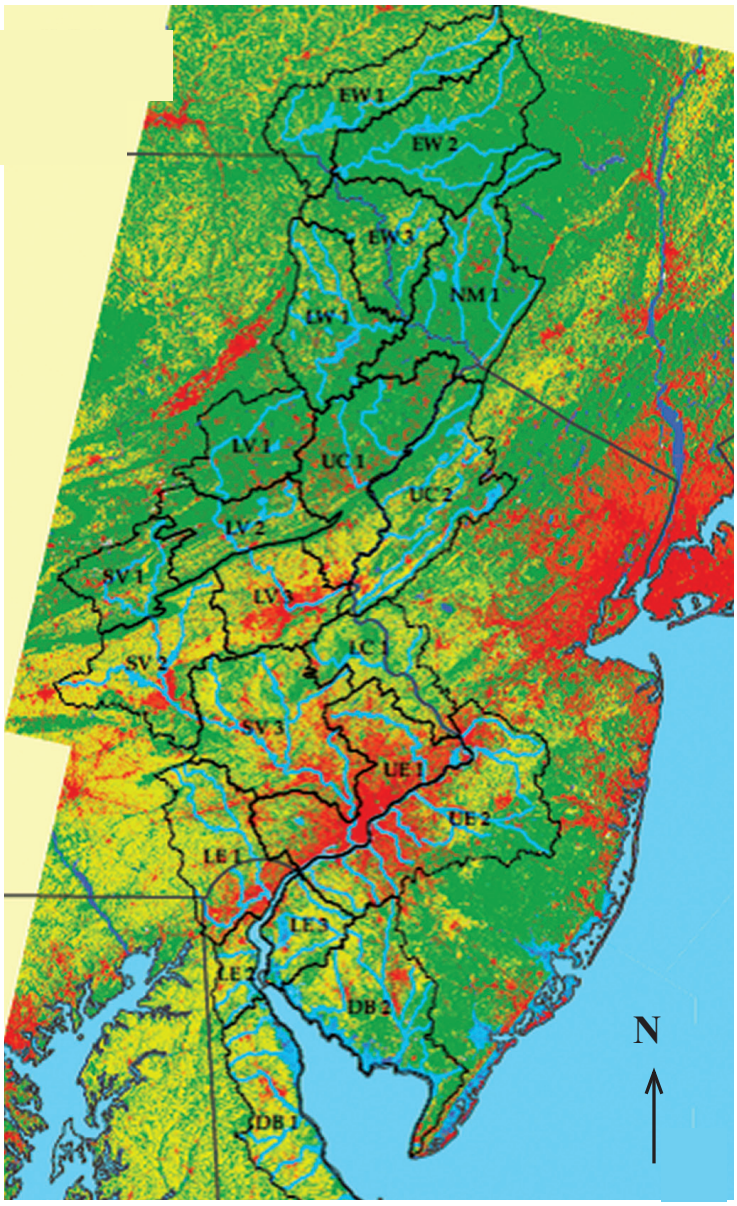


Figure 4. Land use in the Delaware Basin, 2006 (U.S. NOAA CSC 2006). Denoted by Urban/Suburban (Red), Forests/Open Space (Green), Agriculture (Yellow), and Wetlands/Water (Blue). Subwatersheds outlined in black.

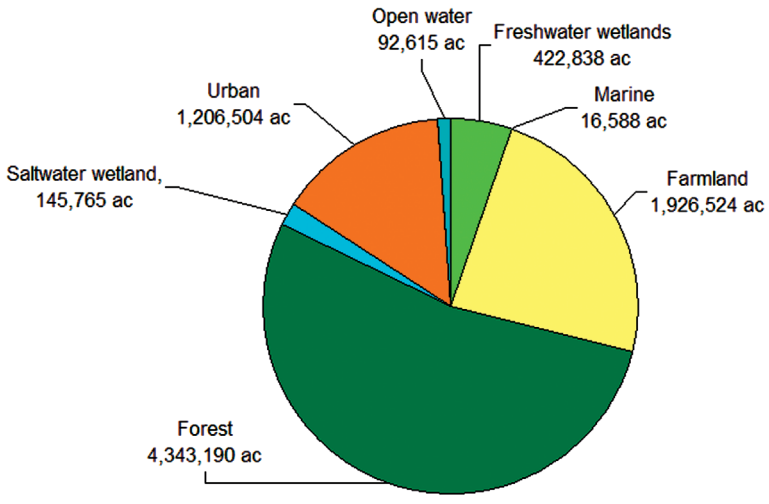


Figure 5. Ecosystem service areas within the Delaware River basin for forest, saltwater wetland, freshwater wetland, open water, marine, farmland, and urban habitat.

Table 4. Direct and indirect jobs and wages related to the Delaware River basin.

Sector	Jobs	Wages ($ million)	Data Source
Direct Basin Related	240,621	4,900	U.S. DOL Bureau of Labor Statistics (2010)
Indirect Basin Related	288,745	4,000	U.S. DOL Bureau of Labor Statistics (2010)
Coastal	44,658	947	National Ocean Economics Program (2009)
Farm	45,865	1,880	USDA Census of Agriculture (2009)
Fishing/Hunting/Birding	44,941	1,476	U.S. Fish and Wildlife Service (2008)
Water Supply Utilities	8,750	485	Corrozi and Seymour (2008); Barr and Sayers (2013)
Wastewater Utilities	1,298	61	Yagecic et al. (2014)
Watershed Organizations	201	10	DRBC (2013)
Ski Area Jobs	1,753	88	Pennsylvania Ski Areas Association (2009)
Paddling-based Recreation	4,226	148	Outdoor Industry Association (2006)
Canoe/Kayak/Rafting	225	8	Canoe and Kayak Liveries (2011)
Wild Trout Fishing	350	4	Maharaj, McGurrin, and Carpenter (1998)
Delaware Water Gap National Recreation Area	2,198	61	Stynes (2011)
Port Jobs	12,121	772	Economy League of Greater Philadelphia (2008)
Delaware Basin Total	**> 600,000**	**>$10 billion**	

Table 5. Coastal employment, wages, and GDP in the Delaware River basin (National Ocean Economics Program 2009).

Sector	Employment	Wages ($ million)	GDP ($ million)
Delaware Basin	44,658	947	1,831
Marine Construction	*ND	ND	12
Living Resources	354	8	195
Offshore Minerals	ND	ND	14
Tourism and Recreation	33,430	151	947
Marine Transportation	1,744	53	560
Ship and Boat Building	ND	ND	72

*ND = No data reported.

Table 6. Direct and indirect basin-related jobs in the Delaware River basin, 2009 (U.S. Department of Labor, Bureau of Labor Statistics 2010).

Sector	Industry	1997 NAICS Codes	Direct Jobs[1]	Direct Wages (x $1,000)	Indirect Jobs[2]	Indirect Wages (x $1,000)
Construction	Marine Related	237120	1,004	63,531	1,205	50,825
	Water and Sewer	23711	6,703	379,443	8,044	303,554
	Construction	237990	750	41,652	900	33,322
Living Resources	Fish Hatcheries	112511	0	0	0	0
	Aquaculture	112512	0	0	0	0
	Fishing/Foresty	11411	138	4,937	166	3,950
	Finfish Fishing	114111	111	5,591	133	4,473
	Shellfish Fishing	114112	28	995	34	796
	Seafood Markets	445220	403	9,345	484	7,476
	Seafood Process.	31171	97	6,734	116	5,387
	Comm. Fisheries		0	0	0	0
Ship/Boat Building	Boat Building Repair	336612	0	0	0	0
Transportation	Deep Sea Freight	483111	0	0	0	0
	Marine Transport.	483112	3,681	146,755	4,417	117,404
	Search/Navigation	334511	755	64,226	906	51,381
	Warehousing	493110	11,186	446,118	13,423	356,894
	Refrigerated Warehousing	493120	698	28,691	838	22,953
	Ports		0	0	0	0
	Dredging/Disposal		0	0	0	0
Education/Research	Environ. Organizations	813312	929	30,039	1,115	24,032
	Environ. Consulting	54162	2,972	80,447	3,566	64,357
Water/Wastewater	Water/Sewage Systems	2213	1,172	30,048	1,406	24,038
	Waste Management	562	5,328	253,609	6,394	202,887
	Septic Tank Services	562991	596	25,343	715	20,275

[1]Direct jobs are directly related to the Delaware basin. [2]Indirect jobs/salaries are derived from purchases of goods and services calculated by multipliers of 1.2 for jobs and 0.8 for wages.

Table 6 (continued). Direct and indirect basin-related jobs in the Delaware River basin, 2009 (U.S. Department of Labor, Bureau of Labor Statistics 2010).

Sector	Industry	1997 NAICS Codes	Direct Jobs[1]	Direct Wages (x $1,000)	Indirect Jobs[2]	Indirect Wages (x $1,000)
Minerals	Sand & Gravel	212321	166	8,109	199	6,487
	Industrial Sand and Gravel Mining	212322	81	3,865	97	3,092
	Oil & Gas	541360	55	4,554	66	3,643
Tourism/Recreation	Recreation	487990	52	1,184	62	947
	Sports and Recreation Instruction	611620	1,044	18,084	1,253	14,467
	Recreation Good Rental	532292	50	774	60	619
	Amusement	713990	16,221	233,566	19,465	186,853
	Misc. Recreation		1,100	16,574	1,320	13,259
	Boat Dealers	441222	355	13,434	426	10,747
	Restaurants	722110	106,472	1,828,487	127,766	1,462,790
	Restaurants	722211	53,260	616,854	63,912	493,483
	Cafeteria	722212	1,715	26,874	2,058	21,499
	Snack Bar	722213	10,958	165,699	13,150	132,559
	Hotels & Lodging	721110	9,938	307,236	11,926	245,789
	Bed & Breakfast InN	721191	92	1,583	110	1,266
	Marinas	713930	202	6,410	242	5,128
	RV Park/Camps	721211	483	15,999	580	12,799
	Scenic Tours	487210	55	1,141	66	913
	Sporting Goods	339920	1,192	19,039	1,430	15,231
	Zoos, Aquaria	712130	55	1,959	66	1,567
	Nature Park	712190	524	31,870	629	25,496
Total			**240,621**	**4,940,799**	**288,745**	**3,952,639**

[1]Direct jobs are directly related to the Delaware basin. [2]Indirect jobs/salaries are derived from purchases of goods and services calculated by multipliers of 1.2 for jobs and 0.8 for wages.

Summary and Conclusions

The Delaware River basin in Delaware, New Jersey, New York, and Pennsylvania is a valuable ecological and economic resource. The 13,000 square mile watershed supplies drinking water to 5% of the United States including the nation's 1st (New York City) and 7th (Philadelphia) largest metropolitan economies. Furthermore, the Delaware River supports the largest freshwater port in the world while sustaining a recovering anadromous shad and striped bass fishery (Kauffman 2010).

The annual economic value of the Delaware River basin is at least $22 billion based on estimates of economic activity, ecosystem services, and jobs and wages related to the waters of the watershed. The Delaware basin contributes over $22 billion in annual economic activity from potential Marcellus Shale gas extraction ($425 million), recreation ($1.2 billion), fish/wildlife ($1.5 billion), public parks ($1.8 billion), water quality ($2.5 billion), navigation ($2.6 billion), agriculture ($3.4 billion), water supply ($3.8 billion), and forest ($5.1 billion) benefits. The value of natural goods and services from ecosystems in the Delaware basin is $21 billion ($2010) including provisions by Delaware ($2.5 billion), New Jersey ($6.6 billion), New York ($3.5 billion), and Pennsylvania ($8.6 billion). The Delaware River basin is a jobs engine that supports 600,000 direct/indirect jobs with $10 billion in annual wages in the coastal, farm, ecotourism, water/wastewater, ports, and recreation industries.

It is also important to note that the estimates presented in this report are not meant to be used to compare/contrast different uses of the basin for their value. Some values were not included in these estimates because the data to assess them is not readily available, or does not exist. Values for the activities and resources vary in how they were applied by value transfer techniques from nearby watersheds to the Delaware basin, making it difficult to accurately compare values across uses and activities. Gathering more primary economic data from research in the Delaware River basin (such as local willingness to pay for clean water surveys) would improve comparability of information across uses, as well as make value estimates more comprehensive by including confidence intervals with range and error bar estimates.

This economic analysis documents the vast worth of the Delaware River as an economic, ecological, and civic resource. The economic data presented in this analysis may be useful in translating the value of the Delaware River basin to the public. Elected officials, public decision makers, and public, private, nonprofit organizations may use the economic data to set priorities for investment in the Delaware River basin. States, counties, and municipalities may employ this monetary data by linking the economy of the watershed to state, regional, and local economic development and workforce development programs. This research demonstrates the Delaware River basin provides significant economic benefits to the region and is worthy of priority investments by elected officials and decision-makers to protect and restore these natural resources.

Acknowledgements

We are grateful for support from the Delaware River Basin Commission and Partnership for the Delaware Estuary while conducting this economic valuation research.

Author Bio and Contact Information

Gerald Kauffman is Director of the University of Delaware Water Resources Center, one of the 54 National Institutes for Water Resources (NIWR) supported by the U.S. Department of the Interior and U.S. Geological Survey at land grant universities in the 50 states, District of Columbia, and island territories of Guam, Puerto Rico, and U.S. Virgin Islands. He may be contacted at the University of Delaware - Water Resources Center, School of Public Policy and Administration, DGS Annex, 261 Academy Street, Newark, Delaware 19716 or jerryk@udel.edu.

References

Austin, J.C., S. Anderson, P.N. Courant, and R.E. Litan. 2007. *Healthy Waters, Strong Economy: The Benefits of Restoring the Great Lakes Ecosystem.* The Brookings Institution.

Barr, K. and D. Sayers. 2013. *Water Use Trends in the Delaware River Basin.* 2013 Mid-Atlantic Conference of the American Water Resources Association. Delaware River Basin Commission. West Trenton, New Jersey.

Bockstael, N.E., K.E. McConnell, and I.E. Stroud. 1989. Measuring the benefits of improvements in water

quality: The Chesapeake Bay. *Marine Resource Economics* 6: 1-18.

Breunig, K. 2003. *Losing Ground: At What Cost? Changes in Land Use and their Impact on Habitat, Biodiversity, and Ecosystem Services in Massachusetts*. Mass Audubon.

Canoe and Kayak Liveries in the Delaware River Basin. 2011. Personal Communication.

Chesapeake Bay Watershed Blue Ribbon Finance Panel. 2003. *Saving a National Treasure: Financing the Cleanup of the Chesapeake Bay.* A Report to the Chesapeake Executive Council.

Cech, T.V. 2005. *Principles of Water Resources History, Development, Management and Policy.* John Wiley and Sons, Inc., New York.

Coleman, J.L., R.C. Milici, T.A. Cook, R.R. Charpentier, M. Kirshbaum, T.R. Klett, R.M. Pollastro, and C.J. Schenk. 2011. *Assessment of Undiscovered Oil and Gas Resources of the Devonian Marcellus Shale of the Appalachian basin Province.* USGS Fact Sheet 2011–3092.

Cordell, H.K., J.C. Bergstrom, G.A. Ashley, and J. Karish. 1990. Economic effects of river recreation on local economies. *Water Resources Bulletin American Water Resources Association* 26(1): 53-60.

Corrozi, M. and M. Seymour. 2008. *Water Rates in Delaware and Surrounding States.* University of Delaware Institute for Public Administration-Water Resources Agency.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. Oneill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Costanza, R., R. de Groot, P. Sutton, S. van der Ploeg, S.J. Anderson, I. Kubiszewski, S. Farber, and R.K. Turner. 2014. Changes in the global value of ecosystem services. *Global Environmental Change* 26: 152-158.

Delaware River Basin Commission. 2013. *Local Watershed Associations in the Delaware River Basin.* Available at www.nj.gov/drbc/basin/watershed/. Accessed July 13, 2016.

Economy League of Greater Philadelphia. 2008. *Maritime Commerce in Greater Philadelphia: Assessing Industry Trends and Growth Opportunities for Delaware River Ports.* Available at http://economyleague.org/uploads/files/243954419196508911-maritime-commerce-in-greater-philadelphia.pdf. Accessed July 13, 2016.

Frederick, K.D., T. VandenBerg, and J. Hansen. 1996. *Economic Value of Freshwater in the United States.* Discussion Paper 97-03. Resources for the Future, Washington, D.C.

Goulder, L.H. and D. Kennedy, 1997. Valuing ecosystem services: Philosophical bases and empirical methods. In: *Nature's Services: Societal Dependence on Natural Ecosystems*, G. Daily (Ed.). Island Press, Washington, D.C.

Greeley-Polhemus Group. 1993. *Final Report: Assessment of Selected Delaware Estuary Economic and Resource Values.* Delaware Estuary Program Science & Tech. Advisory Committee.

Ingraham, M. and S.G. Foster. 2008. The value of ecosystem services provided by the U.S. national wildlife refuge system in the contiguous U.S. *Ecological Economics* 67: 608-818.

Johnston, R.J., T.A. Grigalunas, J.J. Opaluch, M. Mazzotta, and J. Diamantedes. 2002. Valuing estuarine resource services using economic and ecological models: The Peconic Estuary System Study. *Coastal Management* 30: 47-65.

Kauffman, G.J., 2010. The Delaware River revival: Four centuries of historic water quality change from Henry Hudson to Benjamin Franklin to JFK. *Pennsylvania History, A Journal of Mid-Atlantic Studies* 77(4): 432-465.

Kauffman, G.J. and C. Cruz-Ortiz. 2012. *Economic Value of the Barnegat Bay Watershed in New Jersey.* Prepared for the Barnegat Bay Partnership.

Kauffman, G.J. and A.R. Homsey. 2013. Economic value of Marcellus Shale gas in the Delaware basin. *Journal of Oil, Gas, and Mining* 1(1): 1-37.

Krieger, D.J., 2001. *Economic Value of Forest Ecosystem Services: A Review.* The Wilderness Society.

Latham, W.R. and J.E. Stapleford. 1987. *Economic Impacts of the Delaware Estuary.* Delaware Sea Grant College Program. No. DEL-SG-02-87.

Maharaj, V., J. McGurrin, and J. Carpenter. 1998. *The Economic Impact of Trout Fishing on the Delaware River Tailwaters in New York.* American Sportfishing Association and Trout Unlimited.

Mates, W.J. and J.L. Reyes. 2006. *The Economic Valuation of New Jersey State Parks and Forests.* New Jersey Department of Environmental Protection.

Mates, W.J. 2007. *Valuing New Jersey's Natural Capital: An Assessment of the Economic Value of the State's Natural Resources.* New Jersey Department of Environmental Protection.

McCormick, B. 2010. *Measuring the Economic Benefits of America's Everglades Restoration.* The Everglades Foundation.

National Ocean Economics Program. 2009. *State of the U.S. Ocean and Coastal Economies, Coastal and Ocean Economic Summaries of the Coastal States.*

National Marine Manufacturers Association. 2010. *2010 Recreational Boating Statistical Abstract.* Chicago, Illinois.

Narvaez, M.C., G. Kauffman, A. Homsey, N. Minni, C. Cruz-Ortíz, E. McVey, C. Halley. 2012. *Economic Benefits and Jobs Provided by Delaware Watersheds*. University of Delaware Water Resources Agency for Delaware Department of Natural Resources and Environmental Control.

New Jersey Department of Environmental Protection. 2007. *Valuing New Jersey's Natural Capital: An Assessment of the Economic Value of the State's Natural Resources.*

New Jersey Water Supply Authority. 2011. *New Jersey Water Supply Authority Basis and Background Statement.* Available at http://www.njwsa.org/Raritan2012.pdf. Accessed July 13, 2016.

Nowak, D.J., R.E. Hoehn, J. Wang, A. Lee, V. Krishnamurthy, and G. Schwetz. 2008. *Urban Forest Assessment in Northern Delaware.* Delaware Center for Horticulture and U.S. Forest Service.

Outdoor Industry Association. 2006. *The Active Outdoor Recreation Economy.*

Parsons, G.R., E.C. Helm, and T. Bondelid. 2003. *Measuring the Economic Benefits of Water Quality Improvements to Recreational Users in Six Northeastern States: An Application of the Random Utility Maximization Model.* Prepared for the EPA Office of Policy Economics and Innovation.

Pennsylvania Fish and Boat Commission. 2011. *Economic Value of Fishing and Boating in Pennsylvania.*

Pennsylvania Ski Areas Association. 2009. Available at http://www.skipa.com. Accessed July 13, 2016.

Stynes, D.J. 2011. *Economic Benefits to Local Communities from National Park Visitation and Payroll, 2010*. Michigan State University. Natural Resource Report NPS/NRSS/EQD/NRR-2011/481.

Thurston, H.W., M.T. Heberling, and A. Schrecongost. 2009. *Environmental Economics for Watershed Restoration.* CRC Press.

Trust for Public Land and American Water Works Association. 2004. *Protecting the Source: Land Conservation and the Future of America's Drinking Water.*

Trust for Public Land. 2009. *How much Value Does the City of Wilmington Receive from Its Park and Recreation System?*

U.S. Census Bureau. 2010. Property value: 2008-2009. *American Community Survey Briefs.*

U.S. Department of Agriculture. 2009. *2007 Census of Agriculture.*

U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2008. *2006 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation*. Available at https://www.census.gov/prod/2008pubs/fhw06-nat.pdf. Accessed June 8, 2016.

U.S. Department of Labor, Bureau of Labor Statistics. 2010. *State and Metro Area Employment, Hours, and Earnings.* Available at www.bls.gov/sae/. Accessed July 8, 2016.

U.S. Department of Labor, Bureau of Labor Statistics. 2015. *Consumer Price Index.* Available at www.bls.gov/cpi/. Accessed June 28, 2015.

U.S. Energy Information Administration. 2002. *Inventory of Electric Utility Power Plants in the United States 2000.* U.S. Department of Energy, Washington, D.C.

U.S. Energy Information Administration. 2010. *Natural Gas Weekly Update, Residential Natural Gas Prices.* Available at http://205.254.135.24/oog/info/ngw/ngupdate.asp. Accessed July 13, 2016.

U.S. Environmental Protection Agency. 1973. *Benefit of Water Pollution Control on Property Values.* EPA-600/5-73-005.

U.S. Environmental Protection Agency. 2012. *The Importance of Water to the U.S. Economy Part 1: Background Report Public Review Draft.*

U.S. National Energy Technology Laboratory. 2009. *Impact of Drought on U.S. Steam Electric Power Plant Cooling Water Intakes and Related Water Resource Management Issues.* Washington, D.C.

U.S. National Oceanic and Atmospheric Administration. 2006. Land Use/Land Cover Data. Coastal Services Center (CSC).

Weber, T. 2007. *Ecosystem Services in Cecil County's Green Infrastructure.* The Conservation Fund, Annapolis, Maryland.

World Business Council for Sustainable Development. 2011. *Guide to Corporate Ecosystem Valuation: A Framework for Improving Corporate Decision-making.*

Yagecic, J.R., C. Pindar, E. Silldorff. 2014. *Point Discharge Effluent Nutrient Concentrations in the Delaware River Basin: A Preliminary Review.* Delaware River Basin Commission, West Trenton, New Jersey.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 120-131, August 2016

Water and Society: Interdisciplinary Education in Natural Resources

*Laszlo J. Kulcsar[1], Joseph A. Aistrup[2], Thomas Bulatewicz[1], Jeffrey M. Peterson[3], Stephen M. Welch[1], and David R. Steward[1]

[1]Kansas State University, Manhattan, KS, [2]Auburn University, Auburn, AL, [3]University of Minnesota, St. Paul, MN
**Corresponding Author*

Abstract: Natural resource management and education must account for both the natural and human components of a complex system, yet examples of such interdisciplinary approaches are still relatively rare, especially in education. This study discusses a graduate seminar on water management, developed from an interdisciplinary National Science Foundation (NSF) grant. The seminar applied constructivist pedagogy which is not only best suited for interdisciplinary work, but also allows flexibility to redesign the course components and deliverables after each evaluation cycle. While the seminar was a successful experiment, several challenges remain when implementing such courses. These challenges include not only the organization and assessment of course deliverables, but also fitting such courses into the administrative structure of the university when represented disciplines are located in several colleges across campus.

Keywords: *constructivism, interdisciplinary, groundwater, coupled systems, graduate education*

Water is a primary requirement for successful agricultural production to feed the world's population. Yet, water related stresses such as aquifer depletion and climate change pose threats to food security in both the short- and long-term. Today it is well understood that natural resource management must account for both the natural drivers and the human dimension of such complex systems. However, most academic knowledge has been, and continues to be, produced within isolated disciplines (Miller et al. 2008), and the training of both scholars and professionals still takes place within disciplinary environments. While the recognition that human actions are intertwined with biophysical changes is not new (Marsh 1864), the push for research on these relationships that incorporates knowledge across social and natural sciences has intensified in recent decades (Stern et al. 1992; Lubchenco 1998; Grimm et al. 2000; Kinzig 2001; Redman et al. 2004). This recent effort stems, at least in part, from policy makers' demands for science-based assessments of problems that are outgrowths of coupled human and natural systems. While it is important to learn specific skills and perspectives accumulated by various disciplines, it is equally important to gain a comprehensive understanding of coupled natural and human systems to find the best responses to complex challenges that emerge.

Funding agencies such as the National Science Foundation (NSF) have sought to accelerate research integrating the social and natural sciences by creating new streams of research funding, with 2011-2016 Strategic Goals to "Transform the Frontiers" and to "Innovate for Society." Within the last decade, the NSF funded research grants under new crosscutting programs such as Dynamics of Coupled Human-Natural Systems (CNH), Urban Long-Term Research Areas (ULTRA), Integrative Graduate Education and Research Traineeship (IGERT), NSF Research Traineeship (NRT), Water, Society, and Climate (WSC), and the National Ecological Observatory Network

(NEON). Developments in both the philosophy of scientific inquiry and the funding responses from national agencies point toward the integration of research and education within an interdisciplinary framework. While almost all challenges societies face are complex, spanning several disciplines and requiring multidisciplinary solutions, certain fields are better suited for such initiatives.

This study discusses experiences and lessons learned from an experimental graduate course, "Water and Society," an outgrowth of NSF funded interdisciplinary activities on water resources connecting research and education. Constructivist pedagogy was heavily emphasized and focused on the use of water resources in an agricultural setting. The importance of this approach lies in the central component of constructivist pedagogy, which redefines the role of the student to become an active learner, contributing to the delivery of the content. This approach works particularly well for interdisciplinary learning where disciplinary knowledge is not only presented and synthesized, but also actively negotiated throughout the entire course. The class was taught by a team of instructors who themselves had to negotiate this knowledge, producing a dynamic and balanced classroom environment in contrast to the typical strictly separated roles between teachers and learners.

The pedagogical and administrative experiences and challenges of this five year project are discussed in this paper. Since academic knowledge has been advancing toward interdisciplinary perspectives and applications, graduate courses like this one will likely be increasingly important components of higher education, working toward the societal goal of training professionals capable of responding to sustainability challenges in natural resource management.

Conceptual Background

Miller et al. (2008) argue that the main limitation to interdisciplinary research is not that scholars fail to recognize the inherent cross-disciplinary nature of researchable issues, but rather that their disciplinary training exposed them to particular epistemologies, or theories of knowledge. While essential to the progression of knowledge within disciplines, different epistemologies across disciplines are incongruent and cannot be seamlessly merged when scholars in these fields work independently on the same problem. For example, the "knowledge as mechanism" metaphor common in the natural sciences and engineering leads to a very different form of argumentation than the "knowledge as narrative" metaphor that is typical of the humanities and certain social sciences (Miller et al. 2008).

In order for future scholars to meaningfully integrate knowledge across disciplines, they must be exposed to different disciplinary approaches, preferably during their graduate training while their conceptions of knowledge are still being developed. Institutions of higher education have been challenged to reform curriculum and instruction that leads to the preparation of students with the skills, attitudes, and dispositions necessary for the 21st century. No longer is it acceptable for students to graduate from college with only domain specific knowledge from their major; but rather, students are expected to create innovative applications of content knowledge while working collaboratively with others to solve complex real-world problems. Unfortunately, the very structure of universities may hinder such knowledge creation or skill acquisition, as shown by our example where the challenges were as much administrative as conceptual. Nonetheless, interdisciplinary initiatives in graduate education and the general approach of constructivist learning are best suited to shape such initiatives, due to their emphasis on active learning (Yarrow and Millwater 1995).

The fundamental characteristic that makes the constructivist teaching method effective is the idea that learning is an iterative and integrative process where the student builds upon extant knowledge by taking an active role in the construction of knowledge formation (Piaget 1977). This reasoning has been incorporated by educators in constructivist learning environments which are student-centered and learner controlled, orienting instruction towards the active student learner and away from the passive information gatherer (Yarrow and Millwater 1995; Mara 2005). This constructivist learning paradigm has also been effectively extended to include technological innovations available in the contemporary classroom (Jonassen et al. 1999).

Interdisciplinary courses are based on the idea that real world problems are broad, complex, and require tools and information from a variety of different disciplines to formulate solutions. Courses are usually designed around a central topic, and then coursework incorporating different disciplines relevant to that topic is employed. Hollenbeck (2006) gives the example of the Dust Bowl of the 1930s, which is typically confined to a history class. The interdisciplinary class incorporates the historical components of the Dust Bowl, but also includes many scientific disciplines such as climatology, meteorology, ecology, and geology, which are all relevant to the event, but typically would not be incorporated. Students learn the effects, causes, and overall difference in life before and after this historically significant event using multiple disciplinary approaches.

Interdisciplinary learning integrates knowledge from multiple disciplines across a central program theme or focus (Ivanitskya et al. 2002) and requires understanding higher-order relationships and organizing principles (Goldsmith et al. 1991). The principles of constructivist learning theories applied in interdisciplinary courses support the development of student learning outcomes desired for the 21st century. The constructivist approach uses real world examples, and is built around the student designing pertinent questions and actively seeking answers, while the instructor operates as a resource to help the student find the information needed. Constructivist pedagogy also has students practice self-evaluation and extend learning concepts outside the classroom, and fosters the ability of students to apply concepts and skills to new situations (Yager 1991).

The Water and Society Course

Societal problems in natural resource management offer excellent opportunities for interdisciplinary collaboration. They usually combine the two aspects necessary to understand the greater picture: 1) the physical characteristics of a natural phenomenon, and 2) the social and economic drivers of human decision making. Supported by a grant from the CNH program of the National Science Foundation, an instructor team used water resource management as the theme of an experimental graduate course. The objective was to facilitate a constructivist learning experience in which students develop their understanding of the subject through a number of group assignments. Teamwork was emphasized based on the recognition that professionals will increasingly need to work in interdisciplinary teams to solve natural resource challenges.

The course has been offered five times (2006, 2007, 2008, 2011, and 2013) to small groups of master's and doctoral students, and was co-taught by an interdisciplinary team of six faculty from four different colleges. The course was open to any students, but targeted recruitment was employed via the disciplinary networks of the faculty involved. While the general topic remained the same, class assignments, deliverables, and assessment methods were occasionally changed from one year to another, following the annual evaluation results (Tables 1 and 2).

Water and Society started as a one credit hour class. Soon it became clear this hindered the interaction between students as well as between students and instructors due to limited class

Table 1. Basic course characteristics.

Year	Credit Hours	Weekly Meeting Times	Number of Faculty	Number of Students
2006	1	One hour weekly for 16 weeks	5	7
2007	1	One hour weekly for 16 weeks	5	6
2008	1	Two hours weekly for 8 weeks	6	7
2011	3	Three hours weekly for 16 weeks	6	5
2013	3	Three hours weekly for 16 weeks	6	9

Table 2. Disciplinary components and learning outcomes.

Year	Disciplines of Participating Students	Main Learning Outcomes	Main Student Deliverable
2006	Agricultural Economics Agronomy Civil Engineering Landscape Architecture Sociology	• Increasing understanding of water resources systems as coupled human and natural systems • The ability to explicate features of the system from multiple disciplinary viewpoints	Collaborative paper explaining the complexity of a water resource system
2007	Civil Engineering Landscape Architecture Sociology	• Increasing understanding of water resources systems as coupled human and natural systems • The ability to explicate features of the system from multiple disciplinary viewpoints	Collaborative paper explaining the complexity of a water resource system with a stakeholder role-play exercise
2008	Agricultural Economics Agronomy Civil Engineering Landscape Architecture Sociology	• Increasing understanding of water resources systems as coupled human and natural systems • The ability to explicate features of the system from multiple disciplinary viewpoints • The ability to quantitatively analyze interactions among various features of water resources systems	A spreadsheet demonstrating the interplay of various factors in a water resource system with a stakeholder scenario game
2011	Agronomy Civil Engineering Political Science	• Increasing understanding of water resources systems as coupled human and natural systems • The ability to explicate features of the system from multiple disciplinary viewpoints • The ability to quantitatively analyze interactions among various features of water resources systems • The ability to use a variety of analytical aids to develop a complex model of system interactions	Interdisciplinary computer model simulating the complexity of a water resource system
2013	Agricultural Economics Civil Engineering Landscape Architecture Sociology Political Science	• Increasing understanding of water resources systems as coupled human and natural systems • The ability to explicate features of the system from multiple disciplinary viewpoints • The ability to quantitatively analyze interactions among various features of water resources systems • The ability to use a variety of analytical aids to develop a complex model of system interactions	Interdisciplinary computer model simulating the complexity of a water resource system with stakeholder interaction

time. The adjustment in 2008 was to combine two class hours and have the course taught for eight weeks as opposed to sixteen. It helped the learning experience, but as the final deliverable became more complex, the credit hour number was eventually increased to three. At the same time the class returned to the 16 week teaching schedule. This helped facilitate student teamwork outside the class setting.

The class started with five faculty from various fields: agronomy, agricultural economics, civil engineering, landscape architecture, and sociology. Faculty turnover resulted in adding political science and computer science, and losing landscape architecture. The impact these changes had on the course could easily be accommodated by the constructivist pedagogy. With a few exceptions, all faculty members participated in all class sessions, with different professors serving as the main instructor depending on the topic of the day. Faculty participation in class discussion was intensive in the first few weeks to give as much collective background to the conceptualization process as possible, but more limited later when students were tasked to synthesize and extend what they had previously learned. This allowed a natural student transition from disciplinary learner to interdisciplinary problem-solver. The following overview provides a sample on how each discipline approached this main subject.

The *agronomic sciences* cover a broad range from basic research into biological, chemical, and physical aspects of plant and soil systems to applied studies and delivery of new technologies in soil tillage and amendment; crop breeding, genetics, and improved varieties; and integrated best management practices for crop production. The agronomy section in this class described the roles of plants as economic products of an agricultural system and also, at the physiological level, important valves both controlling water transport and creating the need for groundwater exploitation. For example, irrigation to meet plant transpiration needs comprises 95^{+} % of groundwater use in the geographic areas focused on by the class. In one year there was a felicitous synchronization between class scheduling and agricultural field research. It was possible to bring to class samples of experimental sorghum lines being bred for biofuels applications. These plants were visually striking (being over 12 feet tall) and provided an exceptional vehicle for explaining plant processes ranging from growth, development, water relations, plant genetic controls, and their manipulation via breeding.

Plant processes such as these have been the subject of computer simulation efforts for half a century. As the course incorporated more computer simulation, it was also possible to leverage this section to present "how to" information on modeling. This involved steps ranging from the identification of basic system components, to the mathematical description of the relationships governing their behavior, to the computer implementation of those equations, and ending with a systematic evaluation of model characteristics (e.g., stability and sensitivity) and verisimilitude to reality.

For *hydrology* the important properties that control hydrologic processes are mass, flux, pathways, and residence time. Across the globe, the mass of water resides in stores including surface water, glaciers, the atmosphere, and groundwater. Water fluxes occur within each store and pathways exist between the stores (e.g., a portion of precipitation seeps downward in soils past the root zone and eventually recharges groundwater). The residence time in each store varies on the order of hours/days in the atmosphere to years/millennia in glaciers and groundwater.

The *groundwater* component of the hydrologic cycle was illustrated in class using a sandbox demonstration. In this, recharge entered layers of alternating granular porous media, layers of clay that retarded the movement of water, and underlying limestone where water flowed through fractures. Mass was computed by multiplying the volume of media containing water times the porosity (volume of pores/total volume). Flux and pathways were visualized by injecting dye and observing travel times, and the impacts of human forcers, such as pumping water from wells. It was also observed that groundwater has a much larger residence time when traveling through pores between granular media than when traveling through fractures in limestone. The sandbox demonstration helped students visualize hydrologic processes and properties and taught the

differences between porous and fractured media flow, and basic concepts such as head (groundwater elevation) and discharge (pumping rates). These variables controlled the quantity of groundwater that was available for use as well as the flow rates that were available for extraction/diversion.

The *economics* section involved an integrative lecture and an experiential activity. The lecture portion provided an overview of the subdiscipline of environmental and natural resource economics with an emphasis on the types of questions commonly addressed in the field. The inherent links between economic, socio-cultural, and biophysical phenomena were discussed. An example of integrative cross-disciplinary research (Steward et al. 2009) was presented in which economic analysis was coupled with hydrologic modeling to address groundwater decline issues.

The experiential activity was a classroom experiment commonly used by experimental economists called the "common property game." In this exercise the subjects were joint owners of a resource and could make individual decisions about how much of the resource to extract. The game can be presented in various forms, but in our case the property owned was a (hypothetical) shared bank account with an initial balance of \$10 x *N*, where *N* was the number of students. Each student decided to withdraw an amount up to \$10. The remaining balance in the account after the combined withdrawals was multiplied by an interest rate and the final balance was divided equally among the students. Thus, each student's earnings were the amount initially withdrawn and his or her share of the final balance. The maximum group earnings occurred when everyone withdrew zero dollars and the full initial balance accrued interest. At the same time, the individual incentive was to withdraw at least some of the funds initially. As is typical of the results from literature, students chose to withdraw nonzero amounts initially, reducing the group gain. These results and the similarity of the incentives in this game to those of pumping water from a common pool aquifer were discussed.

The *political science* section of the course provided an introduction to the study in policy analysis and policy sciences. Given that most of the students enrolled came from engineering or agronomy backgrounds, this section emphasized the social scientific nature of this field of study. The purpose was to make sure that students understand there is a disciplinary framework for the integration of knowledge from several fields of study to help study and resolve water policy issues.

The section began by noting that political science entails search for systematic explanations for political phenomena. In the field of political science, aggregate patterns of behavior are studied rather than individuals or singular events, and theoretical models are probabilistic and nomothetic. The next part of the discussion covered the differences among policy studies, policy analysis, and policy sciences. *Policy studies* focus narrowly on one policy area using methods from primarily one academic discipline, such as cost-benefit analysis from economics. By contrast, the subfield of *policy analysis* is the process of multidisciplinary inquiry designed to create, critically assess, and communicate information that is useful for understanding, developing, and/or improving policies. *Policy science* is a holistic, multidisciplinary approach, which focuses on solving problems by integrating both qualitative and quantitative information from all disciplines to design policy solutions.

The *sociology* component focused on the conceptualization of natural resource management, the power and inequality inherent to extractive industries, and the social demography of water resource systems via labor market outcomes. A brief lecture on social change and population dynamics illustrated the relationship between the two, and examples from various counties were used to demonstrate the impact of fertility, mortality, and migration on long-term population trends and population composition. The lecture included a discussion of sustainability at various levels of geography from local to global.

Following the lecture, students were given an introduction to population projections, discussing the two most commonly used methods: mathematical extrapolation and cohort component projection. It was demonstrated how various projections could differ based on the time period used for constructing the baseline assumptions. This was followed by a group exercise of a cohort component projection model. The students

discussed assumptions about fertility, mortality, and migration that were entered in the model in class, based on the 2000 population of a randomly selected rural county. The results of the population models were discussed with the objective of comparing the relative weight of each component in the final outcome. The discussion was then connected to potential policy initiatives addressing economic development options (based on the availability of water) and predicted population dynamics.

Two additional disciplines, landscape architecture and computer sciences, were used for additional technical insights in the course. *Landscape architecture* focused on the use of GIS tools in modeling various layers of the land from topography to human interactions. In addition, a unique game was developed around the various perspectives and elements of the water resource system in the state. The goal was to model how communities, with often contradicting perspectives within the community, deal with water allocation in an over appropriated water area. This game was crucial to add the land use perspective in scenario design and decision making.

The *computer science* section introduced the students to the practice of integrated modeling through a combination of group discussion and hands-on experimentation. A short lecture initiated the discussion and outlined the process of formalizing conceptual domain-level process interactions into concrete data exchanges between models. The discussion highlighted the primary challenges of model integration (e.g., resolving differences in programming languages) and the broad spectrum of approaches (e.g., linking models vs. combining them), with emphasis on the additional complexity involved in integrating multidisciplinary models. An integrated computer model that was part of an ongoing research study served as an applied case study of how disciplinary models for groundwater, economics-based crop decision making, and crop production can be linked together using the Open Modeling Interface (OpenMI) (Gregersen et al. 2007). The linked model provided the students with a context for both learning about the OpenMI and learning how to utilize an integrated model. Lab training on executing the linked model and inspecting the simulation output for different input data sets enabled the students to later utilize this model outside of class as part of the final class deliverable.

The number of students taking the course was small, although not unusually so for graduate seminars. Recruitment efforts were focused on getting a relatively even representation from the various disciplines. This worked best in 2006, 2008, and 2013, and created significant challenges in 2007 and 2011.

The main deliverable changed each time the course was taught. In the first year, which was a learning experience for the faculty as much as it was for the students, the course ended with a collaborative paper describing the interaction between the components of a complex water resource system. In the second year, a stakeholder roleplaying exercise was added when students had to present the findings as members of various stakeholder groups, arguing for individual interests while keeping the interdisciplinary perspective. This proved to be very useful, so the third year task was to develop a comprehensive game around these various interests and perspectives. The game included a complex spreadsheet modeling the interactions based on stakeholder decisions. During the final presentation, the students played the game on a sample county landscape game board representing an area with an aquifer. In the fourth year, the emphasis was on computer modeling which integrated a number of existing disciplinary components. In the last iteration, the computer model was supplemented by a live video presentation to faculty and students of a community college in the state (many of them stakeholders outside the classroom) who interacted with the graduate students on campus via videoconferences. That way the final deliverable was not only evaluated by the faculty, but also a stakeholder group in the region. The deliverables therefore evolved from developing a simple conceptual model to writing a more complex computer tool.

The course had three major modules, covering an approximately equal share of the 17 week course. The first module was the period of contrasting the students' own disciplinary perspectives on water resource management systems with a cross-disciplinary vision on the same issue. In this module, students developed an individual

diagram of a water resource system based on their own disciplinary training and background. Once completed, they were instructed to work as a group (which is why keeping the class size small was very important) to create a collective understanding of the issue. This was required to incorporate all disciplinary perspectives.

In the second module the students were exposed to various stakeholder interests and discipline-specific lectures in water resource management. These were highly interactive with discussions and reflective writing components on fundamental literature in each discipline. The purpose was to deepen each student's knowledge regarding the various fields represented in the course.

After the disciplinary perspectives, the third major module of the course focused on the synthesis of the material and developing the main student deliverable for the class. This module varied according to what the main deliverable in a given year was. In the first and second years, it was mostly group discussions. From the third year onward when computer modeling was involved, there was a more comprehensive work not only focusing on the conceptual links, but also on how these could be translated into actual calculations. In the fifth year when the interactive video presentation was added, the final module included a follow-up presentation to the distance stakeholders based on their questions and comments during the first session. A constructivist approach to teaching helped keep the third module flexible enough to revisit any particular fields that proved challenging to the students. The practice of all faculty being present in all sessions maximized the ability to respond to student inquiries and discuss tangents if needed.

The case study for the class in each year was the depleting groundwater of the Ogallala Aquifer. After learning about the disciplinary perspectives, students were required to develop a conceptual model of how various components of the water resource system interact with each other. To connect this model to applied questions, the dynamics had to follow the farmer's decision making considerations from both the natural and the socioeconomic systems (such as water availability and economic incentives).

Once the conceptual model was in place, students were required to collect data on the dynamics of the system, such as recharge rates, water used for irrigation, crop choices, employment structure, and demographic trends. Selecting one study county, students applied the collected data on the conceptual model, and integrated the two using the OpenMI framework. Finally, they were required to demonstrate various policy changing scenarios and their impact on the combined model. Part of this demonstration was done in a live videoconferencing situation where students and faculty from a community college in the study county had the opportunity to ask validating questions about the application of the model on that specific location.

The assessment of student work was as complex as the material covered in class. While there were minor changes for each new iteration of the course, the basic assessment structure remained the same and included both individual and team assessments. Disciplinary components were evaluated through reflective writing assignments that were graded by standard rubrics using Bloom's taxonomy of learning domains. Conceptual work was assessed through the individual and collective diagram of the water resource system, graded separately. The grading emphasis was on the inclusion of various elements of the system as well as on demonstrating the causal and non-causal mechanisms between those elements. Interdisciplinary data projection and model application were assessed as a team project by providing students a specific scenario which they had to apply in the model. Finally, the term paper and its oral presentation were assessed using a revised version of Bloom's taxonomy, putting more emphasis on application and self-evaluation at the team level.

An external evaluator was hired to document the process and the impact of using a constructivist approach (discovery-based, problem-based, cooperative learning, interdisciplinary) to teach the Water and Society course. The major evaluation questions focused on the process of implementation, the constructivist principals that were used, and the application of constructivist components.

A mixed methods evaluation design using both qualitative and quantitative data collection and analysis was implemented. On several occasions

evaluators were present during classroom discussions and other activities. A multi-case study (the student as unit of analysis) was used to explore the relationships among the following student variables: preferences for learning, performance, interaction, and perceptions of the extent to which teaching and learning elements of the constructivist classroom enhanced student's personal learning. Data collection techniques included document review, surveys, performance assessments, and interviews. The evaluation plan included a system of recursive review, analysis, and reporting to allow the instructors to make necessary modifications to the course during the duration of the semester. At the end of each year, the evaluator provided an annual review from which the team made modifications for the next installment of the course.

Discussion and Conclusions

The Water and Society course was taught five times between 2006 and 2013, using constructivist pedagogy to address complex interdisciplinary issues. The external evaluation revealed five themes as the noted benefits of this approach: 1) increased interest in the subject, 2) increased knowledge of water management issues, 3) utilization and communication of multiple perspectives, 4) enhanced problem-solving skills, and 5) improved communication and leadership skills. Most of these would probably apply to any other topic under the same pedagogy. In addition, among the benefits of this learning experience, students emphasized not only the content of the course, but also the applied learning methodology. This indicates a type of knowledge transfer which shows that constructivist pedagogy supports a sustained educational experience.

The constructivist approach provided significant flexibility for the instructors to accommodate not only the different disciplinary processes of graduate education, but also the differences in student background and training. In addition, this approach made it possible to adjust the course schedule and activities if students needed additional time to complete a team assignment or were looking for more in-depth explanation about a particular module or disciplinary context. Given that both the student background and the deliverable were different each year, constructivism was a necessary cornerstone of the instructional methods.

Based on external evaluations and feedback from faculty, an integrated interdisciplinary class setup further enhances the understanding of complex issues, and better prepares students for professional positions. It is widely understood that insights from other disciplines improve understanding of the big picture. Some of these insights can be acquired by taking classes in other academic fields, but putting teams of instructors and students from different fields together to work on a complex problem promotes learning new material and requires applying it at the same time. Such simultaneous knowledge acquisition and application is the best formula for retaining new information and skills.

This course brought up several challenges regarding interdisciplinary education at the graduate level, in addition to its successes. Some of these challenges were anticipated by the instructor team, while others emerged unexpectedly. We believe that these were general enough to merit a discussion and recommendations about course development in an interdisciplinary setting.

One challenge was to find students who could most benefit from the particular knowledge and skills offered in the class. The practice and understanding of interdisciplinary work varies across disciplines, therefore students in various fields had different expectations about the usefulness of the course. At the same time, an interdisciplinary student body was crucial for the success of the course. Recruitment was eventually done through individual faculty efforts where discipline-specific messages were used to target students in various programs. This was one aspect where the constructivist method was as much an obstacle as an advantage. The course structure was kept flexible, which meant that in the recruitment phase little information was available to give to the students, especially in the first years. The experience over the five years was that as the course moved from a conceptual to a more practical application, it was easier to advertise its benefits for students. The more personal, network-based student recruitment could overcome the lack of early information, but this would not allow marketing the class in the traditional manner.

A typical recruitment challenge was that it was not always possible to have students from every pertinent discipline each year. The success of the final deliverable, as well as the general understanding of the course material given the strong emphasis on student teamwork, was contingent upon having a variety of fields represented among the students. It was clearly visible how certain components of the final deliverable were weaker if a particular discipline was not represented. This challenge emerged in the second year, and the instructor response was the introduction of disciplinary modules. However, this could not fully address the problem as a student in one field cannot become an expert in another field after a few in-depth classes. This was particularly true across the natural sciences and social sciences division, and regarding fields that have substantial training in mathematics, such as engineering and computer science. Moreover, having a student from a particular program was not an assurance for successfully completing a given disciplinary task, as students were quite different even if they came from the same field. This was especially true for the social sciences where graduate education could take various forms toward the advanced degree. For example, a sociologist with an emphasis on demography would result in a different final deliverable than a sociologist with training in rural development and inequalities.

Interdisciplinary instruction is easier to imagine than to implement, and the first requirement is that the faculty team agrees on what constitutes knowledge that is transferrable across disciplines and which scientific vocabulary to use in education. Even simple terms such as 'theory' or 'models' mean vastly different things in different disciplines. In this case, the instructor team had been working together for years by the time the course was first offered, and there was a general agreement on how to develop the course. However, teaching this class was as much a learning experience for faculty as it was for the students. Certain aspects of the class, especially assessment, proved very difficult to administer with students used to different evaluation practices. One example of this was the balance between reflective writing assignments and computational exercises. The former was easier for students in social sciences, while the latter came more naturally to those in STEM fields. The relative weights of such assignments had to be carefully designed and evaluated so that a student would not fall behind because his or her discipline has less emphasis on a certain type of assessment.

Faculty members taught this course on an overload basis, without any financial compensation. While this is feasible for a single offering or a short series of offerings, its sustainability is questionable. For example, it was not possible to obtain a course number, because the university structure did not have one available. In the first few years, students took the class as independent study courses within their own disciplines. Since Water and Society not only cut across departments but also colleges, it was a long administrative process to develop a structure which made cross-listing the course with the 'GRAD' prefix possible. Questions to be solved included the allocation of credit hours, tuition income, and the distribution of various fees and expenses related to equipment use.

The growing emphasis on online education brought up the possibility of having both on campus and distance students in the class. In 2011 and 2013, the class had distance students as well. One challenge the team was not able to solve was the successful incorporation of distance students. The technical part of this challenge was less of a problem as the team utilized various technologies, such as a data repository for readings, recorded interactions for those missing class and working in distance mode, a message board to foster communications, and videoconference meetings. However, the intensive face to face teamwork with collective assignments that required several meetings outside of the class made it very difficult to involve distance students in class work. Despite the availability of various communication tools, this aspect is yet to be solved.

Finally, it is important to note the impact of interdisciplinary education on the students' future career. We know little about how interdisciplinary graduate training affects individuals once they enter the workforce. There are, of course, a number of reasons to expect it would be beneficial. Likely in response to the funding models at NSF and other agencies, there are a growing number of research and teaching jobs posted with "interdisciplinary" as a key word.

Presumably students with coursework like Water and Society are better positioned to obtain these jobs and should be better trained to meet the interdisciplinary expectations once they take them. However, support for interdisciplinary scholarship is highly developed in only a handful of institutions. Accordingly, some of the students who find themselves in purportedly interdisciplinary jobs still face a number of obstacles (limited publication opportunities, high startup costs and long learning times, lack of appreciation for interdisciplinary work in promotion and tenure decisions, etc.). Overall the effect of such coursework on future employment and success is unclear, and this calls for future research on tracking those students who participated in interdisciplinary education.

The Water and Society course has been a successful experiment in interdisciplinary graduate education. Student response to the evaluation was overwhelmingly positive. Being challenged to think outside the box and develop a collaborative deliverable more than compensated for the requirements of learning about fields that a student would not normally know much about. The course helped deconstruct the artificial barriers between academic disciplines, as well as stimulate further interdisciplinary collaboration. In addition, senior faculty with many years of experience in various academic fields were challenged to advance their own knowledge, which in turn could open up new opportunities for future research projects.

While the topic of the course focused on the use of water resources, many aspects of this experience could be generalized to advance interdisciplinary courses covering other subjects. The scholarly environment fostered in Water and Society would suit any pedagogy which moves from knowledge acquisition to knowledge construction. None of the challenges we observed are insurmountable. Since academic knowledge has been advancing toward interdisciplinary perspectives and applications, graduate courses like this one are likely to be increasingly important components of higher education.

Acknowledgements

This study was supported by the National Science Foundation's CNH program (award #: 0909515).

Author Bio and Contact Information

Dr. László J. Kulcsár (corresponding author) is a Professor of Sociology and Department Head at the Department of Sociology, Anthropology and Social Work at Kansas State University. His field of expertise is social demography, with a particular emphasis on migration, urbanization, aging, and regional development. Dr. Kulcsár holds a Ph.D. from Cornell University in development sociology, and has master's degrees in sociology and in public administration studies. Prior to his academic career he was a consultant on various European Union rural development projects, and worked in the poll opinion industry in Hungary. He can be reached at kulcsar@ksu.edu and at 204 Waters Hall, Manhattan, KS 66506.

Dr. Joseph A. Aistrup is currently a Professor of Political Science and Dean of the College of Liberal Arts at Auburn University. His research of the High Plains Aquifer region in Kansas focuses on agricultural water use policy and analysis, sustainable uses of groundwater for agriculture, rural community development and rural population growth. He was previously a Professor of Political Science at Kansas State University. In 1989, he received his Ph.D. in Political Science from Indiana University at Bloomington. His direct contact information is jaistrup@auburn.edu and 231 Tichenor Hall, Auburn University, Auburn, AL 36849.

Thomas Bulatewicz is currently the Senior Mobile Architect at ExakTime, Inc. He was previously Adjunct Faculty in the Department of Computing and Information Sciences at Kansas State University where he conducted research in high performance computing and software tools for model integration and coupling. He received a Ph.D. in Computer Science from the University of Oregon in 2006. E-mail contact is tombz@nosleep.net.

Jeffrey M. Peterson is the Director of the Water Resources Center and a professor of Applied Economics at the University of Minnesota. His research focuses on environmental policy analysis with an emphasis on water use and water quality impacts related to agriculture. He earned his Ph.D. from Cornell University in Agricultural and Resource Economics. Prior to coming to Minnesota he was a faculty member in Agricultural Economics at Kansas State University and also held a 1-year visiting faculty appointment at Oregon State University. He can be reached at jmpeter@umn.edu or 173 McNeal Hall, 1985 Buford Avenue, St. Paul, MN, 55108.

Stephen M. Welch currently a Professor of Agronomy at Kansas State University. His specialty is computer simulation of crops, especially interdisciplinary basic

science aspects needed to improve model predictive skills in areas important to future food security. Previous appointments at Kansas State were in the Department of Entomology, the College of Agriculture computer support office, and the (now renamed) campus-wide Computing and Network Services unit. He has a BS in Computer Science and a Ph.D. in Zoology, both from Michigan State University. His direct contact information is welchsm@ksu.edu and 2004 Throckmorton Plant Sciences Center, KSU, Manhattan, KS 66506.

David R. Steward is a professor in the Department of Civil Engineering at Kansas State University. His research addresses groundwater sustainability challenges such as groundwater declines and groundwater/surface water interactions. He is a lead developer of the Analytic Element Method, which provides nearly exact solutions to complicated boundary value problems commonly encountered in groundwater and related fields of engineering and science. He received a Ph.D. in Civil Engineering, along with MS degrees in Civil Engineering and Mathematics, and BSCE all from the University of Minnesota. E-mail contact is steward@ksu.edu.

References

Goldsmith, T.E., P.J. Johnson, and W.H. Acton. 1991. Assessing structural knowledge. *Journal of Educational Psychology* 83: 88-96.

Gregersen, J.B., P.J.A. Gijsbers, and S.J.P. Westen. 2007. OpenMI: Open Modeling Interface. *Journal of Hydroinformatics* 9: 175-191.

Grimm, N.B., J.M. Grove, C.L. Redman, and S.T.A. Pickett. 2000. Integrated approaches to long-term studies of urban ecological systems. *Bioscience* 50: 571-584.

Hollenbeck, J.E. 2006. Making interdisciplinary courses work with constructivism and science, technology and society. *College Quarterly* 9(2).

Ivanitskaya, L., D. Clark, G. Montgomery, and R. Primeau. 2002. Interdisciplinary learning: Process and outcomes. *Innovative Higher Education* 27: 95-110.

Jonassen, D.H., K.L. Peck, and B.G. Wilson. 1999. *Learning with Technology: A Constructivist Perspective.* Merrill, Upper Saddle River, New Jersey.

Kinzig, A.P. 2001. Bridging disciplinary divides to address environmental and intellectual challenges. *Ecosystems* 4: 709-715.

Lubchenco, J. 1998. Entering the century of the environment: A new social contract for science. *Science* 279: 491-497.

Mara, R. 2005. Teacher beliefs: The impact of the design of constructivist learning environments on instructor epistemologies. *Learning Environments Research* 8: 135-155.

Marsh, G.P. 1864. *Man and Nature*. Reprinted in 1965. Belknap Press of Harvard University Press, Cambridge, MA.

Miller, T.R., T.D. Baird, C.M. Littlefield, G. Kofinas, S.F. Chapin III, and C.L. Redman. 2008. Epistemological pluralism: Reorganizing interdisciplinary research. *Ecology and Society* 13: 46-62.

Piaget, J. 1977. *The Development of Thought: Equilibration of Cognitive Structures.* The Viking Press, New York, NY.

Redman, C.L., J.M. Grove, and L.H. Kuby. 2004. Integrating social science into the Long-term Ecological Research (LTER) network: Social dimensions of ecological change and ecological dimensions of social change. *Ecosystems* 7: 161-171.

Stern, P.C., O.R. Young, and D. Druckman. 1992. *Global Environmental Change: Understanding the Human Dimensions*. National Academy Press, Washington DC.

Steward, D.R., J.M. Peterson, X. Yang, T. Bulatewicz, M. Herrera-Rodriguez, D. Mao, and N. Hendricks. 2009. Groundwater economics: An object-oriented foundation for integrated studies of irrigated agricultural systems. *Water Resources Research* 45: W05430.

Yager, R.E. 1991. The constructivist learning model: Towards real reform in science education. *Science Teacher* 58(6): 52-57.

Yarrow, A. and J. Millwater. 1995. SMILE: Student modification in learning environments: Establishing congruence between actual and preferred classroom learning environment. *Journal of Classroom Interaction* 30*:* 11-15.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 132-147, August 2016

Deliberating on Missouri River Water Diversions in Congressional Committee Hearings

Sarah Michaels[1] and *Mike Gruszczynski[2]

[1]University of Nebraska-Lincoln, Lincoln, NE
[2]Austin Peay State University, Clarksville, TN
*Corresponding Author

Abstract: We examine the discursive influence of competing interest groups and how and when U.S. federal legislators use empirical evidence to consider Missouri River diversion projects originating in the relief, recovery, and reform thinking behind President Franklin D. Roosevelt's New Deal. Twelve U.S. Congressional committee hearings that occurred between 1944 and 2006 are content analyzed to demonstrate the nature of instrumental arguments in legislative deliberations. Members of Congress in favor of diversion projects used more economic-based, instrumental arguments than interest group representatives sharing their views while members of Congress opposed to the diversion projects consistently used less science-based, instrumental arguments than did interest group representatives sharing their stance. As a percentage, more instrumental arguments were made in field hearings than in Washington, D.C. hearings. While more economic-based, instrumental arguments were made in Senate-related hearings than in House of Representatives related hearings, in the House more science-based, instrumental arguments were made than in the Senate.

Keywords: *evidence-based policy making, Congressional committee hearings, instrumental arguments, Missouri River, Garrison Diversion Unit*

Evidence-based policymaking, in which legislators draw on best available information from a wide range of reputable sources, has become part of the gold standard of modern policymaking practice in the first part of the twenty-first century (Cabinet Office 1999; Bullock et al. 2001; Michaels et al. 2014). Consequently, appreciating the nature of legislative arguments becomes essential to developing an understanding of water policy debates. To that end, we present a critical case in which we examine the nature of the arguments made by supporters and opponents of developing water management infrastructure concerning the Missouri River in the United States. We focus on instrumental arguments, constituting falsifiable, research-based reasoning intended to outline causal relationships between intervention and outcomes of the proposed policy as a function of the policy (Esterling 2004). As did Fisher et al. (2013b), we focused on statements involving science and economics as representing instrumental arguments.

Why Look at Information in the Legislative Process?

Decision makers draw on information supportive of their stances in discussing specific solutions (Brewer 1973) and use it to further political ends when arguing with those presenting conflicting views (Weible 2008). Scientific information is used to contribute to a deeper and more accurate understanding of fundamental processes of the substance of policy and also as a tool in political deliberations (Caldwell 1984; Litfin 2000; Nicholson-Crotty 2005; Weible 2008; Montpetit 2011) because it is seen to lend political legitimacy (Smith 1992; Demeritt 2001; Keller 2009). In a similar vein, economic advice, specifically cost-benefit analysis, is used to evaluate policy alternatives (Weingast et al. 1981).

In the United States there is a premium placed on "rational" decision making. In Congress this plays out as valuing "a process in which alternatives are identified and the costs and benefits of major alternatives are considered before a choice is made" (Weiss 1989, 417). Science is used as an important means for legitimating government decisions because of a prevailing belief in the rationality of science among those closely associated with the policy process in the rationality of science (Jasanoff 1997; Konig and Jasanoff 2002; Keller 2009). Yet scientific evidence, even when it is highly salient, is not necessarily the basis of the arguments made by proponents and opponents of reform (Appollonio and Bero 2009).

Environmental and Water Legislation

While much scholarship on environmental policy has scrutinized the emergence and implementation of landmark environmental legislation from the second half of the twentieth century (for example, see Sabatier 1975 on the 1970 Federal Air Quality Amendments and the 1972 Federal Water Pollution Control Act Amendments; Czech and Krausman 2001 on the 1973 Endangered Species Act; Stern et al. 2010 on the 1969 National Environmental Policy Act), we contribute to the literature by considering the evolution of deliberations originating in New Deal thinking about "harnessing" natural resources and alleviating unemployment (Pisani 2002). This is valuable because the way in which natural resources are managed is a function of legislation emerging from different eras of thinking about human relations to the natural world. It is vital to acknowledge the historical premises coloring deliberations about changes to the landscape (Kraft and Vig 2010).

Since the early 1820s Congress has recognized the federal government's responsibility to develop water resources (Ferejohn 1974). Historically, U.S. federal legislation pertaining to rivers has been among the most distributive of all policies (Krehbiel 1991), in which economic benefits are concentrated geographically, including expenditures for the projects' inputs, and costs are spread nationwide through general taxation (Shepsle and Weingast 1981). We examine Congressional deliberations because one of Congress' critical roles in interstate river governance is appropriating funding for federal dams (Seligman 2008). Authority over federal funding of infrastructure enabling cross-border diversions is key because the United States has maintained it has absolute sovereignty over territorial waters including boundary waters within its jurisdiction (Hearne 2007).

Diverting Water from the Missouri River

Diverting water from the Missouri River has been a longstanding goal in North Dakota, at one time the United States' most rural state (United States 1975). Arthur A. Link, Governor of North Dakota and Chairman of the North Dakota Water Commission explained in his 1975 testimony in Bismarck, ND before the Conservation, Energy, and Natural Resources Subcommittee of the Committee of Government Operations: "A Missouri River diversion in our State has been the object of well over a half century of effort by many public-spirited North Dakotans. Over the years, State administrations have consistently supported and worked for the development of the project. Our entire congressional delegation has consistently supported the project, as have the State's major farm organizations" (United States 1975, 6-7).

The Missouri River constitutes 96% of the state's surface water (United States 2006) and is the state's primary reliable water resource other than limited groundwater supply (Bartlett 1988). With an economy where agriculture is a key component and a semi-arid climate with annual rainfall in the range of 8 to 16 inches, the appeal of very stable water supplies is understandable. This is particularly true in light of the adverse consequences experienced in the state from extreme weather conditions, such as occurred in the drought associated with the Depression of the 1930s (United States 1975).

The origin of the Garrison Diversion is tied to the 1944 Pick-Sloan Act, which authorized a series of large dams, operated by the U.S. Corps of Engineers, on the Missouri River mainstream between Fort Peck, Montana and Sioux City, Iowa, including Garrison Dam in central North Dakota. Lake Sakakawea, which impounds the water behind the Garrison Dam, is the third largest

reservoir in the United States, after Lake Mead at Hoover Dam and Lake Powell at Glen Canyon Dam (Billington and Jackson 2006). In 1957 the Bureau of Reclamation recommended diverting water from the Garrison Dam and Lake Sakakawea to irrigate over one million acres along with other water development in central and eastern North Dakota. In 1965 Congress enacted legislation for the Garrison Diversion Unit for 250,000 irrigated acres and for municipal and industrial use, fish and wildlife habitat, and recreation. In 1984 construction was halted because of concerns about the Diversion Unit's environmental impact, interbasin water transfer from the Missouri River Basin into the binational (U.S. and Canada) Hudson Bay Basin, land acquisition, and irrigation economics. The passage of the Garrison Diversion Unit Reformulation Act of 1986 authorized the 1984 Garrison Diversion Commission's recommendations including irrigating 130,940 acres, none of which drain outside the Missouri River Basin. The decision by President George H.W. Bush's administration not to fund the Garrison Diversion Unit Project for fiscal year 1991 spurred the Secretary of the Interior to form a task group to develop a policy on funding the authorized project. The Garrison Diversion Reformulation Act of 1986 was further amended by the Dakota Water Resources Act of 2000 (Garrison Diversion Conservancy District 2014). In sum, in successive iterations the extent of irrigation proposed for the Garrison Diversion Unit has been downscaled.

Diverting water from the Missouri River was introduced to Congress on the presumption it was a socially efficient policy (United States 1944a; United States 1944b). As such the expectation was that it served "national interests by creating new opportunities to meet society's aspirations for public policy" (Esterling 2004, 76). The large-scale infrastructure construction was seen as the means to generate jobs for returning World War II veterans in dam construction and in the farming of irrigable lands made possible by dam construction (United States 1944a; United States 1944b). When the predecessor to the Garrison Diversion, a Missouri River based irrigation project, was not initiated at the outset, subsequent and progressively smaller iterations of the project failed to satisfy increasingly demanding benefit-cost evaluation criterion (Caldwell 1984). As one congressional staff member familiar with public works programs noted, "The logic in this program was lost long ago" (Grier 1983).

Congress and Congressional Committee Hearings

Esterling (2004) rejects the "textbook" depiction of Congress predicated on the portrait of the U.S. legislative process as a direct democracy having little regard for policy expertise. In his revised depiction of Congress, citizens may be willing to encourage their representatives to use expertise in policy deliberations and members of Congress who deploy expertise in researching and developing policy are rewarded professionally for doing so.

We look to Congressional committees because within the U.S. Congress, the committee system is the primary means by which information is assembled, analyzed, and disseminated (Arnold 1990). Members of Congress seek out committees where they can serve their constituents' interests directly, including "channeling additional federal money into ventures that benefit them [constituents]" (Weiss 1989, 413). Consequently, the bias in committee recruitment is in favor of those who advocate for the specific mission of a committee (Weiss 1989).

The initial screening of information is important in deciding the final outcome (Schneier and Gross 1993). As Jones and Baumgartner (2004) note, the most pronounced effect of information flows often occur early in the policy process. Information provided through testimonies has often been overlooked as a factor in determining Congressional action since it is not necessarily the most influential consideration (Schneier and Gross 1993; Burstein and Hirsh 2007). There is no consensus an unambiguous connection exists between information presented at Congressional hearings and Congressional action. Those rejecting such a linkage point out that not much information is provided at hearings, rather hearings are theatrical events for advertising and position taking in which witnesses are chosen for reasons of politics and what is said by them is known in advance. Furthermore, members of Congress may already be familiar with the information provided at hearings,

particularly if the issue being discussed is important, has been on the agenda for a while, or the legislator is a long-time member of Congress (Mayhew 1974; Diermeier and Feddersen 2000; United States 2006; Burstein and Hirsh 2007; Oleszek 2011).

These criticisms suggest why hearings might be dismissed as symbolic gestures (Oleszek 2001) and public relations maneuvers (Olzak and Soule 2009) and why there is a lack of agreement among scholars about the precise role of Congressional hearings (Truman 1951; Huitt 1954; Talbert et al. 1995). Yet hearings raise the profile of an issue (King et al. 2007), framing it for the press and the public (Maguire 2013) and are an essential component of the process of agenda setting (King et al. 2007). Arnold (1990), Baumgartner and Jones (1993), Burstein and Hirsh (2007), Liu et al. (2011), Gruszczynski and Michaels (2012), and Fisher et al. (2013a) note hearings are key sources of information in the U.S. legislative process. Fisher et al. (2013b) in particular explain how hearings provide a venue for clashing scientific, political, and economic interests.

Esterling (2004, 257) contends, based on qualitative interviews and statistical evidence, that hearings reflect what committees do behind the scenes and therefore "serve as an empirically valid window on the overall legislative process." He proposes that these hearings "present a useful forum for testing hypotheses about legislative debate, since they are public, fully transcribed, typically focus on a specific policy topic, and committee members and their staff often have the background knowledge and specialization sufficient for informed discussion" (Esterling 2004, 253). Thus, analyzing public hearings provides a means for learning how policy issues are framed and what types of arguments and evidence are presented (Appollonio and Bero 2009).

Constructing the Hypotheses

Esterling's approach (2004; 2007; 2009; 2011) to argument construction in legislative bodies is predicated on members of Congress benefitting from active interest groups' expertise on the effects of policies. That proponents and opponents of a policy each form a coalition supportive of their respective views is a feature of what Dahl (1961) and Truman (1951) identify as a pluralist understanding of how the political process works. For example, Cheon and Urpelainen (2013) use renewable electricity generation to demonstrate the importance of the countervailing effects on environmental policy created by competing interest groups. Yet, while legislators are consistent actors in Congressional deliberations, competing interest groups and their countervailing effects (Cheon and Urpelainen 2013) have not always been ubiquitous in all Congressional arenas. For example, when the dams on the main stem of the Missouri River were proposed in the 1944 Flood Control Act, there was far less organized support and opposition than emerged subsequently about the Garrison Diversion (United States 1944a; United States 1944b; United States 1975; United States 1998).

To understand the discursive influence of the competition between active, opposing interest groups integral to Esterling's conceptualization, we pose the following hypotheses:

> *H1*: The number of instrumental arguments made by members of Congress will not vary as a function of argumentative competition between interest groups in favor or opposed to Missouri River related diversion projects.
>
> *H2*: In hearings when interest groups provide testimony, elected officials' use of instrumental arguments will be consistent with the arguments of interest groups who share their positions.

Esterling (2004) speculates that because Republicans are more concerned about aggregate wealth than Democrats they will be more often the targets of instrumental arguments than Democrats. To test this we pose our third hypothesis:

> *H3*: More instrumental arguments will be employed in committee hearings taking place in Republican-majority Congressional chambers than in Democratic-majority Congressional chambers.

We anticipate the venue of the hearing, whether it is held in the U.S. Capitol Complex in Washington, D.C. or outside Washington, D.C., will affect the extent to which instrumental arguments are presented. To test this we pose our fourth hypothesis:

> *H4:* Field hearings will include fewer instrumental arguments than hearings held in Washington, D.C.

While field hearings and Capitol Hill hearings involve some different administrative considerations, much of the procedural requirements are the same (Heitshusen 2007). Field hearings provide local venues for federal legislators to demonstrate to their constituents that they identify and empathize with them (Esterling 2007), their "fiduciary behavior" is consistent with constituents' policy outcome preferences (Bianco 1994, 12), and their thinking and response to issues are in accord with whom they represent (Fenno 1978). As such, field hearings would seem to be venues where it would be understandable to emphasize the external effects and intrinsic desirability (Caplan 1977; Webber 1984) of the federally funded works proposed.

Based on previous scholarship, we anticipate a difference between Senate and House committee hearings in the amount of instrumental arguments presented. Esterling (2011) and Mucciaroni and Quirk (2006) found evidence-based arguments were more prevalent in Senate committees than in House committees. To test this we pose our fifth hypothesis:

> *H5*: Senate hearings will feature a greater number of instrumental arguments than hearings in the House of Representatives.

Methodology

Using content analysis, we systematically measure the information witnesses present during Congressional hearings (Esterling 2004; Burstein and Hirsh 2007; Keller 2009) through the use of the transcripts from hearings (see Huitt 1954; King 1993; Esterling 2004; Burstein and Hirsh 2007; Keller 2009). Content analysis has a long tradition in public policy research, especially for probing Lasswell's definition of the symbolic meanings of politics (Lasswell and Leites 1949; Edelman 1964; 1971). In the past, most content analyses focused on framing public policies based upon media depictions rather than on the processes followed by policymaking institutions (see Huitt 1954; King 1993; Burstein and Hirsh 2007; Newton 2008; Harris 2010; Tzoumis 2001).

Table 1 lists the 12 Congressional committee hearings we content analyzed. These hearings encompass all legislative committee hearing deliberations about construction of Garrison-related water management infrastructure projects in the U.S. Congress. To ensure we obtained documentation for all of the relevant hearings, we searched the *LexisNexis* Congressional database and the *Library of Congress* database. The earliest hearings we analyzed centered on implementing the Flood Control Act of 1944, authorizing multipurpose dams on the Missouri River. Hearings from the 1950s through the 1980s focused on irrigation projects authorized by the Flood Control Act of 1944 that had not been implemented yet. The hearings of 1998 and 2006 considered implementing elements of the Garrison Diversion not yet developed. Although our data is time-series in nature, we analyze the hearings as a set, examining how instrumental arguments are made in hearings pertaining to one group of projects.

The texts of the Congressional hearings listed in Table 1 were obtained from a variety of sources. Post-1980 hearings were obtained through the web site of the U.S. Congress. Pre-1980 digital copies of hearings were more difficult to come by. Consequently, we either obtained and digitized physical copies ourselves or obtained digitized copies from university libraries. We then coded for statements made by speakers in reference to proposed water infrastructure projects in an Excel spreadsheet. In all, 3,212 statements from written and oral testimonies were coded for the presence and type of either scientific information about the biophysical world or economic information related to policy.

To test for the reliability of the coding scheme employed, a second coder content analyzed 10% of a random sample of pages from each hearing document, which accounted for 34% of the total number of statements coded in the analysis. Krippendorff's alpha was used as the reliability statistic; given that measure accounts for both the overall congruence of rater agreement and the probability that coding disagreements are the result of chance alone (see Krippendorff 2003). Our Krippendorff's alpha of $\alpha = .968$ indicates that our coding exhibited very high reliability.

Defining Instrumental Arguments

Table 1 lists the percentage of statements by hearings that were science-based and economic-

Table 1. Proportion of instrumental arguments that were science-based and economic-based by Congressional committee hearing.

Hearing Year[a]	Chamber with which Committee Associated	Science-based Statements	Economic-based Statements	All Statements
1944	House	0% (0)	0.80% (3)	377
1944	Senate	0% (0)	0.55% (1)	183
1957[1]	House	0.56% (1)	7.39% (13)	176
1960	House	0% (0)	7.50% (9)	120
1961	Senate	0.56% (1)	10.67% (19)	178
1963	Senate	0% (0)	9.10% (25)	275
1964	House	1.05% (5)	4.53% (21)	474
1965	Senate	0% (0)	24.0% (6)	25
1975	House	44.19% (171)	5.17% (20)	387
1989[1]	Senate	0.47% (1)	17.21% (37)	215
1998[2]	House[2]	5.85% (10)	12.87% (22)	171
2006[1,2]	Senate[2]	31.58% (84)	7.14% (19)	266

[a] Specific hearings are listed in the References section as "*United States. Year*."
[1] Field hearing; no superscript indicates held in Washington, D.C.
[2] Chamber controlled by Republicans; no superscript indicates controlled by Democrats.

based. We coded statements as science-based instrumental arguments if they referred to broad science constructs, such as water quality, or to particular measures, such as concentrations of nitrates in the Missouri River. We coded statements as economic-based instrumental arguments if they referred to specific benefits, such as the creation of jobs or tax base growth, or impacts of the proposed projects, such as the economic impacts of flooding or the loss of tax revenue from flooded farmland. We coded each statement for the stance of the witness, witness affiliation, whether the statement was made within the hearing or in submitted materials, and argument type.

The following is an example of a science-based instrumental argument from a House of Representatives Hearing (United States 1975).

> *The FES [Final Environmental Statement] also indicates that the Red River presently has an average total dissolved solids concentration of 350 milligrams per liter with ranges from 200 to 550 milligrams per liter. Although there are questions about the reliability of the Missouri River data, even if this data is accepted as reliable, it is apparent that the water quality of the Red River will be degraded by Garrison.*

The following is an example of an economic-based instrumental argument, from a Senate hearing (United States 2006a).

> *The Bureau is risking a large federal cost with little benefit. It is completely unclear that this project is justified based on the analysis to date. Furthermore, the benefits determined by*

the Bureau are likely to be greatly distorted by the unrealistic population and industrial use projections.

In preparing the data for analysis we aggregated the data to percentages of science-based and economic-based arguments used by each speaker within each hearing. For the first hypothesis in which we posit that members of Congress will not alter the amount of instrumental rationale used as a function of interest group competition, we created a ratio of interest group competition by subtracting the number of groups testifying in opposition to these manipulations from the number of groups testifying in support of the manipulations, and then dividing by the sum total of all interest groups in each hearing. This nets a ratio of interest group competition that ranges from -1 (all interest groups testifying were opposed to the manipulation) through 0 (equal number of interest groups for/against the manipulation) to +1 (all interest groups testifying were in favor of the manipulation). The caveat is that we did not disaggregate groups into categories based on numbers of members or political influence. For the other hypotheses, dichotomous variables were created to indicate the presence (coded '1') or absence of either scientific- or economic-based arguments.

Results

We begin by testing Hypothesis H1, that the number of instrumental arguments made by members of Congress will not vary as a function of argumentative competition between interest groups in favor or opposed to Missouri River related diversion projects. What we uncover, as shown in Table 2, is that the substance of the instrumental argument matters. The use of science-based instrumental arguments by members of Congress was negatively correlated with our interest group competition variable ($r = -0.898$, $p < 0.001$). This indicates that when groups in favor of diversion-related projects dominate the hearings (the competition variable approaches +1), use of science-based instrumental arguments by Congressional members decreased. Conversely, as our competition variable moves in the opposite direction (closer to -1, indicating more dominance of groups opposed), use of science-based arguments increased. The presence of interest group competition on this particular policy issue influences the extent to which members of Congress engage in science-based reasoning within the hearings. Hypothesis H1 is thus rejected for science-based, instrumental arguments.

Examining the correlation between use of economic-based instrumental arguments by members of Congress and interest group competition paints a different picture. Though a moderate correlation was found between economic-based instrumental arguments and interest group competition ($r = 0.324$, $p = 0.331$), the correlation was not statistically significant, which might be attributed in part to the small sample size in our data. Thus we reject Hypothesis H1 as it relates to economic arguments.

Table 2. H1 – Pearson's *r* correlations between interest group competition and use of instrumental arguments by members of Congress.

Instrumental Arguments by Members of Congress	Interest Group Competition Ratio
Science-based	-0.898 ($p < 0.001$)
Economic-based	0.324 ($p = 0.331$)

We turn now to H2, that in hearings when interest groups provide testimony, elected officials' use of instrumental arguments will be consistent with the arguments of interest groups who share their positions. As before, we consider separately science-based and economic-based instrumental arguments. Table 3 presents the breakdown in the proportion of science-based instrumental arguments by members of Congress and interest groups. Proponents of the projects, whether serving as members of Congress or representing interest groups, employed a very low proportion of science-based, instrumental arguments. That said, there was still a significant pattern of relationship in the proportion of science-based arguments used between the two groups ($\chi^2 = 5.710$, $p < 0.05$), with members of Congress using more than double the science-based statements (2.89%) than interest group representatives (1.00%). The use of science-based arguments by opponents of the projects was

also significantly different between the groups ($\chi^2 = 13.110$, $p < 0.01$), with representatives of interest groups opposed to the projects employing three times the percentage of science-based arguments (52.29%) compared to members of Congress opposed to the projects (17.65%). On the whole, this is evidence that H2 is rejected, as little overlap appeared to occur in the use of instrumental arguments, whether in terms of proponents or opponents of the projects.

Table 3. H2 – Proportion of science-based arguments by members of Congress and interest groups.

	Positive	Negative
Members of Congress	2.89% (10)	52.29% (137)
Interest group representatives	1.00% (2)	17.65% (6)
Chi-Square test for column differences	$\chi^2 = 5.710$, $p < 0.05$	$\chi^2 = 13.110$, $p < 0.01$

Table 4 presents the breakdown in the proportion of economic-based instrumental arguments by members of Congress and interest group representatives. Those in favor of the projects employed more economic-based arguments than did those opposed, and a significantly higher ($\chi^2 = 11.258$, $p < 0.01$) proportion of economic-based statements were made by members of Congress (13.58%) compared to representatives of interest groups sharing their stance (6.18%). Both members of Congress and representatives of interest groups opposed to diversion-related projects employed a low proportion of economic-based arguments with no significant difference between them ($\chi^2 = 0$, $p = 1.0$). The lack of significance is explained by there being only one economic-based statement by interest groups opposed to diversion-related projects and Chi-square tests need more than one value in a cell to compute. That said, H2, as it relates to economic-based arguments, is rejected.

In summarizing our findings about Hypotheses 1 and 2 we can say the dynamic between the arguments of members of Congress and those of interest groups is more nuanced than we anticipated. For example, in H1 there was no systematic relationship between economic-based instrumental arguments by members of Congress and interest group competition. In H2 there was no systematic consistency in the use of instrumental arguments between members of Congress and representatives of interest groups on the same side of the issue. Importantly, we consider the likelihood that systematic differences might exist between North Dakota legislators and legislators from outside the state; to that end, we conducted additional analyses of the use of instrumental arguments. We found only small differences (one fewer statement by North Dakota legislators) in the case of science-based instrumental arguments and no differences in the case of economic-based instrumental arguments.

Table 4. H2 – Proportion of economic-based arguments by members of Congress and interest groups.

	Positive	Negative
Members of Congress	13.58% (47)	2.94% (1)
Interest group representatives	6.18% (26)	4.58% (12)
Chi-Square test for column differences	$\chi^2 = 11.258$, $p < 0.01$	$\chi^2 = 0$, $p = 1$

We turn next to Hypothesis 3; more instrumental arguments will be employed in committee hearings taking place in Republican-majority Congressional chambers than in Democratic-majority Congressional chambers. Table 5 presents the use of scientific and economic instrumental arguments employed in Congressional committee hearings with either majority Democratic or Republican memberships. There was a significant difference ($\chi^2 = 115.757$, $p < 0.01$) in the proportion of science-based statements when Republicans controlled a Congressional chamber (27.41%) compared to when a Congressional chamber was under Democratic control (8.02%).

This pattern of relationship holds for the use of economic-based instrumental arguments in Congress ($\chi^2 = 5.646$, $p < 0.05$), with Republican control associated with the use of more economic statements (10.35%) than when there is a Democratic majority chamber (6.83%) (Table 5). As stated previously, we coded statements as economic-based instrumental arguments if they referred to specific benefits, such as the creation of jobs or tax base growth, or impacts of the proposed projects, such as the economic impacts of flooding or the loss of tax revenue from flooded farmland.

Table 5. H3 – Proportion of science and economic-based arguments by majority party.

	Science	Economic
Democratic	8.02% (179)	6.83% (154)
Republican	27.41% (94)	10.35% (41)
Chi-Square test for column differences	$\chi^2 = 115.757$, $p < 0.01$	$\chi^2 = 5.646$, $p < 0.05$

As indicated previously in Table 1, Democrats were the majority party in the Congressional chamber associated with the hearings we analyzed in all but the last two instances: during the 1998 House session and during the 2006 Senate session. We created a variable that captured the proportion of Democrats in each chamber during the time frame of each of the hearings. Subsequently we correlated this measure with the proportion of both science-based and economic-based instrumental arguments. We found no significant correlation between the proportion of Democratic members of Congress and either the proportion of science-based instrumental arguments used in the hearings ($r = -0.067$, $p = 0.828$) or the proportion of economic-based instrumental arguments ($r = 0.277$, $p = 0.360$). This is a noteworthy caveat in thinking about the relationship between use of instrumental arguments and party control. Given this data, we cannot say with confidence whether H3 is rejected or not; rather, it is likely that the relationship between partisanship and instrumentalism is more complex than our data allow us to test.

We turn now to Hypothesis 4, which posits that field hearings will include fewer instrumental arguments than hearings held in Washington, D.C. Of the 12 hearings we analyzed, three were field hearings. Turning first to science-based instrumental arguments, we find that field hearings included a significantly higher proportion of science-based arguments ($\chi^2 = 11.640$, $p < 0.01$, 13.09%) than did hearings held in Washington, D.C. (8.54%) (Table 6). Likewise, a significantly higher proportion of economic-based instrumental arguments were made in field hearings ($\chi^2 = 17.127$, $p < 0.01$, 10.50%) than in Washington, D.C., based hearings (5.75%) (Table 6). Consequently we reject Hypothesis 4.

Table 6. H4 – Proportion of evidence-based arguments by location of Congressional committee hearing.

	Science	Economic
Field	13.09% (86)	10.50% (69)
Washington, D.C.	8.54% (187)	5.75% (126)
Chi-Square Test for Column Differences	$\chi^2 = 11.640$, $p < 0.01$	$\chi^2 = 17.127$, $p < 0.01$

Our fifth hypothesis states that committee hearings in the Senate will feature a greater number of instrumental arguments than those in the House of Representatives. The first column of Table 7 presents the use of science-based instrumental arguments in the committee hearings of the House and Senate. A significantly higher proportion of science-based arguments are made in the House than in the Senate ($\chi^2 = 8.927$, $p < 0.01$). Comparing the use of economic-based, instrumental arguments between the two chambers, shown in the second column of Table 7, reveals the opposite; a significantly higher proportion of economic-based instrumental arguments are made in the Senate compared with the House ($\chi^2 = 18.329$, $p < 0.01$).

Table 7. H5 – Proportion of evidence-based arguments by Congressional chamber.

	Science	Economic
House	10.97% (187)	5.16% (88)
Senate	7.53% (86)	9.37% (107)
Chi-Square Test for Column Differences	$\chi^2 = 8.927$, $p < 0.01$	$\chi^2 = 18.329$, $p < 0.01$

Discussion

The results of our analyses suggest the need for a nuanced understanding of the presence and use of instrumental arguments in Missouri River diversion-related debates. While elected officials may see the value of demonstrating a rational approach to decision making (Esterling 2011), instrumental statements do not dominate the discussion. The use of instrumental arguments falls far short of being pervasive in the Congressional committee hearings we examined.

Argumentation by Members of Congress and Interest Group Competition

Cheon and Urpelainen (2013, 894) point to "competing interest groups" having "potentially complex counteracting effects on policy" formation. We recognize the discursive influence of competing interest groups (Esterling 2004) in how the competition of interest groups concerned with Missouri River diversion projects correlates with how frequently members of Congress made science-based statements within the hearings examined. This seems to confirm Esterling's (2004; 2007; 2009; 2011) suggestion that members of Congress benefit from or at least consider what active interest groups share about their informed understanding of the effects of policies. Yet there was no systematic relationship between interest group competition and economic-based, instrumental arguments made by members of Congress within the hearings we examined. This difference may reflect that many legislators were more confident in making economic-based arguments than science-based arguments.

Comparing the Arguments of Members of Congress and Representatives of Interest Groups

Jones et al. (1993) observe that witnesses in Congressional committee hearings tend to be selected because they share the views of members of the Congressional leadership. Yet we find representatives of interest groups presenting testimony do not employ the same instrumental arguments as the members of Congress with whom they are aligned. While members of Congress may benefit from the active interest groups' expertise on the effects of policies (Esterling 2004; 2007; 2009; 2011), by no means are the arguments they make identical to the arguments of interest groups with whom they share an overall position. With one noteworthy exception, elected officials used more instrumental arguments, science-based and economic-based, in supporting their positions than did interest group representatives who shared their views. The one remarkable exception was among those opposed to diversions; interest group representatives used science-based arguments three times more often as a percentage of total arguments than did members of Congress who shared their views. Caldwell (1984, 852) notes that much of the "active public" opposed to water diversions from Garrison Dam and the reservoir behind it was from people living outside North Dakota who weighed the scientific evidence more heavily than legal rights or historical promises.

Caldwell (1984) considers that the leading consideration in the downsizing of Garrison Diversion projects was economics followed by scientific analysis. While science may be at the core of what makes up evidence (Caldwell 1984; Michaels et al. 2014), in our study science-based arguments form a low percentage of all arguments made. For proponents of the projects, science-based arguments made up a lower percentage of arguments vis-à-vis economic-based arguments, while the opposite was true for opponents. Much to the frustration of scientists, science is one consideration among many that weighs into decision makers' deliberations (Michaels and Tyre 2012). Nonetheless, economic arguments can prove persuasive in debates related to the environment.

Fisher et al. (2013a), in their study of national climate change politics, found that the economic implications of regulating greenhouse gases provided the basis for the largest share of consensus among actors in the 109th and 110th Congress.

Instrumental Arguments and Majority Political Party

Park et al. (2010) investigated how different issues are framed in political debates using Congressional hearings on climate change. They noted that Democrat-controlled Congressional sessions were more likely to feature hearings on climate change than Republican controlled sessions. In hearings during Democrat-controlled sessions, pro-environmental political actors and mainstream scientists tended to provide testimony. In contrast, representatives from business and industry were featured when Republicans were in the majority and their testimonies challenged climate change science and highlighted the adverse effects on economic growth and international relations from policies addressing climate change. In our study, while both science-based and economic-based statements were more prevalent in Republican than Democratic controlled committee hearings, there may be other viable explanations in addition to which party was in the majority.

One of the most striking indicators of the pervasiveness of making instrumental arguments comes from our evidence that, as a percentage, more instrumental arguments were made in hearings held outside of Washington than those held on Capitol Hill. When playing to the hometown crowd, as field hearings make possible (Fenno 1978; Bianco 1994; Esterling 2007; Heitshusen 2007), elected officials apparently found value in presenting instrumental arguments.

Future Research Directions

Our work is part of a recent and growing trend that goes beyond tracking the number of U.S. Congressional hearings on a particular topic to measuring both the quantity and content for topics of interest. For example, Mueller et al. (2004) tracked how many hearings there were per year on communication-information policy. The strategy of exploiting the digitization of Congressional committee hearings using computer assisted content analysis has been particularly attractive to researchers investigating policy change in regard to such contemporary concerns as internet neutrality (Cheng et al. 2012), biofuels (Holleman 2012), and climate change (McCright and Dunlap 2003; Park et al. 2010; Fisher et al. 2013a; Fisher et al. 2013b). In contrast, we have demonstrated the utility of this approach for more than six decades of what was originally framed as a solution to the forthcoming employment needs of returning World War II soldiers. There is further scope in future research to use time-series analysis to examine whether, and if so, how, the use of instrumental arguments in legislative deliberations has changed over time.

The nature of our data set made it possible for us to investigate a little-explored dimension of Congressional committee hearings -- a comparison between the arguments put forth in field committee hearings and those presented at hearings in Washington, D.C. A beneficial next step would be to explore the dynamics between policy change and the framing of argumentative strategies employed in Congressional committee deliberations across policy issues with different Congressional 'life spans' and venues for hearings. Additionally, it would be worthwhile to test the relationship between the academic and professional backgrounds of the legislators and their level of confidence in making science-based arguments compared with economic-based arguments. Equally valuable would be to test the hypothesis that those outside a region impacted or potentially impacted by federal decision making are more willing to form opinions based on instrumental evidence than those impacted or potentially impacted.

Another promising research direction is to probe more deeply than was done in this paper into the dynamic between the arguments presented by members of Congress and the arguments of allied interest groups and competing agencies behaving much like competing interest groups. Revisiting how the 1944 Flood Control Act came about in light of the ways Congress viewed and processed the information members of Congress received from the Bureau of Reclamation and the U.S. Army Corps of Engineers would be revealing, given the sometimes competing agendas of the agencies.

Employing the advocacy coalition framework to do so, with its emphasis on considering how coalition actors behave to influence policy processes directly or indirectly and on learning within coalitions, may prove instructive (Jenkins-Smith et al. 2014).

A valuable parallel to this investigation would be an associated media content analysis to investigate whether the weighting of instrumental arguments (scientific or economic) is transferred from Congressional hearings to those working in the media. While Congressional hearings are often organized spectacles (Jones et al. 1993), little research has examined the interplay between information presented in hearings, particularly instrumental information, and uptake of that information by the mass media. Thus we believe that a fruitful next step in research would be to examine fully how mass media engages with information presented in hearings.

Conclusion

Instrumental arguments played a vital role in the rational discourse that occurred over the deliberations in Congressional committee hearings about diverting water from the Missouri River. This was true whether the arguments were made on Capitol Hill or in field hearings, where a key aspect of the proceedings was that they took place on elected officials' home turf with an audience of constituents.

Further, our results reveal there are different dynamics at work between members of Congress and interest groups when it comes to science-based arguments compared with economic-based arguments. We demonstrate the lack of congruence between the choice of argumentation by elected officials and by members of interest groups and the inconclusive relationship between choice of instrumental arguments and which political party is in the majority. While more economic-based instrumental arguments were made in the Senate than in the House of Representatives, more science-based instrumental arguments were made in House related committee hearings than in Senate related committee hearings. Instrumental arguments did not constitute the majority of arguments made in the Congressional committee hearings examined. Nonetheless, our findings suggest that considering who makes the arguments and under what circumstances can provide useful insights into the dynamics of deliberative policy making. Understanding the use of instrumental arguments, notably science-based and economic-based ones, is particularly important in water policy where considerable analysis, including in environmental sciences and economics, is undertaken to underpin deliberative policy making.

Acknowledgements

The research for this paper was made possible with the support of The Canadian Studies Research Grant Program, Canadian Embassy, Washington, D.C., the University of Nebraska Foundation Fund for Research on the U.S. Congress, Lincoln, Nebraska and the University of Nebraska Foundation Senning Fellowship, Lincoln, Nebraska. Thanks go to Grace York, University of Michigan for making key documents available, Katherine Hunt, Austin Peay State University for providing editorial assistance, and two anonymous reviewers for their comments on an earlier draft.

Author Bio and Contact Information

Sarah Michaels is a Professor in the Department of Political Science and a Faculty Fellow at the Nebraska Public Policy Center, University of Nebraska-Lincoln. She can be contacted at michaels2@unl.edu and at P.O. Box 880328, Lincoln, NE 68588-0328.

Mike Gruszczynski (corresponding author) is an Associate Professor of Political Science at Austin Peay State University. He can be reached at gruszczynskim@apsu.edu and at 601 College St, Clement 143F, Clarksville, TN 37044.

References

Appollonio, D.E. and L.A. Bero. 2009. Evidence and argument in policymaking: Development of workplace smoking legislation. *BMC Public Health* 9: 189.

Arnold, D.R. 1990. *The Logic of Congressional Action*. New Haven, CT, Yale University Press.

Bartlett, R.V. 1988. Adapt or get out: The Garrison Diversion Project and controversy. *Environmental Review* 12(3): 57-74.

Baumgartner, F.R. and B.D. Jones. 1993. *Agendas and Instability in American Politics*. First Edition. Chicago, University of Chicago Press.

Bianco, W.T. 1994. *Trust: Representatives and constituents*. Ann Arbor, MI, University of Michigan Press.

Billington, D.P. and D.C. Jackson. 2006. *Big Dams of the New Deal Era: A Confluence of Engineering and Politics*. Norman, OK, University of Oklahoma Press.

Brewer, G.D. 1973. *Politicians, Bureaucrats, and the Consultants*. New York, Basic Books.

Bullock, H., J. Mountford, and R. Stanley. 2001. *Better Policy-making*. London, Cabinet Office, Centre for Management and Policy Studies.

Burstein, P.R. and E.C. Hirsh. 2007. Interest organizations, information, and policy innovation in the U.S. Congress. *Sociological Forum* 22(2): 174-199.

Cabinet Office. 1999. *Professional Policy-Making for the Twenty-first Century*. Strategic Policy Making Team, London, Cabinet Office.

Caldwell, L.K. 1984. Garrison diversion: Constraints on conflict resolution. *Natural Resources Journal* 24: 839-863.

Caplan, N.S. 1977. A minimal set of conditions necessary for the utilization of social science knowledge in policy formulation at the national level. In: *Using Social Science Research in Public Policy Making,* Carol H. Weiss (Ed.). Lexington, MA, D.C. Heath, 183-198.

Cheng, A.S., K.R. Fleischmann, P. Wang, E. Ishita, and D.W. Oard. 2012. The role of innovation and wealth in the net neutrality debate: A content analysis of human values in congressional and FCC hearings. *Journal of the American Society for Information Science and Technology* 63(7): 1360-1373.

Cheon, A. and J. Urpelainen. 2013. How do competing interest groups influence environmental policy? The case of renewable electricity in industrialized democracies 1989–2007. *Political Studies* 61(4): 874-897.

Czech, B. and P.R. Krausman. 2001. *The Endangered Species Act*. Baltimore, Johns Hopkins University Press.

Dahl, R.A. 1961. *Who Governs?* New Haven, CT, Yale University Press.

Demeritt, D. 2001. The construction of global warming and the politics of science. *Annals of the Association of American Geographers* 91(2): 307-337.

Diermeier, D. and T.J. Feddersen. 2000. Information and congressional hearings. *American Journal of Political Science* 44(1): 51-65.

Edelman, M.J. 1964. *The Symbolic Uses of Politics*. Urbana, IL, University of Illinois Press.

Edelman, M.J. 1971. *Politics as Symbolic Action: Mass Arousal and Quiescence*. New York, Academic Press.

Esterling, K.M. 2004. *The Political Economy of Expertise: Information and Efficiency in American National Politics*. Ann Arbor, MI, University of Michigan Press.

Esterling, K.M. 2007. Buying expertise: Campaign contributions and attention to policy analysis in congressional committees. *American Political Science Review* 101(1): 93-109.

Esterling, K.M. 2009. Does the federal government learn from the states? Medicaid and the limits of expertise in the intergovernmental lobby. *Publius* 39(1): 1-21.

Esterling, K.M. 2011. Deliberative disagreement in U.S. health policy committee hearings. *Legislative Studies Quarterly* 36(2): 169-198.

Fenno, R.F., Jr. 1978. *Home Style: House Members in their Districts*. NP, HarperCollins.

Ferejohn, J.A. 1974. *Pork Barrel Politics: Rivers and Harbors Legislation 1947-1968*. Stanford, CA, Stanford University Press.

Fisher, D.R., P. Leifeld, and Y. Iwaki. 2013a. Mapping the ideological networks of American climate politics. *Climatic Change* 116(3-4): 523-545.

Fisher, D.R., J. Waggle, and P. Leifeld. 2013b. Where does political polarization come from? Locating polarization within the U.S. climate change debate. *American Behavioral Scientist* 57(1): 70-92.

Garrison Diversion Conservancy District. 2014. History and federal legislation. Available at http://www.garrisondiv.org/about_us/history_federal_legislation/. Accessed June 29, 2016.

Grier, P. 1983. Garrison diversion: Needed, or a clear case of 'pork'? *Christian Science Monitor* May 4, 1983. Available at http://www.csmonitor.com/1983/0504/050436.html. Accessed June 29, 2016.

Gruszczynski, M.W. and S. Michaels. 2012. The evolution of elite framing following enactment of legislation. *Policy Sciences* 45(4): 359-384.

Harris, D.B. 2010. Partisan framing in legislative debates. In: *Winning with Words: The Origins and Impact of Political Framing,* B.F. Schaffner and P.J. Sellers (Eds.). New York, Routledge, pp. 41-59.

Hearne, R.R. 2007. Evolving water management institutions in the Red River Basin. *Environmental Management* 40(6): 842-852.

Heitshusen, V. 2007. Field hearings: Fact sheet on purposes, rules, regulations, and guidelines. Washington, D.C., Congressional Research Service.

Holleman, H. 2012. Energy policy and environmental possibilities: Biofuels and key protagonists of ecological change. *Rural Sociology* 77(2): 280-307.

Huitt, R.K. 1954. The congressional committee: A case study. *The American Political Science Review* 48(2): 340-365.

Jasanoff, S.1997. Civilization and madness: The great BSE scare of 1996. *Public Understanding of Science* 6(3): 221-232.

Jenkins-Smith, H.C., D. Nohrstedt, C.M. Weible, and P.A. Sabatier. 2014. Advocacy coalition framework: Foundations, evolution, and ongoing research. In: *Theories of the Policy Process,* P.A. Sabatier and C. Weible (Eds.). Third Edition. Boulder, CO, Westview Press, pp. 183-224.

Jones, B.D. and F.R. Baumgartner. 2004. Representation and agenda setting. *Policy Studies Journal* 32(1): 1-24.

Jones, B.D., F.R. Baumgartner, and J.C. Talbert. 1993. Destruction of issue monopolies in congress. *American Political Science Review* 87(3): 673-687.

Keller, A.C. 2009. *Science in Environmental Policy: The Politics of Objective Advice*. Cambridge, MA, MIT Press.

King, D.C. 1993. *Public Goods, Private Interests, and Participation in Congressional Committee Hearings*. Cambridge, MA, Harvard University Press.

King, B.G., K.G. Bentele, and S.A. Soule. 2007. Explaining fluctuation in congressional attention to rights issues 1960-1986. *Social Forces* 86(1): 137-163.

Konig, A. and S. Jasanoff. 2002. The credibility of expert advice for regulatory decision-making in the U.S. and EU. Regulatory Policy Program Working Paper RPP-2002-07. Cambridge, MA, Center for Business and Government, John F. Kennedy School of Government, Harvard University.

Kraft, M.E. and N.J. Vig. 2010. Environmental policy over four decades: Achievements and new directions. In: *Environmental Policy: New Directions for the Twenty-First Century,* N.J. Vig and M.E. Kraft (Eds.). Washington, D.C., CQ Press, pp. 1-26.

Krehbiel, K. 1991. *Information and legislative organization*. Ann Arbor, University of Michigan Press.

Krippendorff, K.A. 2003. *Content Analysis: An Introduction to its Methodology*. Newbury Park, CA, Sage.

Lasswell, H.D. and N. Leites. 1949. *Language of Politics: Studies in Quantitative Semantics.* Cambridge, MA, MIT Press.

Litfin, K.T. 2000. Advocacy coalitions along the domestic-foreign frontier: Globalization and Canadian climate change policy. *The Policy Studies Journal* 28(1): 236-252.

Liu, X., E. Lindquist, and A. Vedlitz. 2011. Explaining media and congressional attention to global climate change 1969-2005: An empirical test of agenda-setting theory. *Political Research Quarterly* 64(2): 405-419.

Maguire, L. 2013. The U.S. congress and the politics of Afghanistan: An analysis of the senate foreign relations and armed services committees during George W. Bush's second term. *Cambridge Review of International Affairs* 26(2): 430-452.

Mayhew, D.R. 1974. *Congress: The Electoral Connection*. New Haven, Yale University Press.

McCright, A.M. and R.E. Dunlap. 2003. Defeating Kyoto: The conservative movement's impact on U.S. climate change policy. *Social Problems* 50(3): 348-373.

Michaels, S. and A.J. Tyre. 2012. How indeterminism shapes ecologists' contributions to managing socio-ecological systems. *Conservation Letters* 5(4): 289-295.

Michaels, S., J. Holmes, and L. Shaxson. 2014. Science communication and the tension between evidence-based and inclusive features of policy making. In: *New Trends in Earth Science Outreach and Engagement: The Nature of Communication,* J.L. Drake, , Y.Y. Kontar, and G.S. Rife (Eds.). Advances in Natural and Technological Hazards Research, Springer, New York, N.Y., pp. 38: 83-92.

Montpetit, É. 2011. Scientific credibility, disagreement, and error costs in 17 biotechnology policy subsystems. *Policy Studies Journal* 39(3): 513-533.

Mucciaroni, G. and P.J. Quirk. 2006. *Deliberative Choices*. Chicago, University of Chicago Press.

Mueller, M., C. Page, and B. Kuerbis. 2004. Civil society and the shaping of communication-information policy: Four decades of advocacy. *Information Society* 20(3): 169-185.

Newton, L. 2008. *Illegal, Alien, or Immigrant: The Politics of Immigration Reform.* New York, New York University Press.

Nicholson-Crotty, S. 2005. Bureaucratic competition in the policy process. *Policy Studies Journal* 33(3): 341-361.

Oleszek, W.J. 2001. *Congressional Procedures and the Policy Process.* Fifth edition. Washington, D.C., Congressional Quarterly.

Oleszek, W.J. 2011. *Congressional Procedures and the Policy Process.* Eighth edition. Washington, D.C., Congressional Quarterly.

Olzak, S. and S.A. Soule. 2009. Cross-cutting influences of environmental protest and legislation. *Social Forces* 88(1): 201-225.

Park, H.S., X. Liu, and A. Vedlitz. 2010. Framing Climate Policy Debates: Science, Network, and US Congress 1976–2007. Conference proceedings of the Policy Networks Conference 2010 Paper 41. Available at http://opensiuc.lib.siu.edu/pnconfs_2010/41. Accessed June 29, 2016.

Pisani, D.J. 2002. *Water and American Government: The Reclamation Bureau, National Water Policy, and the West 1902-1935*. Berkeley, University of California Press.

Sabatier, P.A. 1975. Social movements and regulatory agencies: Toward a more adequate—and less pessimistic—Theory of clientele capture. *Policy Sciences* 6(3): 301-342.

Schneier, E.V. and B.M. Gross. 1993. *Legislative Strategy: Shaping Public Policy*. New York, St. Martin's Press.

Seligman, D. 2008. *World's Major Rivers: An Introduction to International Water Law with Case Studies*. Las Vegas, Colorado River Commission of Nevada. Available at http://crc.nv.gov/docs/Worlds_Major_Rivers.pdf. Accessed June 29, 2016.

Shepsle, K.A. and B.R. Weingast. 1981. Political preferences for the pork barrel: A generalization. *American Journal of Political Science* 25(1): 96-111.

Smith, B.L.R. 1992. *The Advisors: Scientists in the Policy Process*. Washington, D.C., Brookings.

Stern, M.J., A. Predmore, M.J. Mortimer, and D.N. Seesholtz. 2010. From the office to the field: Areas of tension and consensus in the implementation of the National Environmental Policy Act within the U.S. Forest Service. *Journal of Environmental Management* 91(6): 1350-1356.

Talbert, J.C., B.D. Jones, and F.R. Baumgartner. 1995. Nonlegislative hearings and policy change in congress. *American Journal of Political Science* 39(2): 383-405.

Truman, D.B. 1951. *The Governmental Process: Political Interests and Public Opinion.* New York, Alfred A. Knopf.

Tzoumis, K. 2001. *Environmental Policymaking in Congress: The Role of Issue Definitions in Wetlands, Great Lakes, and Wildlife Policies*. New York, Routledge.

United States. 1944a. *Flood Control*. Hearings before a Subcommittee of the Committee on Commerce, United States Senate, Seventy-eighth Congress, second session on H.R. 4485 May 29-June 15, 1944. Washington, U.S. Government Printing Office.

United States. 1944b. *Flood Control Plans and New Projects*. 1943 and 1944 Hearings before the Committee on Flood Control, House of Representatives, Seventy-eighth Congress, first[-second] session on H.R. 4485, a bill authorizing the construction of certain public works on rivers and harbors for flood control, and for other purposes. Washington, U.S. G.P.O.

United States. 1957. *Garrison Diversion Unit, Missouri River Basin Project*. Hearings before a Special Subcommittee on Irrigation and Reclamation, of the Committee on Interior and Insular Affairs, House of Representatives Eighty-fifth Congress, first session, October 30, 1957, Devils Lake, North Dakota. Washington, U.S. Government Printing Office.

United States. 1960. *Garrison Diversion Unit, Missouri River Basin Project*. Hearings before the Subcommittee on Irrigation and Reclamation, of the Committee on Interior and Insular Affairs, House of Representatives Eighty-sixth Congress, second session, June 10, 1960, Washington, D.C. Washington, U.S. Government Printing Office.

United States. 1961. *Garrison Diversion Unit*. Hearings before the Subcommittee on Irrigation and Reclamation, of the Committee on Interior and Insular Affairs, Senate, Eighty-seventh Congress, first session, June 21, 1961, Washington, D.C. Washington, U.S. Government Printing Office.

United States. 1963. *Garrison Diversion Unit, Missouri River Basin Project*. Hearings before the Subcommittee on Irrigation and Reclamation, of the Committee on Interior and Insular Affairs, Senate, Eighty-eighth Congress, first session, June 6, 1963, Washington, D.C. Washington, U.S. Government Printing Office.

United States. 1964. *Garrison Diversion Unit, Missouri River Basin Project*. Hearings before the Subcommittee on Irrigation and Reclamation,

of the Committee on Interior and Insular Affairs, House of Representatives, Eighty-eighth Congress, second session, February 20, 1964, Washington, D.C. Washington, U.S. Government Printing Office.

United States. 1965. *Garrison Diversion Unit of the Missouri River Basin Project*. Hearings before the Subcommittee on Irrigation and Reclamation, of the Committee on Interior and Insular Affairs, Senate, Eighty-ninth Congress, first session, March 29, 1965, Washington, D.C. Washington, U.S. Government Printing Office.

United States. 1975. *Garrison Diversion Unit Irrigation Project: Prospects and Problems*. Hearings before a Subcommittee of the Committee on Government Operations, House of Representatives, Ninety-fourth Congress, first session, September 15, [November 19] 1975. Washington, U.S. Government Printing Office.

United States. 1998. *Garrison Unit Reformulation*. Oversight Hearing before the Subcommittee on Water and Power of the Committee on Resources, House of Representatives, One Hundred Fifth Congress, second session, September 29, 1998, Washington, D.C., Washington, U.S. Government Printing Office.

United States. 2006. *Red River Valley Water Supply Project*. Hearings before a Subcommittee on Appropriations, Senate, One Hundred Ninth Congress, second session, August 24, 2006, Fargo, North Dakota. Washington, U.S. Government Printing Office.

Webber, D.J. 1984. Political conditions motivating legislators' use of policy information. *Review of Policy Research* 4(1): 110-118.

Weible, C.M. 2008. Expert-based information and policy subsystems: A review and synthesis. *Policy Studies Journal* 36(4): 615-635.

Weingast, B.R., K.A. Shepsle, and C. Johnsen. 1981. The political economy of benefits and costs: A neoclassical approach to distributive politics. *Journal of Political Economy* 89(4): 642-664.

Weiss, C.H. 1989. Congressional committees as users of analysis. *Journal of Policy Analysis and Management* 8(3): 411-431.

Universities Council on Water Resources
Journal of Contemporary Water Research & Education
Issue 158, Pages 148-171, August 2016

Dynamic Assessment of Current Management in an Intensively Managed Agroecosystem

Christopher G. Wilson[1], Kenneth M. Wacha[2], *A.N. (Thanos) Papanicolaou[1,3], Heather A. Sander[4], Violet B. Freudenberg[1], Benjamin K.B. Abban[1], and Chang Zhao[4]

*[1]Hydraulics and Sedimentation Lab, Department of Civil & Environmental Engineering, University of Tennessee, Knoxville, TN, [2]IIHR, Hydroscience & Engineering, Department of Civil & Environmental Engineering, University of Iowa, Iowa City, IA, [3]Tennessee Water Resources Center, Knoxville, TN, [4]Department of Geographical and Sustainability Sciences, University of Iowa, Iowa City, IA, *Corresponding Author*

Abstract: To assess management impacts on the functionality of intensively managed agroecosystems, a modeling framework was developed with a bottom-up approach and spatially distributed, process-based models. The framework is equipped with dynamic, data-informed indicators and indices to illuminate the factors influencing sustainability. The proposed dynamic indices consider natural and human aspects of an agroecosystem such as erosion, biogeochemistry, and economics. Most current indicators are static, or slow-changing, soil characterization parameters that reflect better long-term interactions between landscape features, climate, and biology. However, the ever-changing land management and climate necessitates the use of dynamic parameters that reflect agroecosystem responses to different land management on similar timescales (e.g., seasonally). Our framework examines the performance of different ecosystem services including crop productivity, carbon storage, and net income under three different strategies with varying degrees of tillage intensity. The strategy with the highest intensity produced the highest yields, but also had the highest production costs. The second most intense strategy also had high yields, as well as the highest net income. However, these two strategies produced high erosion rates, which depleted the recalcitrant soil carbon, a critical component of system productivity and health. The index that provided the clearest picture of improvement within an agroecosystem was the Carbon Management Index (*CMI*), which incorporates carbon lability and the implicit accounting of soil carbon redistribution from erosion. The *CMI* for the more intense strategies decreased in recent years, showing that they are not sustainable despite their high short-term productivity or profitability. The least intense strategy had the lowest Soil Organic Carbon (SOC) depletion through erosion and the highest *CMI* with a trend that is still-increasing through the present. Our study shows that to augment SOC storage, it important not only to increase the overall organic matter input, but also increase the amount of recalcitrant carbon in the soil and the longevity of all soil carbon through aggregate formation.

Keywords: *bottom-up modeling framework, management practices, erosion, soil organic carbon, Carbon Management Index, Clear Creek, IA, ecosystem services*

As the Earth's population approaches nine billion people in the upcoming decades, food production must increase by nearly 70% to satisfy this global need (Reytar et al. 2014). To quell global food demands, productivity has increased in recent years through agrotechnological advances (e.g., Fisher et al. 2014) and crop breeding (Klumper and Qaim 2014), but the ramp-up of production and associated intensive management are degrading agroecosystems (Lal 2004), thereby threatening their sustainability (Karlen et al. 2014).

Ecosystem degradation due to intensive agriculture is seen vividly in Iowa, which has become a world-leading, commodity-based agroecosystem due to its overall fertile soils. However, the enhanced erosion triggered by accelerating production levels, in conjunction with more frequent shifts between high intensity rainfall and deep droughts, is so extreme that two-thirds of the farmland in Iowa is losing soil at a rate over 11 Mt/ha/yr, which is equivalent to a little over a dump truck load per hectare per year (Tevis

2015). During a week of intense storms in 2014, Iowa farmers lost an estimated $1 billion in yield due to erosion (Eller 2014).

Additionally, intensive management has depleted the soils of essential nutrients (Pimentel et al. 1995) making Iowa a "nutrient-hungry system." To maintain high production, artificial fertilizers are broadly applied in mass quantities, but nearly half of these applied fertilizers are lost through leaching, runoff, and erosion (Burkart et al. 2005). As a result, recent record nitrate levels in the Raccoon and Des Moines Rivers in Iowa, predominantly from agricultural runoff, are taxing the aging infrastructure of the Des Moines Water Works and have even prompted lawsuits against three upstream, agricultural counties (Eller 2015). Further downstream, the problems persist as nutrient fluxes from the Upper Mississippi River Basin have been linked to hypoxia in the Gulf of Mexico (Porter et al. 2015).

The widespread ramifications of this ecosystem degradation have pushed the development of adaptive, watershed-scale management plans for improving the sustainability of agroecosystems like those in Iowa. But, do we have the necessary tools to assess accurately at the watershed scale the performance of the different practices currently in use or possible alternatives (e.g., Robertson et al. 2014)? As a substitute to extensive field monitoring studies, computer models can work as an indirect form of adaptive management (Prato 2003) to evaluate watershed-scale responses to different combinations of current and projected practices.

One factor to consider in developing these assessment modeling tools is that intensive land management and the resulting soil redistribution can augment the natural heterogeneity observed in agroecosystems along a hillslope for properties like Soil Organic Carbon (SOC) and other nutrients by dictating flowpath development and soil erodibility (e.g., Wilson et al. 2009; Dlugoβ et al. 2010; Stavi and Lal 2011; Papanicolaou et al. 2015b). Hence, the selective entrainment and resulting fluxes of these constituents may differ significantly in areas where erosion dominates relative to areas where deposition occurs (Van Oost et al. 2006; Wang et al. 2015), resulting in the increased variability across the landscape.

Since most biogeochemical models (e.g., ROTH-C, DNDC, CENTURY) used to quantify changes in SOC or other nutrients focus within a soil column, it is unlikely that they will capture this augmented, field-level heterogeneity. As a result, existing models will either overestimate or underestimate stock predictions depending on the location within a field, since they do not consider selective mobilization (e.g., Parton et al. 1987; Paustian et al. 1992; Harden et al. 1999; Manies et al. 2001).

The majority of studies that have attempted field-scale assessments of SOC linked these existing biogeochemical models with lumped erosion models, such as those based on the Universal Soil Loss Equation, USLE (e.g., Monreal et al. 1997; Zhang et al. 2014), to account for redistribution along the downslope. These erosion models provide gross, annual average estimates but do not capture the seasonal variability in concentrations and fluxes (Harden et al. 1999) or account for deposition and hence in-field storage (Gregorich et al. 1998; Van Oost et al. 2006).

A handful of recent, but significant modeling efforts have accounted for the collective dynamics of erosion and deposition on SOC redistribution and storage (e.g., Billings et al. 2010; Dlugoβ et al. 2012). Fewer studies have incorporated the effects of selective entrainment and deposition of the finer size fraction of organic rich soils by the flow (Papanicolaou et al. 2015b). However, these modeling advances have not yet transferred from the field-scale to the watershed-scale.

In general, integrative assessments at the watershed scale typically use Earth system models that follow a top-down approach, working to identify emergent patterns of SOC. These integrative assessments still use lumped erosion models, along with coarse field erosion estimates and decomposition rates from regression equations with climate variables to determine Net Ecosystem Exchange (e.g., Brenner et al. 2001; Ahmadov et al. 2007; Yadav and Malanson 2008). These estimates can potentially lead to inaccurate estimates for ecosystem services related to SOC, with direct implications on the design of efficient economic policies for providing incentive payments or credits to potential producers (Antle and Mooney 2002; Babcock 2010). Thus, these assessments are not truly dynamic in that they provide only gross averaged estimates of erosion,

SOC formation, and decomposition on an annual basis (e.g., Polasky et al. 2008).

To reflect better the impacts of agricultural management, dynamic indices are needed that consider the spatial and temporal variability of key biogeochemical properties and processes in agricultural fields and pertain to ecosystem services, like food security, nutrient cycling, and climate regulation. Dynamic indices are better suited to assess functionality and sustainability of an agroecosystem than the static, or slow-changing, soil characterization parameters that more accurately reflect long-term interactions between landscape features, climate, and biology (Doran and Parkin 1996; Lehman et al. 2015), as they reflect the response of the agroecosystem to different land management on relevant time scales. Dynamic indices, which combine different parameters or different aspects of similar parameters, can capture this complexity, meaning they are able to change on similar time scales as the dominant drivers of climate and management. Thus, they provide a more complete accounting of sustainability (Hester and Little 2013).

An extensive literature review was performed to identify different indices currently used to assess sustainability (e.g., Costanza et al. 1997; de Groot et al. 2002; Millennium Ecosystem Assessment 2005). The selection was based on certain criteria (Dale et al. 2013). The indices had to be (1) practical, easy to measure, and inexpensive; (2) responsive to different drivers from both nature (e.g., climate) and humans (e.g., land management); and (3) responsive in a known and predictable manner, to facilitate better understanding of the associated changes in the measurements. In the end, most of these indices related to SOC.

SOC is closely linked to different services of an agroecosystem like food security, greenhouse gas regulation, and climate regulation, because it is inherently tied to several soil functions, such as nutrient cycling and habitat productivity. Soil carbon is the main source of energy for soil microorganisms, which drive decomposition and transformation of soil nutrients into a form that is readily available for plant uptake. More available nutrients leads to higher crop yields for food security and reduced fertilizer needs, which lowers production costs. Additionally, SOC can regulate the ability of the soil to hold water, especially in times of drought (e.g., Vereecken et al. 1990; Righetti and Lucarelli 2007; Papanicolaou et al. 2015a). The organic matter also acts as a "glue" holding the soil particles together and preventing erosion (e.g., Tisdall and Oades 1982; Papanicolaou et al. 2015b). Lastly, soils can be a storage reservoir for atmospheric CO_2 (e.g., West et al. 2010), helping in climate regulation.

In this study, the central objective was to develop a bottom-up, yet practical modeling framework to assess the performance of current management strategies (e.g., different combinations of crops and tillage intensities) in an intensively managed agroecosystem of the U.S. Midwest for sustainability. Herein, we define sustainability as the agroecosystem's ability to lessen the negative effects of erosion and SOC loss on the environment while remaining productive and profitable in the long-term. In developing this framework, we sought to overcome the limitations of Earth system models used for watershed assessments. Our modeling framework integrates spatially distributed, process-based erosion and biogeochemical models with an economic assessment. The framework was used to provide indices that reflect the dynamic response of the landscape to the various management strategies and distinguish the differences between short-term productivity/profitability and long-term sustainability resulting from the different strategies.

With this framework, we can help address the following questions: (1) are current management strategies geared towards high production ultimately degrading our agroecosystems beyond the point of functioning effectively, even with added supplements?; and (2) how can we better govern these systems so that they will still be productive in 50 years? Therefore, in order to ensure long-term, high-yield production and establish a sense of food security in light of volatile market demands and climate, we must develop more effective, data-informed, watershed management and governance plans for our agroecosystems that are also sustainable. A major contribution of this study is the development of a watershed-scale, modeling framework that captures the cause-and-effect relationship between land management and soil health to cultivate better management/governance plans.

Study Site

We tested the developed framework in the Clear Creek, IA watershed (Figure 1), which is part of the National Science Foundation Intensively Managed Landscapes - Critical Zone Observatory (IML-CZO; http://criticalzone.org/iml). The IML-CZO has available hydrologic, erosion, biogeochemical, and economic databases for educating the modeling framework. Additional data from remote sensing sources were also used to look at information from the watershed scale and develop carbon budgets for Clear Creek that incorporate the erosion effects of different land management practices, namely tillage.

The Clear Creek watershed is a 270-km^2, mixed agroecosystem in east-central Iowa (Figure 1). Clear Creek is representative of Iowa and most of the Midwest in terms of land use (predominantly agricultural with developing small pockets of urban areas), soil (Alfisols and Mollisols), and climate (humid-continental).

Land Use

Over 80% of the watershed has been converted from prairie and forest to row-crop agriculture and pasture, a pattern that typifies the American Midwest (Rayburn and Schulte 2009). Since 1991, the dominant management strategies in the watershed are 2-year, corn-soybean rotations that use in parts conservation tillage practices (reduced and no-till). These strategies are identified as the following: no till corn – fall till bean (NTC-FTB); spring till corn – no till bean (STC-NTB); and fall till corn – no till bean (FTC-NTB). A detailed description of the associated practices and equipment is in Abaci and Papanicolaou (2009).

For the purposes of this study, it is important to note that fall tillage is typically in October with a chisel plow and is deeper than the reduced tillage of spring, performed with a field cultivator in April (Reicosky 2015). Table 1 lists the areal extents for each strategy, as well as their relative intensities, based on the Soil Disturbance Rating (SDR). The SDR considers the impact of the agricultural equipment used in a particular management strategy through inversion, mixing, lifting, shattering, aeration, and compaction of the soil. Higher values reflect a greater disturbance to the soil. The order of the different strategies in terms of decreasing intensity is as follows: NTC-FTB (53) > FTC-NTB (41) > STC-NTB (32).

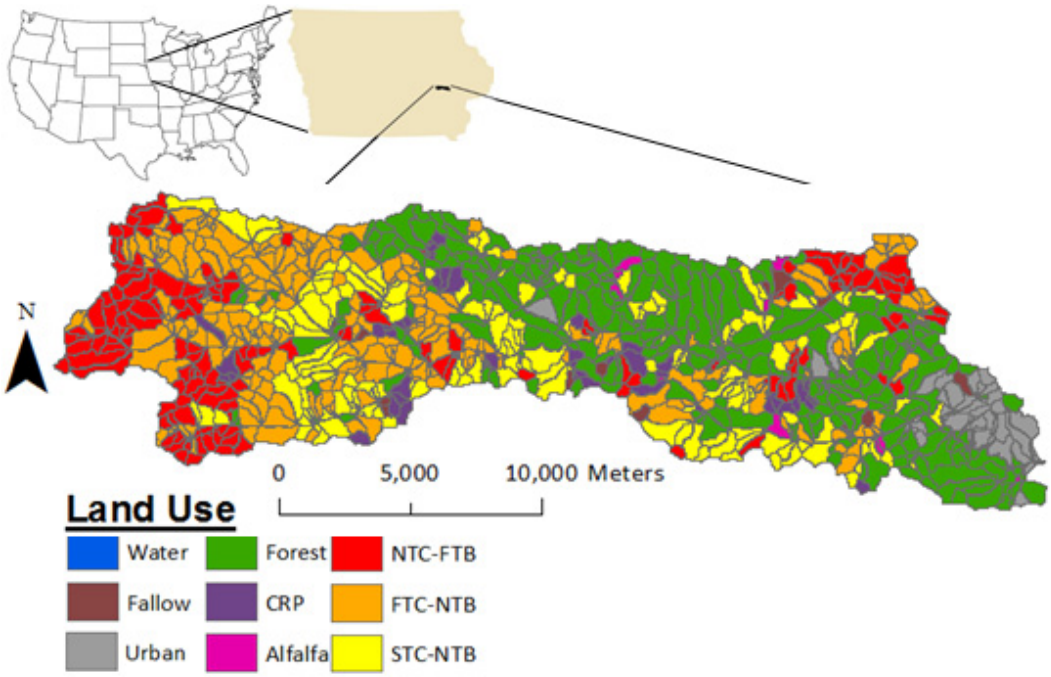


Figure 1. The Clear Creek, IA watershed and a reclassified land management map for Clear Creek. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Table 1. Dominant corn-bean rotations in Clear Creek, IA.

Rotation	Areal Coverage (%)	Soil Disturbance Rating	Tillage Intensity
NTC-FTB	26	53	High
FTC-NTB	40	41	Med
STC-NTB	34	32	Low

See Abaci and Papanicolaou 2009 for areal coverage and details on management strategies.
NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Soils

The watershed is situated in the Southern Iowa Drift Plain, which is comprised of glacial deposits broken up by rolling hills (slopes in Clear Creek range up to 12%) with many small creeks and streams. Windblown Peorian loess deposits cover the area to depths ranging from 1.5 to 9 m (Ruhe 1969). The loess-derived soils are very productive agricultural lands due to their ability to hold water and nutrients (Jones et al. 1967), but being loess-derived, the soils are relatively homogeneous, consisting predominantly of silt-sized particles, and are highly erodible under intensive land management, namely tillage (Bettis et al. 2003).

Mollisols and Alfisols are the dominant soil orders found in the watershed. The most common upland soil associations are the Tama-Downs and Fayette-Downs associations (Dideriksen et al. 2007), which are both well drained. The dominant lowland association, which forms on either stream terraces or flood plains, is the Colo-Nevin-Nodaway association with drainage classes ranging from poorly drained to moderately well drained (Highland and Dideriksen 1967). Moving downstream, the dominant soil texture changes from silty clay loam in the headwaters to a silty loam near the mouth of Clear Creek.

Climate

Due to the mid-continental location of Iowa, the climate in Clear Creek is characterized by hot summers, cold winters, and wet springs (Highland and Dideriksen 1967). Average annual precipitation is approximately 889 mm/yr with convective thunderstorms prominent in the summer and snowfall in the winter. Monthly precipitation averages range from approximately 20 mm/month between December and February to almost 120 mm/month in June. The majority of stream flow occurs during spring and summer, with peak stream flows in May and June.

Erosion

Fields in Clear Creek have some of the highest erosion rates in Iowa. These elevated rates are mainly due to a combination of swelling (i.e., smectite-rich) and highly erodible soils with steep slopes (up to 12%) and intensive agriculture (Piest et al. 1975; Cruse et al. 2006; Gilroy 2006). Reported erosion rates in Clear Creek averaged 20 Mt/ha/yr, with individual fields reporting erosion rates up to 150 Mt/ha/yr (e.g., Abaci and Papanicolaou 2009; Papanicolaou et al. 2009; Wilson et al. 2009).

Modeling Simulations

The simulation time period used for this study was from 1991 to 2012, which captures the time frame that conservation practices (namely, reduced tillage) have been practiced in Clear Creek. Daily values were determined during the simulation and the results were aggregated annually. The first 10 years were used as a spin-up period to allow the system to adjust to the new rotation (Wilson et al. 2009), with the results from the latter 10 years being averaged and used as the study results.

Modeling Framework

The modeling framework (Figure 2) used herein incorporates a bottom-up approach that integrates spatially distributed, process-based models to address erosion and biogeochemistry in an agroecosystem, coupled with an economic component. This bottom-up approach allows for capturing the cause-and-effect relationships between different cropping rotations and the processes of infiltration, runoff, and water-driven/tillage-enhanced erosion, which shape biogeochemical stabilization and degradation of SOC. Additionally, this framework allows us to incorporate the influence of heterogeneity in the vertical, downslope, and planar directions for

different landscape properties on these processes. These benefits translate to more accurate assessments of ecosystem services, as coarser estimates from lumped approaches fail to capture the combined effects of management and local conditions on nutrient cycling or the effects of processes like deposition within a field.

Erosion

This modeling framework (Figure 2) uses the Water Erosion Prediction Project, or WEPP *model (version* 2012.8) for simulating runoff and erosion (Papanicolaou et al. 2015b). WEPP is well established as a process-based model that works at both the hillslope and small watershed scales (e.g., Papanicolaou and Abaci 2008; Dermisis et al. 2010). Further details for WEPP can be found in Flanagan and Livingston (1995), Ascough et al. (1997), and Abaci and Papanicolaou (2009).

WEPP has a strong management component for agroecosystems with input parameters to describe the different crops, tillage regimes/schedules, and other practices. However, due to the importance of land management, and especially tillage, on soil erosion, soil respiration, and hence carbon budgets, a more concerted effort was made to capture the different degrees of tillage intensity across the watershed. Many past studies have shown that the degree of tillage intensity directly relates to the level at which it affects soil, biogeochemical, and hydrologic processes (e.g., Papanicolaou et al. 2015b). Therefore, to assess accurately the role of management on the overall functions of agroecosystems, it is important to distinguish between conservation and conventional tillage.

For our framework, a more-detailed land use classification map (Figure 1) was developed using satellite and aerial imagery with a correlation between soil reflectance and tillage intensity, namely the Normalized Difference Tillage Index, NDTI (e.g., Daughtry et al. 2006; Serbin et al. 2009). More disturbed soils appear darker due to less surface litter following tillage (Van Deventer et al. 1997). In this study, an implicit land management detection algorithm was developed using the NDTI to reclassify the row crop category in the National Land Cover Database (NLCD) maps into more detailed land management rotations (Figure 3).

Landsat images for the month of November were collected for areas in Clear Creek classified as row crop. November images were used to ensure that

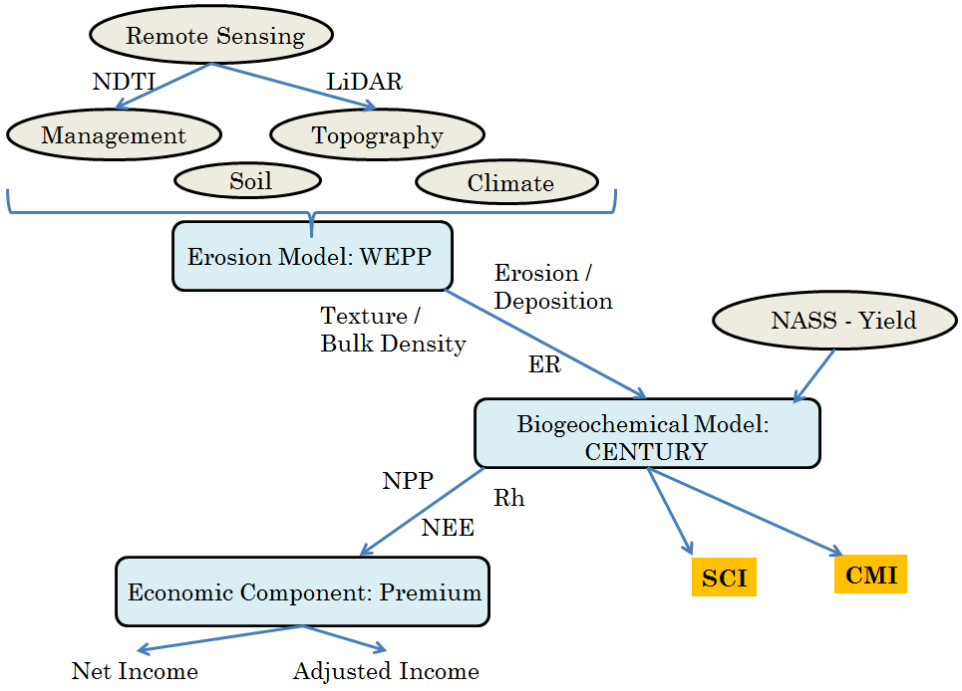


Figure 2. The linked WEPP-CENTURY modeling framework. From Papanicolaou et al. (2015b).

harvest and fall tillage practices were complete (Abaci and Papanicolaou 2009). The NDTI (Figure 3) was determined through a manipulation of reflectance wavebands (band 5, 1550–1750 nm; band 7, 2080–2350 nm) from the Landsat images (Daughtry et al. 2006). The average NDTI values for these fields ranged from 0.245 to 0.489. The NDTI values were then distributed throughout the corn and soybean fields of Clear Creek using the areal distribution of the selected management strategies found in Clear Creek (Table 1), with the lower NDTI values relating to higher tillage disturbance as shown in Figure 3a.

Along with the detailed management component, another benefit of WEPP is the ability to simulate multiple non-uniformities along a hillslope in terms of soils, slopes, and management by dividing the hillslope into smaller Overland Flow Elements (OFEs). This sub-division into OFEs by WEPP makes it a spatially distributed model that can effectively capture the overall effects of spatial heterogeneity in terms of hydropedological properties, land use, and hillslope curvature on runoff and erosion for different event, seasonal, and inter-annual periods (e.g., Abaci and Papanicolaou 2009; Dermisis et al. 2010). Capturing this heterogeneity in landscape properties is important due to its impact on observed flowpaths.

To define accurately the flowpaths, LiDAR data for the watershed, collected as part of the IML-CZO, were used to produce a digital elevation model (DEM), reclassified to a 5-m resolution. The DEM was then used to delineate the channel and hillslopes.

The hillslopes and channels in Clear Creek were connected using the geospatial version of WEPP, which incorporates subroutines like TOPAZ (Garbrecht et al. 1996) to pass water (and constituents) from one OFE to the next along topographic changes. Other WEPP subroutines are then applied individually in each OFE along the identified flowpaths for calculating runoff, infiltration, and net rill and interrill erosion. The "flowpath" approach takes our framework from the soil column to the landscape by allowing the physical mechanisms of runoff and erosion to redistribute SOC along a hillslope, in light of different management schedules for tillage and fertilizer, as well the presence of tiles, which influence carbon stocks at depth (Liu et al. 2011).

In addition to runoff and erosion rates, enrichment ratios (*ER*) were determined through WEPP, which

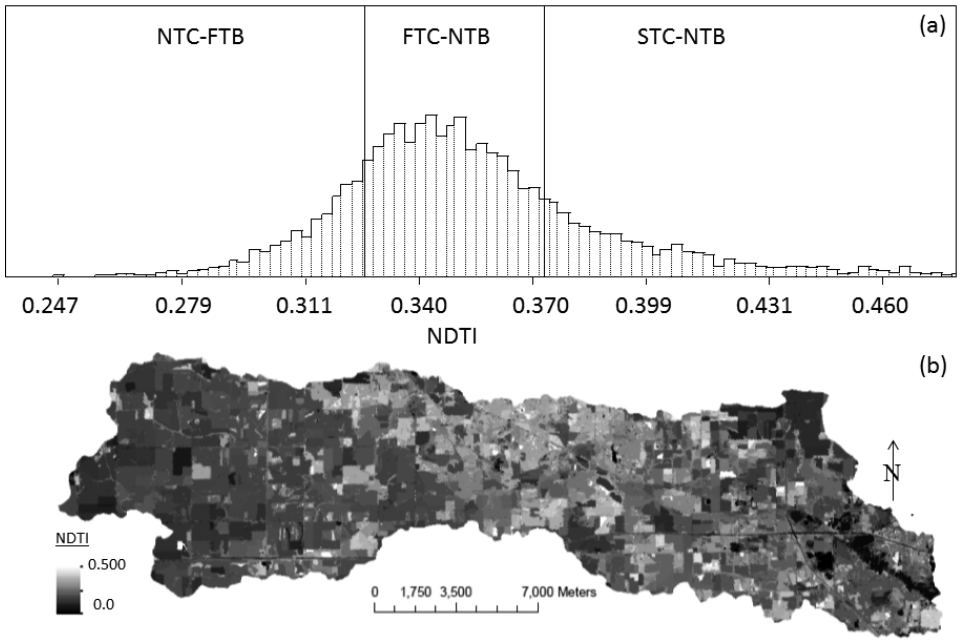


Figure 3. (a) Distribution of 10-year average Normalized Difference Tillage Index (NDTI) values for Clear Creek. (b) A map of the 10-year average NDTI values. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

are needed for quantifying the fraction of SOC that has been eroded (Papanicolaou et al. 2009; 2015b). *ER* is important for capturing the selective entrainment of lighter, but more organically rich soil. *ER* is expressed as the proportion of SOC in transported sediment compared to that of SOC in uneroded soil. The enrichment ratio was determined using the specific surface area of the soil (SSA_{SOIL}) within each hillslope and the specific surface area of the different size fractions of eroded sediment exiting a hillslope (SSA_{SED}) as follows:

$$ER = \frac{SSA_{Sed}}{SSA_{Soil}} \quad (1)$$

An $ER > 1$ indicates that the eroded sediment is enriched in SOM relative to the uneroded soil, whereas an $ER < 1$ denotes the opposite. In this study, it is important to note that due to long-term averaging, the functionality of the *ER* is limited since *ER* best capture the preferential redistribution of SOC at the event scale. Enrichment ratios within the Clear Creek watershed range between 0.8 and 3.0 for single events (Papanicolaou et al. 2009; 2015b), suggesting the likelihood of high rates of SOM loss during runoff events. Over the long-term, however, the *ER* approaches unity.

To show the benefits of using the process based WEPP model when determining erosion and SOC lost, the erosion rates from WEPP were compared against other lumped models, namely the E_{30} method and the USLE. The section below provides a short description of each method although the more detail-oriented reader is referred to the following publications: Hazarika and Honda (2001) for the E_{30} method, Wischmeier and Smith (1978) for the USLE, and Flanagan and Nearing (2000) for further information on WEPP.

The E_{30} method was developed for quantifying erosion in areas that are inaccessible for collecting physical measurements. The method relies on only the field slope, *S*, (normalized to a 30 degree slope, S_{30}) and remote sensing images for vegetative indices, such as the Normalized Difference Vegetation Index, NDVI (Hazarika and Honda 2001) to estimate annual erosion rates, *E*.

$$E = E_{30}\left(\frac{S}{S_{30}}\right)^{0.9} \quad (2)$$

where E_{30} is the rate of soil erosion for a 30 degree slope. It is quantified as below:

$$E_{30} = Exp\left[\left(\frac{ln(10)-ln(114)}{NDVI_{max}-NDVI_{min}}\right) \text{x } (NDVI-NDVI_{min}) + ln(114)\right] \quad (3)$$

The maximum and minimum erosion rates for a 30 degree slope in Clear Creek using a representative soil type and management were 10 and 114 mm/yr, respectively. These become the coefficients in Equation 3. The NDVI data were obtained from the Landsat-TM images for the region. Minimum NDVI values were obtained from April images spanning the 1991-2012 simulation period, while August images were used as the maximum NDVI images.

The USLE was also used to determine annual average erosion rates, *E*. The model develops individual indices for the dominant factors controlling erosion: rainfall and runoff erosivity (*R*), soil erodibility (*K*), slope length (*L*), slope steepness (*S*), vegetative cover/management practices (*C*), and conservation measures (*P*) (Equation 4) (Wischmeier and Smith 1978). The parameters of the USLE are empirically based on a large historical dataset. The values used in this study come from the USLE manual (Wischmeier and Smith 1978) and the available hydrologic, erosion, and biogeochemical databases of the IML-CZO (Table 2).

$$E = R \text{ x } K \text{ x } LS \text{ x } C \text{ x } P \quad (4)$$

The erosion rates from the E_{30} and USLE analyses were compared with the annual average rates provided by WEPP in the herein-developed modeling framework. WEPP determines erosion using an excess shear relationship:

$$\frac{\delta E(x)}{\delta x} = (\tau - \tau_c)(E_e - E(x)) \quad (5)$$

where τ is the shear stress of the runoff; τ_c is the critical shear stress of the soil; E_e is the equilibrium erosion rate; and $E(x)$ is the local erosion rate for a unit of the hillslope. It is important to note that the WEPP rates are summed along discrete lengths of the hillslope, which include both the upland erosional areas and the downslope depositional areas.

Biogeochemistry

Our framework (Figure 2) loosely couples WEPP with the CENTURY soil biogeochemical model (version 4.6) to address nutrient cycling (Papanicolaou et al. 2015b). CENTURY simulates carbon, nitrogen, phosphorus, and sulfur dynamics through plant-soil interactions in different grassland, agricultural, and forest ecosystems. Multiple sub-models (e.g., soil organic matter/ decomposition sub-model, water budget sub-model, and grassland/crop sub-model) are coupled with a management/event scheduling function to compute nutrient fluxes. For CENTURY, more detail can be found in Parton et al. (1987), Paustian et al. (1992), Harden et al. (1999), Manies et al. (2001), Jarecki et al. (2008), Tornquist et al. (2009), and Wilson et al. (2009).

WEPP and CENTURY were coupled to address certain limitations of existing biogeochemical models. As mentioned above, most biogeochemical models like CENTURY are limited to the soil column and cannot adequately capture the effects that spatial heterogeneity across the landscape (e.g., soil properties, tillage-induced roughness, crop coverage, patchiness), organizational complexity (e.g., relief, curvature, drainage network), and soil erosion/deposition have on SOC stocks (Van Oost et al. 2007; Kuhn et al. 2009; Papanicolaou et al. 2015b).

The net erosion, enrichment ratios, bulk density, and texture for each hillslope were passed from WEPP to CENTURY, which further accounts for the effects of plant phenology, litter fall, and decomposition on SOC concentrations. Net erosion rates and ER were used to quantify the loss of SOC due to redistribution during a runoff event. The flux of eroded carbon (SOC_{EROD}) was determined using the following relationship:

$$SOC_{EROD} = (TSOC)\left(\frac{ER \text{ x } EROD_{Net}}{BD \text{ x } d}\right) \quad (6)$$

where $TSOC$ is soil organic carbon; $EROD_{Net}$ is net erosion provided by WEPP; BD is soil bulk density; and d is depth of the active layer, which is the top 20 cm.

The loss of SOC due to erosion was added as a correction to the Net Ecosystem Exchange (*NEE*) calculations, which were used to determine carbon budgets in Clear Creek. The effects of erosion on SOC are often neglected in watershed carbon budgets (e.g., Brenner et al. 2001; West et al. 2010); however, erosion depletes SOC stocks directly (Papanicolaou et al. 2015b), and indirectly through changes in production and respiration.

NEE represents the net exchange of CO_2 from the biosphere to the atmosphere (Kirschbaum et al. 2001). It is the difference between Net Primary Productivity (*NPP*) and the microbial respiration from the soil (R_h):

$$NEE = -NPP + R_h \quad (7)$$

NPP is the gross plant production minus the CO_2 respired by the plants (Kirschbaum et al. 2001). In our framework, *Aboveground Net Primary Productivity* (*ANPP*) is determined using annual crop yields, *Y*, from the U.S. Department of Agriculture (USDA) - National Agriculture Statistics Service (NASS) or calculated with NDVI. These spatially averaged values are adjusted for each hillslope using the Corn Suitability Rating (Secchi and Babcock 2007) provided by the Iowa Soils Properties And Information Databases (ISPAID). The *ANPP* is then adjusted using a moisture weight correction for the plant, *MW* (normally 0.845), the fraction of carbon in the plant, C_{plant} (assumed to be 0.43; Latshaw and

Table 2. Universal Soil Loss Equation (USLE) parameters.

Rotation	R	K	LS	C	P	E (Mt/ha/yr)
NTC-FTB	63 – 250	0.31 ± 0.03	0.83 ± 0.54	0.3498	0.55 ± 0.05	20.5 ± 12.9
STC-NTB	63 – 250	0.32 ± 0.06	1.30 ± 0.57	0.3449	0.52 ± 0.04	33.9 ± 16.2
FTC-NTB	63 – 250	0.31 ± 0.04	1.16 ± 0.63	0.3527	0.53 ± 0.04	28.6 ± 16.7

NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

Miller 1924), and the average Harvest Index, *HI*, for corn-soybean rotations, reported as 0.50 (e.g., Prince et al. 2001; De bruin and Pederson 2009):

$$ANPP = \left(\frac{Y \text{ x } MW}{HI} \right) C_{plant} \quad (8)$$

The *Belowground Net Primary Production* (*BNPP*) is then estimated using the *ANPP* and the root to shoot ratio, *R:S,* in the following equation:

$$BNPP = ANPP \text{ x } R{:}S \quad (9)$$

For corn-soybean rotations, the *R:S* values are 0.18 (Prince et al. 2001).

The soil respiration component, R_h, in our framework, is determined by adjusting the maximum decomposition rate (K_i) for the effects of climate (*CDI*), soil texture (*TEX*), the prevalence of anaerobic conditions (*A*), and tillage ($Cult_i$), such that:

$$R_h = C_i \text{ x } K_i \text{ x } CDI \text{ x } TEX \text{ x } A \text{ x } Cult_i \quad (10)$$

where the *i* represents the various pools of carbon and *C* is the concentration of carbon. It is important to note that our framework accounts for the impact that tillage events have on respiration rates by increasing decomposition to reflect the breaking apart of soil aggregates and enhanced mineralization (Harden et al. 1999; Reicosky et al. 2005).

Economics

The economic component of this framework is represented by the net incomes of the different management strategies. The net incomes are determined as the difference between the crop yield returns and the production costs. The relative crop yield returns are quantified using average statewide prices and crop yields determined from above. The production costs (Table 3) attributed to machinery, planting, fertilizer, pesticides, and harvest, as well as costs for labor, land rental, and crop insurance (Duffy 2012) are included.

The approaches of Antle et al. (2007) and Feng et al. (2004) were adopted because they account for the production of ecosystem services based on the relative revenues of alternative production systems.

$$Net\ Income = Crop\ Yield\ Returns - Production\ Costs + Premium \quad (11)$$

The net incomes in this study are adjusted with a premium that is attributed to the additional ecosystem service benefits of lessening erosion as it relates to production costs and carbon storage.

For the soil loss reduction component of the premium term, the cost of fertilizer lost was considered. For each ton of eroded soil, which contains 1.05 kg of nitrogen and 0.45 kg of phosphorus, the cost of lost fertilizer alone is $2.10 (Iowa Learning Farm 2013). High erosion not only depletes the soil of organic matter and other nutrients, but it also contributes to water quality concerns. For example, in the winter of 2014-2015, the Des Moines Water Works spent $540,000 to run their nitrate removal equipment (Eller 2015). This cost for maintaining water quality was not considered in this study, but shows the extent of possible ramifications.

The part of the premium term related to SOC requires the carbon sequestered in a field determined from WEPP-CENTURY. The premium associated with SOC storage was set at a fixed price ($100/metric ton of C stored). This price is used for only demonstration purposes and is not a recommendation. This premium was estimated using different published sources. The average rate for a ton of carbon currently used by the cap and trade, emission trade schemes, and carbon taxes in place around the world is $115 (World Bank Group 2014). Past studies have proposed rates ranging for $10 to $125 per ton of carbon (e.g., Lewandrowski et al. 2004). Finally, enrolling a field in the Conservation Reserve Program (CRP), on average stores carbon at a rate of 0.92 Mt/ha/yr (Follett 2001; Post et al. 2004). With the current CRP rental rate in Iowa averaging $437/hectare, this translates to $66/metric ton of carbon. Further analysis would be needed to propose suggested target prices.

Indicators and Indices

The distinction between short-term productivity/profitability and long-term sustainability for each management strategy is established by assessing the response of different indicators and indices that reflect the associated management intensity.

Table 3. Production costs for different practices in Clear Creek ($/ha).

	FTC	STC	NTC	FTB	NTB	Grass
Planting	$47.20	$26.93	$38.30	$49.42	$38.30	$23.15
Seed	$238.20	$238.20	$238.20	$135.90	$135.90	n/a
Nitrogen	$152.14	$152.14	$152.14	$0.00	$0.00	$4.45
Herbicide	$87.72	$103.11	$103.11	$65.48	$65.48	n/a
Insurance	$30.15	$33.61	$33.61	$19.52	$19.52	n/a
Harvest	$103.90	$109.19	$109.19	$58.49	$58.49	$160.19
Labor	$83.52	$72.15	$73.88	$72.28	$72.28	$128.49
Land	$560.91	$674.57	$674.57	$560.91	$560.91	$328.64
Total	**$1,303.73**	**$1,409.91**	**$1,423.00**	**$962.00**	**$950.88**	**$644.92**

See Duffy (2012).

The specific indicators for productivity include Net Primary Productivity, which reflects crop production, and soil respiration, which measures biological decomposition activity and reflects the capacity of the soil to support life. In addition, the Net Ecosystem Exchange is also determined to provide a carbon budget for each rotation. In terms of profitability, the net income is quantified, as well as an adjusted income. The adjusted income includes a premium for lessening erosion and storing SOC in the soil.

To assess the sustainability of the different crop rotations, two indices are used that consider both erosion and carbon storage, which are dynamic parameters of an agroecosystem, namely the Soil Conditioning Index (*SCI*) and the Carbon Management Index (*CMI*). These dynamic parameters are better suited to assess a watershed's functionality and sustainability than the more basic soil health parameters (Doran and Parkin 1996; Lehman et al. 2015), as they reflect the response of the agroecosystem to different land management on similar timescales (e.g., seasonally). Traditional soil health parameters are more static and reflect long-term interactions between landscape features, climate, and biology (Stott et al. 2013).

The *SCI* (USDA - Natural Resource Conservation Service, NRCS 2002) is essentially a simple weighted sum of three, empirically derived factors: Organic Matter; Field Operations; and Erosion. However, the factors are complex in that they contain multiple input parameters and researchers must use care to select the correct terms to calculate the factors. The *Organic Matter* factor reflects the effect of biological inputs in terms of residue, roots, cover crops, and manure returned to the soil. The *Field Operations* factor reflects the effect of management on stimulating organic matter breakdown using the Soil Disturbance Rating. The *Erosion* factor reflects the effect of sheet and rill erosion at removing organic matter. It is based on the soil erosion rate in relation to the Soil Loss Tolerance value, or T-value, which defines the maximum rate of erosion at which the quality of a soil as a medium for plant growth can be maintained (USDA - Natural Resource Conservation Service, NRCS 2002).

The second index used, the Carbon Management Index, *CMI* (Blair et al. 1995), accounts for both the level of disturbance to the SOC stock relative to a reference value (i.e., Carbon Pool Index, *CPI*), and the change in the type of carbon due to a disturbance relative to a reference value (Lability Index, *LI*).

$$CMI = CPI \times LI \times 100 \quad (12)$$

The *CPI* is simply the ratio of the total SOC in the soil of an agricultural field relative to a reference value, such as an undisturbed prairie. *CMI* values closer to 100 mean you are closer to the reference condition. Larger differences between the total SOC in a field relative to the reference values reflects the intensity of the disturbance to

the field. Additionally, the *LI* is the ratio of labile carbon to non-labile carbon in an agricultural soil relative to a reference value. In this study, labile SOC is considered the material of the active pool of carbon determined from the CENTURY model, while non-labile SOC is considered from the more recalcitrant slow and passive pools derived in CENTURY (e.g., Cambardella and Elliot 1992; Vieira et al. 2007).

Results and Discussion

For the discussion below, we focus on only those hillslopes in Clear Creek under row-crop agriculture (Figure 1) to determine which management strategy is most beneficial for the long-term sustainability of the agroecosystem. As identified above, the three main row-crop strategies in Clear Creek are the following: no till corn – fall till bean (NTC-FTB); spring till corn – no till bean (STC-NTB); fall till corn – no till bean (FTC-NTB). Additionally, we define a sustainable practice as one that minimizes erosion and SOC loss while remaining productive and profitable in the long-term.

Productivity

The corn yields for all fields in Clear Creek averaged 393 ± 59 bu/ha, which fit in the range of yields (336 – 427 bu/ha) for east-central Iowa. Soybean yields averaged 121 ± 17 bu/ha, which also fit in the east-central Iowa range of values (106 – 128 bu/ha). Combined, these average corn-soybean yields for all rotations convert to an average annual total *NPP* of 599 ± 89 g C/m^2/yr. The NTC-FTB strategy, which had the highest tillage intensity, also had the highest average production 620 ± 100 g C/m^2/yr. In comparison, the least intense STC-NTB strategy had the lowest production at 567 ± 78 g C/m^2/yr. Although the difference is only about 9%, it is statistically significant (ANOVA; df = 2; F = 17.8; p < 0.001).

Past literature provides mixed results when relating tillage intensity and production (e.g., Phillips et al. 1980; Steiner 2002; Reicosky 2015), so it is difficult to attribute the yield differences entirely as a result of the specific management. A part of this difference may be due to where in the watershed the farmers are using the specific strategies. The inherent soil "fertility" also influences the productivity. Approximately 35% of the fields using FTC-NTB and 45% of the fields using NTC-FTB were in the 6 sub-watersheds with the highest average corn suitability ratings. This is in contrast to only 22% of the fields using STC-NTB. Therefore due to complicating factors including soil characteristics and the mixed results when relating tillage intensity and production, yield should not be the only indicator used to assess management sustainability.

Regarding soil respiration, the more intense strategies of NTC-FTB (385 ± 47 g C/m^2) and FTC-NTB (386 ± 50 g C/m^2) had respiration rates similar to the STC-NTB rotation (391 ± 51 g C/m^2; ANOVA; df = 2; F = 0.85; p = 0.4). From a modeling perspective, CENTURY accounts for tillage intensity, along with texture and the soil microclimate when determining soil respiration. However, the net effect of management on the R_h term can be similar despite the intensity.

Soil respiration is a direct reflection of microbial activity in the soil (Schlesinger and Andrews 2000). However, higher microbial activity may result from multiple mechanisms. Tillage will spark activity by bringing residue at the surface in contact with the microbes in the soil and by aerating the soil (e.g., Reicosky and Lindstrom 1993). Yet, the associated higher erosion rates will remove a high number of the microbes residing there, which will also tend to reduce respiration (Pimentel and Kounang 1998). Practices that leave more crop residue at the soil surface, like reduced and no-till practices (i.e., STC-NTB), add more organic matter to the soil thereby promoting soil respiration. These practices will also improve soil aggregation and porosity, meaning the soils are better aerated and hold more moisture (Papanicolaou et al. 2015a; Wacha 2016). Hence soil respiration is also not a good sole indicator to assess management.

For the NEE budgets, the more intense NTC-FTB (-194 ± 133 g C/m^2) and FTC-NTB (-196 ± 108 g C/m^2) strategies created larger sinks of carbon than the STC-NTB strategy (-155 ± 107 g C/m^2; ANOVA; df = 2; F = 7.63; p < 0.001). In looking at these carbon budget values for the three strategies, one may get the impression that the more intense strategies are suitable because they increase carbon storage. However, what

these numbers fail to show is that the majority of this stored carbon is in the above ground plant material. As used in Equation 9, the root-to-shoot ratio for corn-soybean rotations is 0.18 (Prince et al. 2001), which indicates very little of the biomass is belowground. Moreover, the harvest index for these crops used in Equation 8 is 0.5, so half of the aboveground C is removed entirely from the system and never goes into storage in the soil.

Enhanced crop production can mask the negative effects of intense tillage practices (Gliessman 2004). Advances in agrotechnology have pushed yields higher; however, it is important to look not only *where* the carbon is stored (i.e., aboveground), but also at the residence *time* of the SOC that is being stored (Burdock and Crawford 2015). For simplicity sake, SOC can be separated into two pools - one that decomposes quickly (or labile carbon), which reflects better the role of nutrient cycling, and one that decomposes slowly (or recalcitrant carbon), which reflects more the storage of carbon in the system. So are we storing fast, labile carbon which drives nutrient cycling? Or, is it slow, non-labile carbon that promotes long-term storage and sustainability of SOC, as well as physical stability of the soil? Land managers can enhance SOC storage by increasing overall organic matter input, the partition of recalcitrant carbon, and the formation of soil aggregates (Post et al. 2004).

Figure 4 shows the simulated trends of the recalcitrant carbon pool for the three strategies since their introduction around 1991. The amount of recalcitrant carbon for STC-NTB is highest and is steadily increasing throughout the simulation period ranging from 137 to 158 g C/m^2. The current stocks of the recalcitrant carbon for FTC-NTB and NTC-FTB (128 and 84 g C/m^2, respectively) are actually less than the initial values of 137 g C/m^2. In fact, the NTC-FTB stocks are continuing to decrease (Figure 4). Although the stocks for FTC-NTB are increasing, they are increasing at a rate of only 0.42 g C/m^2/yr, while the rate of increase for the STC-NTB is 3.5 times greater (1.47 g C/m^2/yr).

One central influence on the recalcitrant carbon stocks is the amount lost due to erosion, as the slow decomposition rates limit the carbon lost due to respiration. The effect of erosion is most apparent in the trend line for the NTC-FTB strategy with the punctuated drops in 1993, 1996, and 2008 (Figure 4). During those years, heavy rains resulting in large floods occurred in Clear Creek and the Midwest. As a result, the corresponding high amounts of erosion removed large quantities of SOC. These large drops are not seen in the STC-NTB strategy, where the decreased tillage keeps a high abundance of residue on the ground to protect against rainsplash mobilization and promotes aggregate formation (Abaci and Papanicolaou 2009; Wacha 2016). In summary, the STC-NTB strategy is losing less SOC, thereby allowing it to build up in the soil column.

Profitability

To assess the profitability of the different strategies, the net income was determined as the yield profit less the production costs. The NTC-FTB strategy had the highest production, and hence the highest yield profit. However, it also had the highest average production cost of $1191/ha/yr (Table 3). The highest net income was seen with the FTC-NTB strategy resulting from the combination of high yields and lower production costs. For comparison, FTC-NTB fields on average had 47% more profit than the STC-NTB fields.

In addition to the net income, a second indicator was determined to assess the profitability of the different strategies. The adjusted income accounts for the possibility of enhancing other ecosystem services, like lessening erosion and corresponding nutrient losses.

The NTC-FTB strategy had the highest tillage intensity, which resulted in the highest erosion and consequently the highest loss of SOC due to erosion. NTC-FTB lost, on average, more than double the soil (30 ± 20 Mt/ha/yr) and SOC (41.7 ± 48.6 g C/m^2/yr) compared to STC-NTB (12 ± 7 Mt/ha/yr and 20.5 ± 21.2 g C/m^2/yr, respectively). Comparatively, the second most intense strategy of FTC-NTB lost 54% more soil (19 ± 15 Mt/ha/yr) and 15% (23.6 ± 30.2 g C/m^2/yr) more SOC than STC-NTB (ANOVA; df = 2; F = 59.7 for erosion and 19.9 for SOC; $p < 0.001$).

To put this in different terms, in Iowa, nearly all soils have a Soil Loss Tolerance value of 11 Mt/ha/yr and the T-value provides a qualitative assessment at best over the long-term. On average, all three strategies had erosion rates greater than

the T-value (Figure 5). Yet for STC-NTB almost half of the hillslopes (48%) had erosion rates less than the T-value and only 10% of the hillslopes had values more than twice the T-value. For the other two strategies, less than a quarter of the hillslopes had erosion rates below the T-value with more than a third of the hillslopes having values more than double the T-value.

High erosion depletes the soil of organic matter and other nutrients. Over time, it will require more work (which translates to higher production costs) to sustain the productivity levels in light of these higher erosion rates depleting the soil. Again, for each ton of eroded soil, the cost of lost fertilizer is $2.10 (Iowa Learning Farm 2013). On average, the NTC-FTB and FTC-NTB strategies

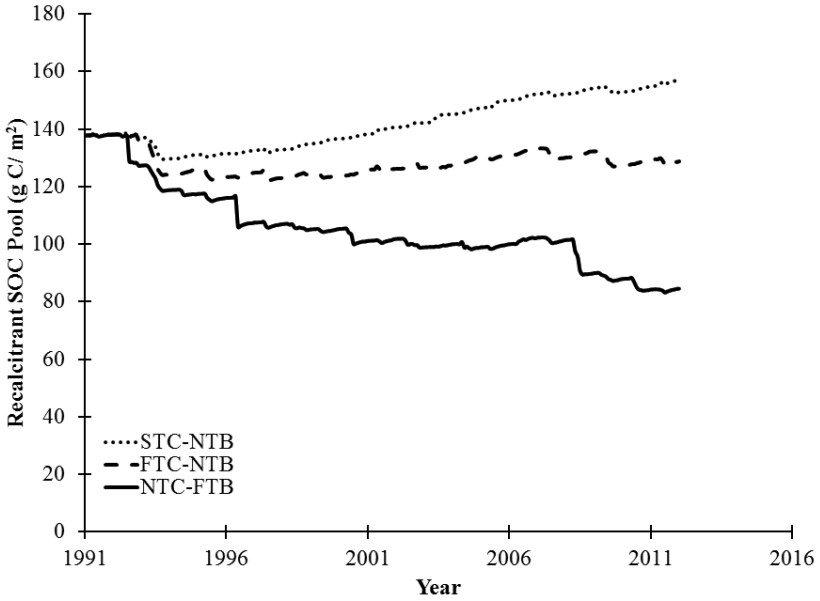


Figure 4. Changes in the recalcitrant pool of soil organic carbon. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

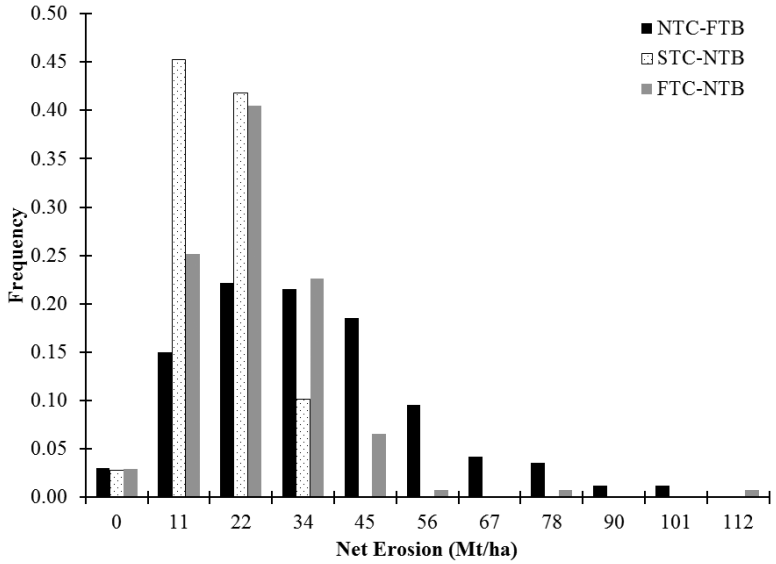


Figure 5. Net soil erosion histogram for each hillslope in Clear Creek that is practicing one of the three major crop rotations. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

lost 18 Mt/ha/yr and 6.5 Mt/ha/yr, respectively more soil than the STC-NTB strategy. This translates to a respective savings of $41.29 and $15.25 per hectare per year in added fertilizer costs, if the less intense STC-NTB is used. As fertilizer costs continue to rise (Huang 2009), this could dramatically alter the net incomes from the various practices. Moreover, it is unclear how the recent lawsuits brought by the Des Moines Water Works will translate to production costs (Eller 2015) through repercussions to the landowners.

A second component of the adjusted income was an additional incentive payment for storing SOC. Although there is no active carbon trading currently in the U.S., the process is firmly established around the globe (World Bank Group 2014). With dwindling government assistance for BMPs in this country, a market-based approach for carbon storage could be revisited as a means of promoting conservation efforts (Eller 2014). One positive example that could be followed is the Ohio River Basin Trading Project that focuses on water quality (i.e., nutrients such as phosphorus and nitrogen) and allows permitted emitters to purchase nutrient reductions from another source (http://wqt.epri.com/).

This premium for storing SOC is set initially, for demonstration purposes, as a fixed price of $100 per ton of carbon stored in the soil per year. On average the STC-NTB strategy had the highest premium at $17,017 ± $14,240/ha/yr. For the other two more intense strategies, the average premiums were less, but only slightly. Their higher production rates helped balance the carbon losses due to erosion. However, when accounting for the savings due to less erosion and the added benefit of the premium, the differences between the income from the STC-NTB strategy and the other two strategies drop to less than 16%, and they are no longer significant (ANOVA; df = 2; F = 1.33; p = 0.26).

Sustainability

The Soil Conditioning Index (Figure 6) is an ecological/biogeochemical index that reflects the ability of a soil to respond to the external influences of erosion and land management using SOC (Papanicolaou et al. 2009). It provides a sense of how well your system is handling these external stresses, either positively or negatively. The *SCI* in the fields practicing STC-NTB had the highest average value of 0.29 ± 0.16, relative to the other two strategies (ANOVA; df = 2; F = 65.9; $p < 0.001$). FTC-NTB still had a positive *SCI* of 0.19 ± 0.23, but NTC-FTB had an average value of 0.00 ± 0.31. The more positive *SCI* suggests the STC-NTB fields have an improving soil condition, and are more capable of handling potential shocks to the system, whether they come from the climate or from humans (Lehman et al. 2015). The *SCI* time series in Figure 6 shows considerable variability as the crops switch between corn and soybean. This variability prevents any clear trends from being discerned in the soil condition that may suggest advancement towards sustainability, thereby limiting its utility.

The Carbon Management Index or *CMI* (Blair et al. 1995), considers two aspects of SOC. One is the level of disturbance to the SOC stock relative to a reference value (i.e., Carbon Pool Index, *CPI*). The second relates to residence time and looks at the lability of the SOC due to a disturbance (i.e., Lability Index, *LI*). Changes in SOC lability can be a measure of sustainability. Short-term productivity is driven by nutrient cycling of labile carbon; however, long-term, sustainable productivity is driven by the recalcitrant SOC levels (Post et al. 2004). Both are reflections of management and microbial activity, despite the different residence times. A balance should be maintained to keep current productivity levels high, but having an eye on future sustainable production.

The *CMI* for each strategy increased initially once reduced tillage was initiated in the watershed (Figure 6). However, the *CMI* for the two more intense strategies of FTC-NTB and NTC-FTB began to decrease in recent years. On average, the *CMI* for NTC-FTB rose initially from a value of 50 to 66 after 10 years, then it began to decrease to an average value around 60. For FTC-NTB, it took about 18 years for the *CMI* to peak at a value of 75 and then decrease. The decreasing trend is a reflection that these strategies are not sustainable despite that they may be more productive or have higher net income in the short-term. The least intense tillage schedule in the STC-NTB strategy

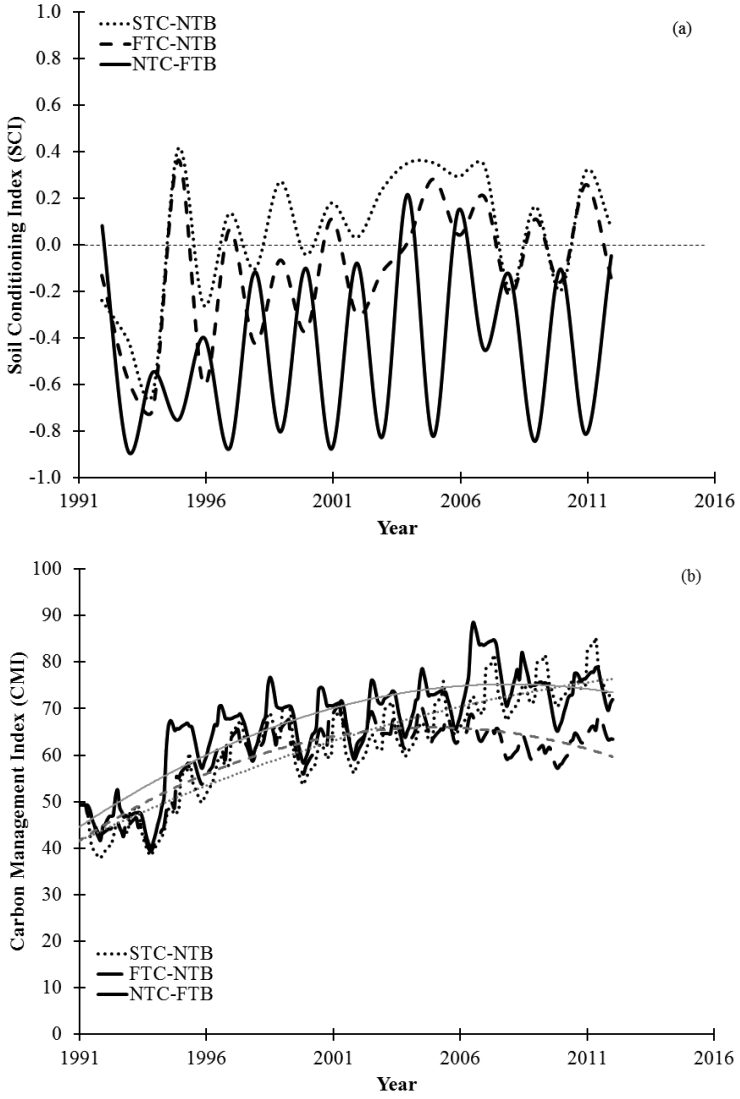


Figure 6. Sustainability figures of Soil Conditioning Index (a) and Carbon Management Index (b). The *SCI* and *CMI* values for the three main crop rotations in Clear Creek. *CMI* values closer to 100 represent a practice that is closest to the reference site of a prairie. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

helped minimize the depletion of SOC through erosion. As a result, it has the highest *CMI* at the end of the period with an average value of 77 and the trend is still increasing. In addition, despite the higher productivity from the FTC-NTB and NTC-FTB strategies, they had lesser amounts of non-labile carbon, which is needed to maintain SOC stocks over the long-term, relative to the STC-NTB strategy (Figure 4). This is most likely related to the higher erosion losses removing non-labile carbon, as well as the breaking of soil aggregates (Wacha 2016).

In summary, the *CMI* is a valuable addition to the list of indices for system functionality as it provides a clear picture of improvement within an agroecosystem. The *CMI* uncovers the underlying soil degradation occurring with some intensive management strategies, despite the fact that they are producing high yields with substantial inputs of labile SOC to the system. The added labile SOC cannot replace the more recalcitrant SOC lost with erosion. In this case it appears that the least intense strategy of STC-NTB is best for sustaining the productivity of the field.

The Role of Heterogeneity

One of the benefits of the bottom-up approach used in this study is its capability to capture heterogeneity related to different soil properties, which dictates the flowpaths of runoff, sediment, carbon, and other nutrients. As the soil and attached SOC are redistributed across the landscape, the natural heterogeneity is augmented. The importance of capturing this heterogeneity is shown using a three level analysis, which included the use of the E_{30} method, USLE, and WEPP. The E_{30} method was used as a reference condition for soil mobilization, as it only really incorporates the role of hillslope gradient. The USLE method considers not only the hillslope gradient, which is pseudo-normalized using the slope length, but also considers soil characteristics, rainfall/runoff, and management effects in a lumped sense to determine annual average erosion rates. These erosion rates are for gross erosion though, since deposition is not considered. This lumped nature of the USLE is compared against the process-based, spatially distributed WEPP model. WEPP can break the hillslope into more discrete OFEs in order to capture the depositional areas at the bottom of the downslope.

The comparison results of the three methods are shown in Figure 7 and clearly heterogeneity matters. The annual rates of net erosion determined by WEPP (average = 20 ± 16 Mt/ha/yr) are less when compared to the E_{30} (average = 31 ± 13 Mt/ha/yr) and USLE (average = 27 ± 16 Mt/ha/yr) methods (ANOVA; df = 2; F = 91.5; $p < 0.001$). We attribute this reduction in net erosion due to the deposition that occurs in the hillslopes. The E_{30} and USLE methods do not capture the beneficial effects that conservation practices have on reducing soil erosion and over-predict annual erosion rates relative to the spatially-distributed WEPP values.

For the NTC-FTB, the average net erosion rates predicted by WEPP are higher than the ones estimated by the E_{30} and USLE. Clearly the latter two methods cannot account for the dynamic within-season changes due to the absence of cover and residue during the high storm-intensity period of May – June in Clear Creek (Abaci and Papanicolaou 2009). The 3-method simulations highlight the need for using a physically based, spatially oriented model like WEPP for addressing issues of soil quality, estimation of dynamic indices, and sustainability in intensively managed landscapes.

Summary and Conclusions

Our central objective for this study was to develop a framework that can assess how well

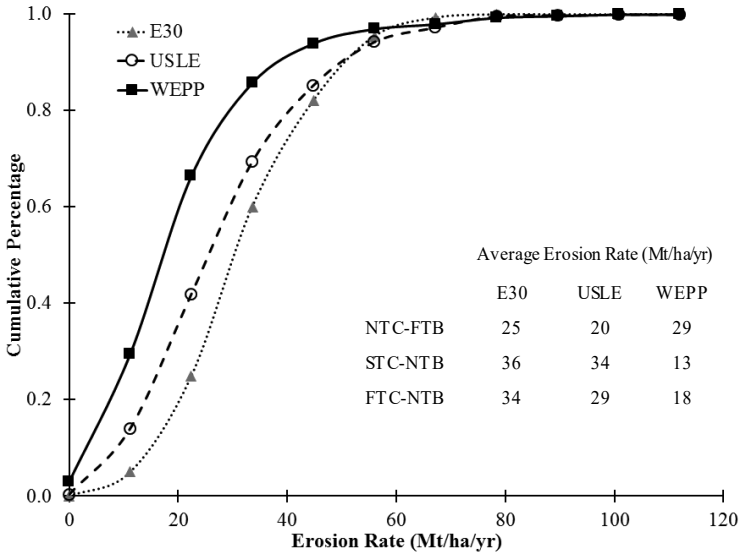


Figure 7. Cumulative frequency distribution of annual erosion rates using different methods that incorporate higher degrees of spatial heterogeneity: E30, USLE, and WEPP. NTC-FTB = no till corn – fall till bean; STC-NTB = spring till corn – no till bean; and FTC-NTB = fall till corn – no till bean.

land management strategies balance short-term productivity and profitability with the long-term sustainability of the agroecosystem. Our presented framework uses a bottom-up approach that integrates two established, process-based models, namely WEPP and CENTURY, with an economic assessment that considers the production of ecosystem services related to food security and soil organic carbon. The bottom-up approach allows us to capture better the overall effects of spatial heterogeneity, in terms of hydropedological properties, land use/cover, and hillslope curvature, on erosion for different event, seasonal, and inter-annual periods.

Most biogeochemical models (including CENTURY) are limited to the soil column and cannot adequately reproduce the effects that spatial heterogeneity across the landscape, organizational complexity, soil erosion/deposition, and land-use management practices have on SOC stocks. These drawbacks have direct implications on the design of efficient economic policies for providing incentive payments or credits to potential producers. WEPP applies a landscape-oriented approach to CENTURY, and allows us to consider physical mechanisms affecting SOC storage along a hillslope, including erosion and deposition, management schedules for tillage and fertilizer, and tiles which affect carbon stocks at depth. Hence, our framework allows us to improve our carbon budget predictions by adding a correction to the NEE equation that accounts for erosion directly and indirectly through changes in respiration and production.

However, it is not enough to just account for erosion. Herein we show the potential errors in accounting for erosion through the use of lumped models that neglect the redistribution of soil (and associated SOC). The E_{30} methods and USLE methods do not capture the beneficial effects that conservation practices have on lessening soil erosion and over predict annual erosion rates compared to the WEPP derived erosion rates.

Finally, our framework used the Carbon Management Index, *CMI*, which provided the clearest picture of improvement within an agroecosystem. The *CMI* captures both the level of the management disturbance through the loss of SOC compared to a reference value, and a reflection of the type of carbon, in terms of residence time, being stored in the system. In Clear Creek, it appears that the least intense strategy of STC-NTB is best for sustaining the productivity of a field. This is seen through the combination of the Carbon Pool Index, *CPI*, which captures the level of the management disturbance through the loss of SOC compared to a reference value, with the Lability Index, *LI*, which reflects the type of carbon being stored in the system.

The "take-home" message from this study is that when developing future land management strategies for agroecosystems, it is important to consider those practices that promote the production of organic matter in the soil. SOC is inherently tied to nearly all key soil functions in some fashion. Current practices are geared for short-term high production. However, there is a high level of soil degradation associated with these practices that threaten the long-term sustainability of our agroecosystems.

In this study, it was shown that no-till or reduced tillage practices allow SOC to build back in the soil after years of high erosion. Additionally, cover crops can provide seasonal protection from raindrop impact and runoff, as well as increase soil organic matter as a form of green manure. Multiple crop rotations have long been seen as a way to avoid depleting the soil of nutrients and improving soil resiliency against pest propagation. The mixing of legumes with a high nitrogen-demanding crop, followed by a low nitrogen-demanding crop, can help reduce fertilizer applications and production costs. The application of manure may be another way of reducing fertilizer costs, as well as providing additional soil health benefits. Finally, the use of precision farming can target only those areas needing extra work and can help minimize production costs, while potentially limiting water quality problems.

These practices can be readily implemented by the farmers as they rely more on the farmer rather than on institutional help. BMPs, such as grassed waterways and riparian buffer strips, can be beneficial but they are more focused to minimize water quality concerns, as opposed to soil health concerns. In addition, funding for these practices is dwindling away. Therefore, when developing an outreach message for farmers

and all other stakeholders in the decision-making tree of the agroecosystem, it should be stressed that implementing SOC-enhancing practices are the best bet to improve system functionality, and hence system health.

Acknowledgements

This research was primarily supported by the Leopold Center for Sustainable Agriculture [agreement #XP-2013-10]. Additionally, this research was partially supported by the NASA EPSCoR Program [Grant #NNX10AN28A] and the Iowa Space Grant Consortium [Grant # NNX10AK63H], as well as the National Science Foundation Grant # EAR1331906 for the Critical Zone Observatory for Intensively Managed Landscapes [IML-CZO], a multi-institutional collaborative effort. The second author was partially supported by a Graduate Assistantship in Areas of National Need [GAANN] Fellowship and, along with the sixth author, by the University of Iowa NSF IGERT program, Geoinformatics for Environmental and Energy Modeling and Prediction. The funds from an organized research unit (ORU) grant at the University of Tennessee provided partial support for PI Papanicolaou and the fifth author.

Author Bio and Contact Information

Christopher G. Wilson is an Assistant Research Professor in the Department of Civil and Environmental Engineering at the University of Tennessee, Knoxville. He may be reached at cgw24@utk.edu.

Kenneth M. Wacha was a Ph.D. student in IIHR – Hydroscience & Engineering, Department of Civil & Environmental Engineering, at the University of Iowa. He has recently graduated and is currently a post-doctoral research associate at the USDA-ARS National Lab for Agriculture and the Environment in Ames , IA. He may be reached at Ken.Wacha@ARS.USDA.GOV.

Thanos Papanicolaou (corresponding author) is Professor and Henry Goodrich Chair of Excellence in Civil and Environmental Engineering at the University of Tennessee, Knoxville is currently the Director of the Tennessee Water Resources Center. Additionally, Papanicolaou is co-Director of the NSF Intensively Managed Landscapes – Critical Zone Observatory, as well as Director of the UTK Hydraulics and Sedimentation Laboratory and Chief Editor of the Journal of Hydraulic Engineering, ASCE. His research agenda examines the interplay between key driving factors of erosion, climate, topography, soil biogeochemistry, and land management with a focus on how climate and anthropogenic activities have affected land productivity, as well as water, food, and energy supplies over the years. He may be contacted at tpapanic@utk.edu or at Department of Civil and Environmental Engineering, 851 Neyland Dr., Room 325, Knoxville, TN 37996.

Heather A. Sander is Assistant Professor in the Department of Geographical and Sustainability Sciences at the University of Iowa. She may be reached at heather-a-sander@uiowa.edu.

Violet B. Freudenberg is a M.Sc. student in the Department of Civil and Environmental Engineering at the University of Tennessee, Knoxville. She may be reached at vfreuden@vols.utk.edu.

Benjamin K.B. Abban is a Ph.D. student in the Department of Civil and Environmental Engineering at the University of Tennessee, Knoxville. He may be reached at babban@vols.utk.edu.

Chang Zhao is a Ph.D. student in the Department of Geographical and Sustainability Sciences at the University of Iowa. She may be reached at chang-zhao@uiowa.edu.

References

Abaci, O. and A.N. Papanicolaou. 2009. Long-term effects of management practices on water-driven soil erosion in an intense agricultural sub-watershed: Monitoring and modelling. *Hydrological Processes* 23(19): 2818-2837.

Ahmadov, R., C. Gerbig, R. Kretschmer, S. Koerner, B. Neininger, A.J. Dolman, and C. Sarrat. 2007. Mesoscale covariance of transport and CO_2 fluxes: Evidence from observations and simulations using the WRF-VPRM coupled atmosphere-biosphere model. *Journal of Geophysical Research* 112: D22107. DOI:10.1029/2007JD008552.

Antle, J.M. and S. Mooney. 2002. Designing efficient policies for agricultural soil carbon sequestration. In: *Agricultural Practices and Policies for Carbon Sequestration on Soil*, J.M. Kimble, R. Lal, and R.F. Follett (Eds.). Lewis Publishers, Boca Raton, FL. pp. 323-336.

Antle, J.M., S.M. Capalbo, K. Paustian, and M.K. Ali. 2007. Estimating the economic potential for agricultural soil carbon sequestration in the Central United States using an aggregate econometric-

process simulation model. *Climatic Change* 80: 145-171.

Ascough, J.C., C. Baffaut, M.A. Nearing, and B.Y. Liu. 1997. The WEPP watershed model: I. Hydrology and erosion. *Transactions of the American Society of Agricultural Engineers* 40: 921-933.

Babcock, B.A. 2010. Costs and benefits of moving to county ACRE program. CARD Policy Brief 10 – PB 2, Center for Agricultural and Rural Development, Iowa State University. Available at http://www.card.iastate.edu/policy_briefs/display.aspx?id=1128. Accessed December 2010.

Bettis III, E.A., D.R. Muhs, H.M. Roberts, and A.G. Wintle. 2003. Last glacial loess in the conterminous USA. *Quaternary Science Review* 22: 1907-1946.

Billings, S.A., R.W. Buddemeier, D. Richter, K. Van Oost, and G. Bohling. 2010. A simple method for estimating the influence of eroding soil profiles on atmospheric CO_2. *Global Biogeochemical Cycles* 24: GB2001. DOI:10.1029/2009GB003560.

Blair, G.J., R.D.B. Lefroy, and L. Lisle. 1995. Soil carbon fractionations based on the degree of oxidation and the development of a Carbon Management Index for agricultural systems. *Australian Journal of Agricultural Research* 46: 1459-1466.

Brenner, J., K. Paustian, G. Bluhm, J. Cipra, M. Easter, T. Elliott, T. Kautza, K. Killian, J. Schuler, and S. Williams. 2001. Quantifying the change in greenhouse gas emissions due to natural resource conservation practice application in Iowa: "The Iowa Carbon Storage Project". A Final Report to the Iowa Conservation Partnership. Des Moines, IA.

Burdock, R.P. and J.W. Crawford. 2015. Combining micro-bottom-up and macro-top-down modelling responses to nutrient cycles in complex agricultural systems. *Nutrient Cycling in Agroecosystems* 103: 257-278.

Burkart, M., D. James, M. Liebman, and C. Herndl. 2005. Impacts of integrated crop-livestock systems on nitrogen dynamics and soil erosion in western Iowa watersheds. *Journal of Geophysical Research* 110: G01009. DOI:10.1029/2004JG000008.

Cambardella, C.A. and E.T. Elliott. 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. *Soil Science Society of America Journal* 56: 777-783.

Costanza, R., R. d'Arge, R. de Groot, S. Farberk, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Cruse, R.M., D.C. Flanagan, J.R. Frankenberger, B.K. Gelder, D. Herzmann, D.E. James, W. Krajewski, M. Kraszewski, J.M. Laflen, J. Opsomer, and D. Todey. 2006. Daily estimates of rainfall, water runoff, and soil erosion in Iowa. *Journal of Soil and Water Conservation* 61(4): 191-199.

Dale, V.H., R.A. Efroymson, K.L. Kline, M.H. Langholtz, P.N. Leiby, G.A. Oladosu, M.R. Davis, M.E. Downing, M.R. Hilliard. 2013. Indicators for assessing socioeconomic sustainability of bioenergy systems: A short list of practical measures. *Ecological Indicators* 26: 87-102.

Daughtry, C.S., P.C. Doraiswamy, E.R. Hunt, A.J. Stern, J.E. McMurtrey, and J.H. Prueger. 2006. Remote sensing of crop residue cover and soil tillage intensity. *Soil & Tillage Research* 91(1): 101-108.

De Bruin, J.L. and P. Pedersen. 2009. Growth, yield, and yield component changes among old and new soybean cultivars. *Agronomy Journal* 101: 124-130.

de Groot, R.S., M.A. Wilson, and R.M.J. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41: 393-408.

Dermisis, D., O. Abaci, A.N. Papanicolaou, and C.G. Wilson. 2010. Evaluating grassed waterway efficiency in southeastern Iowa using WEPP. *Soil Use and Management* 26: 183-192.

Dideriksen, R.O., M.R. LaVan, K.K. Norwood, S.R. Steckly, and J.E. Steele. 2007. *Soil survey of Iowa County, Iowa*. USDA-NRCS.

Dlugoß, V., P. Fiener, and K. Schneider. 2010. Layer-specific analysis and spatial prediction of soil organic carbon using terrain attributes and erosion modeling. *Soil Science Society of America Journal* 74: 922-935.

Dlugoß, V., P. Fiener, K. Van Oost, and K. Schneider. 2012. Model based analysis of lateral and vertical soil carbon fluxes induced by soil redistribution processes in a small agricultural catchment. *Earth Surface Processes & Landforms* 37(2): 193-208.

Doran, J.W. and T.B. Parkin. 1996. Quantitative indicators of soil quality: A minimum data set. In: *Methods for Assessing Soil Quality*, J.W. Doran and A.J. Jones (Eds.). Soil Science Society of America Special Publication No. 49. Soil Science Society of America, Madison, WI. pp. 25-37.

Duffy, M. 2012. *Estimated Costs of Crop Production in Iowa, 2012*. Iowa State University Extension Publication, FM1712, Dec. 2011.

Eller, D. 2014. Erosion estimated to cost Iowa $1 billion in yield. *Des Moines Register* May 3, 2014. Available at http://www.desmoinesregister.com/story/money/agriculture/2014/05/03/erosion-estimated-cost-iowa-billion-yield/8682651/. Accessed July 2015.

Eller, D. 2015. Fight over Des Moines Water Works lawsuit heats up. *Des Moines Register* May 19, 2015. Available at http://www.desmoinesregister.com/story/money/agriculture/2015/05/14/water-works-nitrates-lawsuit/27331305/. Accessed July 2015.

Feng, H., C.L. Kling, and P.W. Gassman. 2004. Carbon sequestration, co-benefits, and conservation programs. *Choices* 19(3): 19-23.

Fischer, T., D. Byerlee, and G. Edmeades. 2014. Crop yields and global food security. Will yield increases continue to feed the world? ACIAR Monograph. 158. Australian Centre for Int. Agric. Res., Canberra.

Flanagan, D.C. and S.J. Livingston. 1995. Water Erosion Prediction Project (WEPP) user summary. In: *USDA – Water Erosion Prediction Project: Hillslope Profile and Watershed Model Documentation*. National Soil Erosion Research Laboratory Report #11.1–11.2.-NSERL report No.11. USDA-ARS, National Soil Erosion Research Laboratory, West Lafayette, IN.

Flanagan, D.C. and M.A. Nearing. 2000. Sediment particle sorting on hillslope profiles in the WEPP model. *Transactions of the American Society of Agricultural Engineers* 43(3): 573-583.

Follett, R.F. 2001. Soil management concepts and carbon sequestration in cropland soils. *Soil & Tillage Research* 61: 77-92.

Garbrecht, J., P.J. Starks, and L.W. Martz. 1996. New digital landscape parameterization methodologies. In: *Proceedings of the 32nd Annual Conference and Symposium on GIS and Water Resources, American Water Resources Association,* C.A. Hallam, J.M. Salisbury, K.J. Lanfear, and W.A. Battaglin (Eds.). Herndon, VA, pp. 357-365.

Gilroy, A. 2006. Detecting change in soil erosion in 8-digit Hydrologic Units in Iowa: Correlation between level of soil erosion and active conservation practices from 1992-1997. *Papers in Resource Analysis* 8: 1-10.

Gliessman, S.R. 2004. Chapter 2: Agroecology and Agroecosystems. In: *Agroecosystems Analysis*, American Society of Agronomy, Madison, WI, pp. 19-30.

Gregorich, E.G., K.J. Greer, D.W. Anderson, and B.C. Liang. 1998. Carbon distribution and losses: Erosion and deposition effects. *Soil & Tillage Research* 47: 291-302.

Harden, J.W., J.M. Sharpe, and W.J. Parton. 1999. Dynamic replacement and loss of soil carbon on eroding cropland. *Global Biogeochemical Cycles* 13: 885-901.

Hazarika, M.K. and K. Honda. 2001. Estimation of soil erosion using remote sensing and GIS, its valuation, and economic implications on agricultural production. In: *Sustaining the Global Farm*. *Proceedings from the 10th International Soil Conservation Organization Meeting*. D.E. Stott, R.H. Mohtar, and G.C. Steinhardt (Eds.). West Lafayette, IN.

Highland, J.D. and R.I. Dideriksen. 1967. *Soil survey of Iowa County, Iowa*. U.S. Department of Agriculture, Soil Conservation Service, in cooperation with the Iowa Agricultural Experiment Station.

Hester, E.T. and J.C. Little. 2013. Measuring environmental sustainability of water in watersheds. *Environmental Science & Technology* 47: 8083-8090.

Huang, W.Y. 2009. Factors contributing to the recent increase in U.S. Fertilizer Prices, 2002-08. A Report to the Economic Research Service AR-33. Washington, D.C., pp. 21.

Iowa Learning Farm. 2013. *The Cost of Soil Erosion*. Iowa State University, Ames, IA.

Jarecki, M.K., T.B. Parkin, A.S.K. Chan, J.L. Hatfield, and R. Jones. 2008. Greenhouse gas emissions from two soils receiving nitrogen fertilizer and swine manure slurry. *Journal of Environmental Quality* 37: 1432-1438.

Jones, R.L., B.W. Ray, J.B. Fehrenbacher, and A.H. Beavers. 1967. Mineralogical and chemical characteristics of soils in loess overlying shale in northwestern Illinois. *Soil Science Society of America Proceedings* 31: 800-804.

Karlen, D.L., D.E. Stott, C.A. Cambardella, R.J. Kremer, K.W. King, and G.W. McCarty. 2014. Surface soil quality in five Midwestern cropland Conservation Effects Assessment Project watersheds. *Journal of Soil and Water Conservation* 69(5): 393-401.

Kirschbaum, M.U.F., D. Eamus, R.M. Grifford, S.H. Roxburgh, and P.J. Sands. 2001. Definitions of some ecological terms commonly used in carbon accounting. NEE Workshop Proceedings, April 18-20, 2001.

Klumper, W. and M. Qaim. 2014. A meta-analysis of the impacts of genetically modified crops. *PLoS ONE* 9(11): e111629. DOI:10.1371/journal.pone.0111629.

Kuhn, N., T. Hoffmann, W. Schwanghart, and M. Dotterweich. 2009. Agricultural soil erosion and global carbon cycle: Controversy over? *Earth Surface Processes and Landforms* 34: 1033-1038.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304: 1623-1627.

Latshaw, W. and E. Miller. 1924. Elemental composition of the corn plant. *Journal of Agricultural Research* XXVII: 845-860.

Lehman, R.M., C.A. Cambardella, D.E. Stott, V. Acosta-Martinez, D.K. Manter, J.S. Buyer, J.E. Maul, J.L. Smith, H.P. Collins, J.J. Halvorson, R.J. Kremer, J.G. Lundgren, T.F. Ducey, V.L. Jin, and D.L. Karlen. 2015. Understanding and enhancing soil biological health: The solution for reversing soil degradation. *Sustainability* 7: 988-1027.

Lewandrowski, J., M. Peters, C. Jones, R. House, M. Sperow, M. Eve, and K. Paustian. 2004. *Economics of Sequestering Carbon in the U.S. Agricultural Sector*. USDA – Economic Research Service, Technical Bulletin No. 1909, Washington, D.C., pp. 69.

Liu, S., Z. Tan, Z. Li, S. Zhao, and W. Yuan. 2011. Are soils of Iowa USA currently a carbon sink or source? Simulated changes in SOC stock from 1972 to 2007. *Agriculture, Ecosystems, and Environment* 140: 106-112. DOI:10.1016/j.agee.2010.11.017.

Manies, K.L., J.W. Harden, L. Kramer, and W. Parton. 2001. Carbon dynamics within agricultural and native sites in the loess region of western Iowa. *Global Change Biology* 7: 545-555.

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-being: Biodiversity Synthesis*. World Resources Institute, Washington, D.C.

Monreal C.M., R.P. Zentner, and J.A. Robertson. 1997. An analysis of soil organic matter dynamics in relation to management, erosion and yield of wheat in long-term crop rotation plots. *Canadian Journal of Soil Science* 77(4): 553-563.

Papanicolaou, A.N. and O. Abaci. 2008. Upland erosion modeling in a semi-humid environment via the water erosion prediction project model. *Journal of Irrigation & Drainage Engineering* 134: 796-806.

Papanicolaou, A.N., M. Elhakeem, C.G. Wilson, C.L. Burras, L.T. West, H. Lin, B. Clark, and B.E. Oneal. 2015a. Spatial variability of saturated hydraulic conductivity at the hillslope scale: Understanding the role of land management and erosional effect. *Geoderma* 243-244: 58-68. DOI:10.1016/j.geoderma.2014.12.010.

Papanicolaou, A.N., K.M. Wacha, B.K. Abban, C.G. Wilson, J. Hatfield, C. Stanier, and T. Filley. 2015b. From soilscapes to landscapes: A landscape-oriented approach to simulate soil organic carbon dynamics in Intensely Managed Landscapes. *Journal of Geophysical Research – Biogeosciences* 120(11): 2375-2401.

Papanicolaou, A.N., C.G. Wilson, O. Abaci, M. Elhakeem, and M. Skopec. 2009. Soil quality in Clear Creek, IA: SOM loss and soil quality in the Clear Creek, IA Experimental Watershed. *Journal of Iowa Academy of Science* 116(1-4): 14-26.

Parton, W.J., D.S. Schimel, C.V Cole, and D.S. Ojima. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Science Society of America Journal* 51: 1173-1179.

Paustian, K., W.J. Parton, and J. Persson. 1992. Modeling soil organic matter in organic-amended and nitrogen-fertilized long-term plots. *Soil Science Society of America Journal* 56: 476-488.

Phillips, R.E., R.L. Blevins, G.W. Thomas, W.W. Frye, S.H. Phillips. 1980. No-tillage agriculture. *Science* 208(4448): 1108-1113.

Piest, R.F., L.A. Kramer, and H.G. Heinemann. 1975. Sediment movement from loessial watersheds. In: *Proceedings of the Sediment-yield Workshop: Present and Prospective Technology for Predicting Sediment Yields and Sources*. ARS-S-40. Agricultural Research Service, Washington D.C., pp. 130-141.

Pimentel, D., C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Crist, L. Sphpritz, L. Fitton, R. Saffouri, and R. Blair. 1995. Environmental and economic costs of soil erosion and conservation benefits. *Science* 267: 1117-1123.

Pimentel, D. and N. Kounang. 1998. Ecology of soil erosion in ecosystems. *Ecosystems* 1: 416-426.

Polasky S., E. Nelson, J. Camm, B. Csuti, P. Fackler, E. Lonsdorf, C. Montgomery, D. White, J. Arthur, B. Garber-Yonts, R. Haight, J. Kagan, A. Starfield, and C. Tobalske. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biological Conservation* 141(6): 1505-1524.

Porter, P.A., R.B. Mitchell, and K.J. Moore. 2015. Reducing hypoxia in the Gulf of Mexico: Reimagining a more resilient agricultural landscape

in the Mississippi River Watershed. *Journal of Soil and Water Conservation* 70(3): 63A-68A.

Post, W.M., R.C. Izaurralde, J.D. Jastrow, B.A. McCarl, J.E. Amonette, V.L. Bailey, P.M. Jardine, T.O. West, and J. Zhou. 2004. Enhancement of carbon sequestration in U.S. Soils. *BioScience* 54(10): 895-908.

Prato, T. 2003. Adaptive management of large rivers with special reference to the Missouri River. *Journal of the American Water Resources Association* 39(4): 935-946.

Prince, S.D., J. Haskett, M. Steininger, H. Strand, and R. Wright. 2001. Net primary production of U.S. Midwest croplands from agricultural harvest data. *Ecological Applications* 11: 1194-1205.

Rayburn, A.R. and L.A. Schulte. 2009. Landscape change in an agricultural watershed in the U.S. Midwest. *Landscape and Urban Planning* 93(2): 132-141.

Reicosky, D.C. 2015. Conservation tillage is not conservation agriculture. *Journal of Soil & Water Conservation* 70(5): 103A-108A.

Reicosky, D.C. and M.J. Lindstrom. 1993. Tillage-induced CO_2 loss across an eroded landscape. *Agronomy Journal* 85(6): 1237-1243.

Reicosky, D.C., M.J. Lindstrom, T.E. Schumacher, D.E. Lobb, and D.D. Malo. 2005. Fall tillage method: Effect on short-term carbon dioxide flux from soil. *Soil & Tillage Research* 81: 183-194.

Reytar, K., C. Hanson, and N. Henninger. 2014. Indicators of Sustainable Agriculture: A Scoping Analysis. Working Paper, Installment 6 of Creating a Sustainable Food Future. Washington, D.C.: World Resources Institute. Available at http://www.wri.org/sites/default/files/wrr_installment_6_sustainable_agruiculture_indicators.pdf. Accessed June 27, 2016.

Righetti, M. and C. Lucarelli. 2007. May the Shields theory be extended to cohesive and adhesive benthic sediments? *Journal of Geophysical Research* 112(C05039) DOI:10.1029/2006JC003669.

Robertson, G.P., K.L. Gross, S.K. Hamilton, D.A. Landis, T.M. Schmidt, S.S. Snapp, and S.M. Swinton. 2014. Farming for ecosystem services: An ecological approach to production agriculture. *BioScience* 64(5): 404-415.

Ruhe, R.V. 1969. *Quaternary Landscapes in Iowa*. Iowa State University Press, Ames, IA.

Schlesinger, W.H. and J.A. Andrews. 2000. Soil respiration and the global carbon cycle. *Biogeochemistry* 48: 7-20.

Secchi, S. and B.A. Babcock. 2007. Impact of High Crop Prices on Environmental Quality: A Case of Iowa and the Conservation Reserve Program. CARD Working Papers. Paper 471. Ames, IA.

Serbin, G., E.R. Hunt, C.S. Daughtry, G.W. McCarty, and P.C. Doraiswamy. 2009. An improved ASTER index for remote sensing of crop residue. *Remote Sensing* 1(4): 971-991.

Stavi, I. and R. Lal. 2011. Variability of soil physical quality in uneroded, eroded, and depositional cropland sites. *Geomorphology* 125(1): 85-91.

Steiner, K.G. 2002. Producing in harmony with nature through conservation tillage. African Conservation Tillage Network. Information series No. 1, Harare: ACT.

Stott, D.E., D.L. Karlen, C.A. Cambardella, and R.D. Harmel. 2013. A soil quality and metabolic activity assessment after fifty-seven years of agricultural management. *Soil Science Society of America Journal* 77: 903-913.

Tevis, C. 2015. Urgent need to calculate costs of soil erosion. Available at http://www.agriculture.com/farm-management/conservation/urgent-need-to-calculate-costs-of-soil_556-ar47910?print. Accessed March 22, 2015.

Tisdall, J.M. and J.M. Oades. 1982. Organic matter and water-stable aggregates in soils. *Journal of Soil Science* 33: 141-163.

Tornquist, C.G., J. Mielniczuk, and C.E.P. Cerri. 2009. Modeling soil organic carbon dynamics in Oxisols of Ibiruba´ (Brazil) with the CENTURY model. *Soil & Tillage Research* 105: 33-43.

USDA - NRCS. 2002. National agronomy manual. Natural Resources Conservation Service, Washington, D.C.

Van Deventer, A.P., A.D. Ward, P.H. Gowda, and J.G. Lyon. 1997. Using Thematic Mapper data to identify contrasting soil plains and tillage practices. *Photogrammetric Engineering and Remote Sensing* 63: 87-93.

Van Oost, K., G. Govers, T.A. Quine, and G. Heckrath. 2006. Modeling soil erosion induced carbon fluxes between soil and atmosphere on agricultural land using SPEROS-C. In: *Soil Erosion and Carbon Dynamics*, E.J. Roose, R. Lal, C. Feller, B. Barthes, and B.A. Stewart (Eds.). CRC Press, Boca Raton, FL, USA, pp. 37-51.

Van Oost, K., T.A. Quine, G. Govers, S. De Gryze, J. Six, J.W. Harden, J.C. Ritchie, G.W. McCarty, G. Heckrath, C. Kosmas, J.V. Giraldez, J.R. Marques da Silva, and R. Merckx. 2007. The impact of

agricultural soil erosion on the global carbon cycle. *Science* 318(5850): 626-629.

Vereecken, H., J. Maes, and J. Feyen. 1990. Estimating unsaturated hydraulic conductivity from easily measured soil properties. *Soil Science* 149: 1-12.

Vieira, F.C.B., C. Bayer, J.A. Zanatta, J. Dieckow, J. Mielniczuk, and Z.L. He. 2007. Carbon management index based on physical fractionation of soil organic matter in an Acrisol under long-term no-till cropping systems. *Soil & Tillage Research* 96: 195-204.

Wacha, K.M. 2016. Aggregate stability dynamics within intensely managed landscapes: Methods and controls. University of Iowa Dissertation, Iowa City, IA.

Wang, Z., S. Doetterl, M. Vanclooster, B. van Wesemael, and K. Van Oost. 2015. Constraining a coupled erosion and soil organic carbon model using hillslope-scale patterns of carbon stocks and pool composition. *Journal of Geophysical Research - Biogeosciences* 120: 452-465. DOI:10.1002/2014JG002768.

West, T.O., C.C. Brandt, L.M. Baskaran, C.M. Hellwinckel, R. Mueller, C.J. Bernacchi, V. Bandaru, B. Yang, B.S. Wilson, G. Marland, R.G. Nelson, D.G. De La Torre Ugarte, and W.M. Post. 2010. Cropland carbon fluxes in the United States: Increasing geospatial resolution of inventory-based carbon accounting. *Ecological Applications* 20(4): 1074-1086.

Wilson, C.G. A.N. Papanicolaou, and O. Abaci. 2009. SOM dynamics and erosion in an agricultural test field of the Clear Creek, IA watershed. *Hydrologic and Earth System Science Discussions* 6: 1581-1619.

Wischmeier, W.H., and D.D. Smith. 1978. Predicting rainfall erosion losses: A guide to conservation planning. U.S. Department of Agriculture, Agricultural Handbook No. 537. Washington, D.C.

World Bank Group. 2014. State and trends of carbon pricing. International Bank for Reconstruction and Development. Washington, D.C.

Yadav, V. and G.P. Malanson. 2008. Modeling impacts of erosion and deposition on soil organic carbon in the Big Creek Basin of southern Illinois. *Geomorphology* 106(3): 304-314.

Zhang, H., S. Liu, W. Yuan, W. Dong, A. Ye, X. Xie, Y. Chen, D. Liu, W. Cai, and Y. Mao. 2014. Inclusion of soil carbon lateral movement alters terrestrial carbon budget in China. *Science Reports* 4. DOI:10.1038/srep07247.

UCOWR
UNIVERSITIES COUNCIL ON WATER RESOURCES

Mark your calendars!

NIWR
The National Institutes for Water Resources

2017 UCOWR/NIWR Annual Water Resources Conference Fort Collins, CO

June 13-15, 2017
Colorado State University campus



Online abstract submission opens in October.
Abstract deadline is January 20, 2017.

Have an idea for a special session?
Submit a one paragraph proposal to ucowr@siu.edu by September 23, 2016.

For upcoming details:
Check our website (www.ucowr.org) and like us on Facebook (facebook.com/ucowr).

Contents

Letter from the Editors
Karl W.J. Williard and Jackie F. Crim 1

Adaptive Water Governance: Integrating the Human Dimensions into Water Resource Governance
Kofi Akamani 2

Institutional Constraints to Collaborative Ecosystem Management within a Wetlands Conservation Partnership
Jennifer A. Behnken, John W. Groninger, and Kofi Akamani 19

The Effects of Policy Implementation Uncertainty on Groundwater Extraction
Dacheng Bian, Ryan B. Williams, Aaron Benson, and Eduardo Segarra 34

Climate Change, Agriculture and Water Resources in the Southwestern United States
Emile Elias, Al Rango, Ryann Smith, Connie Maxwell, Caiti Steele, and Kris Havstad 46

Irrigation Technology Choice as Adaptation to Climate Change in the Western United States
George Frisvold and Ting Bai 62

Building Watershed Management Capacity in Nigeria: Expanding the Role of Agriculture Colleges
John W. Groninger 78

Heard it through the Grapevine - Social Network Analysis of a Hydrology Class
E. Annette Hernandez and Venkatesh Uddameri 85

Economic Value of Nature and Ecosystems in the Delaware River Basin
Gerald J. Kauffman 98

Water and Society: Interdisciplinary Education in Natural Resources
Laszlo J. Kulcsar, Joseph A. Aistrup, Thomas Bulatewicz, Jeffrey M. Peterson, Stephen M. Welch, and David R. Steward 120

Deliberating on Missouri River Water Diversions in Congressional Committee Hearings
Sarah Michaels and Mike Gruszczynski 132

Dynamic Assessment of Current Management in an Intensively Managed Agroecosystem
Christopher G. Wilson, Kenneth M. Wacha, A.N. (Thanos) Papanicolaou, Heather A. Sander, Violet B. Freudenberg, Benjamin K.B. Abban, and Chang Zhao 148



2017 UCOWR / NIWR
Annual Water Resources Conference
Colorado State University
Fort Collins, CO
June 13-15, 2017

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